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Temperature changes wreak havoc in deforested areas

By Trent Knoss • Feb. 22, 2016

A clear-cut forest area near Eugene, Oregon. Photo by Calibas / Wikipedia.

The newly exposed edges of deforested areas are highly susceptible to drastic temperature changes, leading to hotter, drier and more variable conditions for the forest that remains, according to new research from the University of Colorado Boulder.

The findings suggest that thermal biology—an emerging discipline that examines the effects of temperature on biological and ecological processes—could be an effective tool for understanding how temperature changes in fragmented habitats can potentially wreak havoc on species activity and other critical ecosystem functions.

A study outlining a framework for applying thermal biology to deforestation research was recently published in the journal *Ecology Letters.*

Previous research has shown that widespread deforestation is a threat to global biodiversity, but scientists are only just now beginning to examine the role that temperature may play when trees are removed from an ecosystem—especially near the newly-cut edges of forests.

"When you chop down trees, you create hot spots in the landscape that are just scorched by the sun," said Kika Tuff, a PhD candidate in the Department of Ecology and Evolutionary Biology at CU-Boulder and the lead author of the new paper. "These hot spots can change the way that heat moves through the landscape."

In some cases, this creates a phenomenon known as the 'vegetation breeze,' where low air pressure in the cleared areas pulls the cool, moist air out of the forest and feeds hot, dry air back in.

"So now the cleared areas get all the rain and the forests gets sucked dry," said Tuff.

Kika Tuff

The warming effect is most pronounced between 50 and 100 meters of a forest edge, the study reports, where temperatures can be anywhere from 4 to 18 degrees Fahrenheit warmer than in the forest interior.

"This is like climate change on steroids," Tuff said, "and it happens over much more rapid time scales."

Biologists estimate that 20 percent of the world's remaining forests lie within 100 meters of an edge, while more than 70 percent lie within a kilometer of one. This means that much of the world's forests may be experiencing the vegetation breeze and other warming effects.

Increased temperature variation near forest edges could affect species' ability to regulate their body temperatures, resulting in behavioral changes that could alter the local ecosystem.

One such example, Tuff said, could be the feeding patterns of animals living at the forest edge. Animals are very temperature sensitive, so they hunt for food when they have sufficiently warmed up in the morning and stop hunting for food when it is too hot in the afternoon. If temperatures are higher at the forest edge, species may respond by retreating to hunt in the cooler, deeper forest, where they become dependent on new types of food, sparking a domino effect in the food chain.

Another example might be the timing and duration of species activity. If temperatures were to increase due to tree loss, predators may start foraging later in the day to avoid the heat. Such a change could increase how frequently predators come across their prey, intensifying predation events and resulting in localized prey population crashes in some cases.

Such effects should also apply in instances of natural treefall, Tuff said. In Colorado, for example, high winds, wildfires and beetle kill can create edges in the treeline where greater sunlight exposure would subsequently increase the solar radiation and temperature in localized areas.

The sensitivity of animals and plants to temperature could have implications for future conservation strategies as deforestation and habitat fragmentation continue worldwide.

“Applying thermal biology on the scale of landscapes is a fairly new idea,” Tuff said. “Thermal biology presents a new imperative for forest conservation and makes the value of forests all the greater because of what they do for thermal regulation. Trees aren’t just habitat for animals, they are the world’s insurance for a thermally stable and habitable planet.”

The study was co-authored by Assistant Professor Kendi Davies and graduate student Ty Tuff, both of the Department of Ecology and Evolutionary Biology (EBIO) at CU-Boulder.

Trent Knoss is a science editor at the CU Office of News Services.

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Positive effects of fire on birds may appear only under narrow combinations of fire severity and time-since-fire

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Abstract. We conducted bird surveys in 10 of the first 11 years following a mixed-severity fire in a dry, low-elevation mixed-conifer forest in western Montana, United States. By defining fire in terms of fire severity and time-since-fire, and then comparing detection rates for species inside 15 combinations of fire severity and time-since-fire, with their rates of detection in unburned (but otherwise similar) forest outside the burn perimeter, we were able to assess more nuanced effects of fire on 50 bird species. A majority of species (60%) was detected significantly more frequently inside than outside the burn. It is likely that the beneficial effects of fire for some species can be detected only under relatively narrow combinations of fire severity and time-since-fire. Because most species responded positively and uniquely to some combination of fire severity and time-since-fire, these results carry important management implications. Specifically, the variety of burned-forest conditions required by fire-dependent bird species cannot be created through the application of relatively uniform low-severity prescribed fires, through land management practices that serve to reduce fire severity or through post-fire salvage logging, which removes the dead trees required by most disturbance-dependent bird species.

Additional keywords: Black-backed Woodpecker, conifer forest, ecological integrity, fire severity, mixed-severity fire, restoration, salvage logging, wildfire.

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Introduction

The earliest synthesis of fire effects on birds (Kotliar *et al.* 2002) revealed that many species respond positively, others negatively and still others in a mixed fashion to burned forest conditions. Perhaps the most important pattern that emerged from this synthesis is that some species (the more extreme including the American Three-toed Woodpecker (*Picoides dorsalis*), Black-backed Woodpecker (*Picoides arcticus*), Mountain Bluebird (*Sialia currucoides*) and Tree Swallow (*Tachycineta bicolor*)) are relatively abundant in burned forest conditions. One (the Black-backed woodpecker) is even relatively restricted in its distribution to such conditions. For example, Hutto (1995) reported that 15 species were more abundant in burned forests than they were in any of the other 14 vegetation types included in his meta-analysis. This carries important management implications because those species may depend to a large extent on fire to create the habitat conditions they need for persistence – habitat conditions that are severely compromised by fire prevention, fire suppression, and post-fire salvage logging, seeding, tree planting and removal of native shrubs (Saab and Dudley 1998; Kotliar *et al.* 2002; DellaSala *et al.* 2006; Hutto and Gallo 2006; Hutto 2008; Saab *et al.* 2009; Swanson *et al.* 2011; DellaSala *et al.* 2014; Tingley *et al.* 2014).

Until very recently, studies of fire effects did not distinguish the effects of low-severity, mixed-severity and high-severity fires. Therefore, reported responses of species were oftentimes different from one study to the next, and terms like 'mixed responder' were included in tables generated from synthetic work on fire effects (Kotliar *et al.* 2002). Kotliar *et al.* (2005) noted that fire severity, time-since-fire, vegetation type and other considerations could probably explain some of the variation among studies, but it was not until Smucker *et al.* (2005) characterised the severity of the fire surrounding each of a series of survey points that bird responses to fire became much less ambiguous and remarkably consistent. Smucker *et al.* (2005) proposed that most bird species respond predictably to fire, but that the type of response (positive or negative) depends strongly on fire severity. Subsequently, numerous studies (e.g. Covert-Bratland *et al.* 2006; Kirkpatrick *et al.* 2006; Conway and Kirkpatrick 2007; Koivula and Schmiegelow 2007; Kotliar *et al.* 2007; Hanson and North 2008; Kotliar *et al.* 2008; Vierling and Lentile 2008; Nappi *et al.* 2010; Nappi and Drapeau 2011; Dudley *et al.* 2012; Fontaine and Kennedy 2012; Lee *et al.* 2012; Lindenmayer *et al.* 2014; Rush *et al.* 2012; Hutto *et al.* 2015; Stephens *et al.* 2015) have demonstrated a marked effect of fire severity on either the occurrence or breeding success of selected

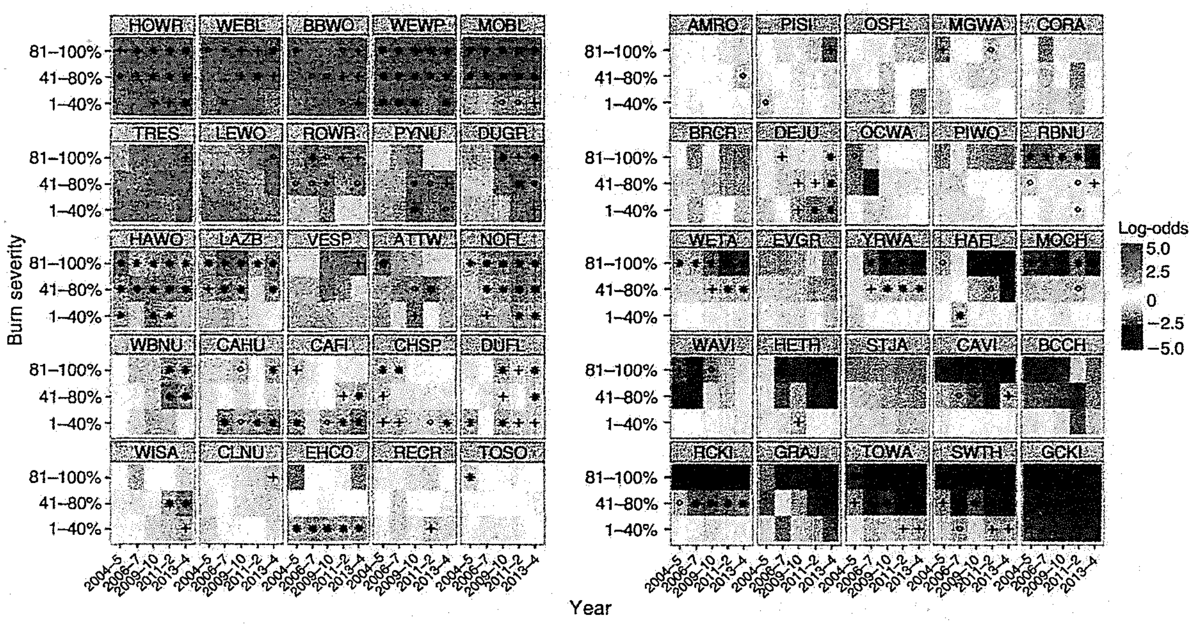

Fig. 2. Heat maps reflecting the log-odds ratio associated with the percentage occurrence in each combination fire severity and time-since-fire in comparison with the percentage occurrence in unburned forest outside the fire perimeter for each of 50 bird species (four-letter mnemonic codes provided in Table 2; species are organised by their average log-odds scores, from those that had a large average positive response to those that had a large average negative response to fire). Hotter (more red) blocks represent positive responses to fire and cooler (more blue) blocks represent negative responses to fire. The symbols correspond with Bonferroni adjusted P-values ($\Diamond = 0.01 < P < 0.05$; $+ = 0.001 < <0.01$; $* = P < 0.001$). Thirty of 50 species (60%) were significantly more abundant in burned forest at some combination of severity and time-since-fire than in unburned, mature green-tree forest.

Golden-crowned Kinglet (*Regulus satrapa*)) were detected less frequently after fire, regardless of fire severity, and their detection rates generally continued to decrease over time (Fig. 2).

Discussion

Following the most common approach to assessing fire effects, we first looked at whether there were significant differences in bird abundances between the burned and surrounding unburned forest. The results from this analysis were consistent with those reported in many other studies of fire effects on birds (see Kotliar *et al.* 2002) – roughly half the bird species appeared to benefit and half did not (Table 2). Unfortunately, this kind of analysis hid positive responses that became apparent only after accounting for fire severity and time-since-fire. By dividing the burned-forest data into 15 combinations of fire severity and time-since-fire, we found results that were more nuanced than those obtained from a simple 'burned vs unburned' analysis. Specifically, 30 of 50 (60%) of the bird species considered were significantly more likely to occur inside the burned forest (at 1 or more combinations of fire severity and time-since-fire) than outside the burned forest. The distinct location of the greatest probability of detection for any 1 species across the 2-dimensional fire-severity and time-since-fire gradient, combined with differences in those locations among species (Fig. 2), suggests that the bird occurrence patterns are accurate reflections of bird abundance and not artefacts of some kind of sampling bias that might affect all species similarly. Other recent work (Stephens *et al.* 2015) has also revealed that the locations of peak abundances across a fire-severity/time-since-fire gradient differ among species.

Many of these significantly positive responses would not have been evident without partitioning the data into multiple severity and time-since-fire categories. This kind of analysis is difficult to conduct with data from any one fire because sample sizes (the number of independent survey points in each severity-by-year category) are generally much smaller than what we were able to achieve here (Table 1). Even with the sample sizes we achieved, we were still forced to use fewer categories than the number used in the field to assess the statistical significance of fire effects. Although each bird species responded uniquely to the combination of fire severity and time-since-fire (Fig. 2), four general classes of response are worth noting, along with some of the most probable biological underpinnings behind each.

Response Pattern 1

This pattern is illustrated by species that showed an abrupt increase in abundance within the first few years following fire, and the elevated abundance persisted until the end of the 11-year study primarily (but not exclusively) in locations that burned at higher severities. Several woodpecker species (Black-backed Woodpecker, Hairy Woodpecker, American Three-toed Woodpecker and Northern Flicker) showed this response pattern (Fig. 2). The biological basis behind the abrupt increase in woodpecker populations is well established and unambiguous: bark and wood-boring beetle populations increase as individual beetles detect the newly created abundance of fire-killed trees.

In turn, developing larvae provide food for woodpeckers, which then respond numerically with an abrupt increase in abundance during the first year or two after fire. Relatively rare species like the American Three-toed and Black-backed woodpeckers are known to appear within months after fire (Blackford 1955; Villard and Beninger 1993; Villard and Schieck 1997). Their abundances then increase at a rate that is more rapid than expected if the increase were entirely the result of recruitment from inside the burn, so the increases are undoubtedly associated with colonisation by birds from outside the burned area, as others have noted (e.g. Van Tyne 1926; Koplin 1969; Yunick 1985; Hoyt and Hannon 2002; Huot and Ibarzabal 2006; Siegel *et al.* 2016). Perhaps the most iconic indicator of a severely burned mixed-conifer forest throughout the Sierra Nevadas of California, the Intermountain West and Canadian boreal forest is the Black-backed Woodpecker (Hutto 1995; Hanson and North 2008; Hutto 2008; Hutto *et al.* 2008; Nappi and Drapeau 2009; Swanson *et al.* 2011; Bond *et al.* 2012; Hutto *et al.* 2015). Although its pattern of response to fire is not much different than that of the American Three-toed Woodpecker, Hairy Woodpecker or Northern Flicker (Fig. 2), the key difference between this woodpecker and the others lies with the extent to which the species is restricted in its distribution to burned forest conditions. Other woodpecker species occur in green-tree forests to a much greater extent than the Black-backed Woodpecker; hence, the Black-backed Woodpecker is a better 'indicator' of severely burned forest conditions. The relatively high Black-backed Woodpecker occurrence rates across an 11-year period is somewhat surprising given the existing literature (Apfelbaum and Haney 1985; Hutto 1995; Murphy and Lehnhausen 1998; Hobson and Schieck 1999; Hoyt and Hannon 2002; Saab *et al.* 2007), which suggests that a 4–8-year window of opportunity is about all one can expect for this species before they begin to decline in abundance. It may very well be that the mixed-severity fire allowed this species to persist longer in this study than in most others because there was an abundance of weakened trees in the moderate- to low-severity border areas, which continued to provide adequate food resources after conditions in the more severely burned portions became less suitable. This mechanism of persistence beyond durations expected on the basis of averages gleaned from the literature has been suggested to operate elsewhere as well (Nappi *et al.* 2010; Dudley *et al.* 2012).

Other bird species (e.g. Cassin's Finch; Clark's Nutcracker, *Nucifraga columbiana*; Red Crossbill; Pine Siskin) that appear to respond abruptly and positively to more severe fire, do so for reasons that are most likely related to the abrupt increase in availability of seeds that are retained in cones that open in response to fire. Still other species in this group respond quickly to what is probably an increase in the availability of the combination of nest sites associated with standing-dead trees and open areas for foraging either on the ground (e.g. Western Bluebird and Mountain Bluebird) or in the air (e.g. Western Wood-Pewee) or to an increase in the availability of relatively predator-free nest sites at ground level from burned-out roots and root wads associated with wind-thrown trees (e.g. Rock Wren; Townsend's Solitaire; Dark-eyed Junco, *Junco hyemalis*). The Olive-sided Flycatcher (*Contopus cooperi*) makes special use of nest sites that are located in green-needled or brown-needled trees on the edges of severely burned forest where they can sally into the openings created by crown fire.

Response Pattern 2

This pattern is illustrated by species that showed a slightly delayed increase in abundance, primarily (but not necessarily) in locations that burned at higher severities. If we had confined the study to the first year or two after fire, these positive responses to fire would have gone undetected. One of these species (Vesper Sparrow) appeared after a brief delay in the development of suitable grass-dominated patches, while several others (e.g. Dusky Grouse; Lazuli Bunting; Orange-crowned Warbler, *Oreothlypis celata*) were probably responding to the development of suitable shrub and seedling layers for foraging and nesting, which varied markedly from one place to another within the burn perimeter. Still other species (e.g. House Wren, *Troglodytes aedon*; Lewis's Woodpecker; Tree Swallow; Williamson's Sapsucker) were probably responding to a delayed increase in the availability of cavity nest sites that became available either after they were excavated by woodpeckers soon after fire or after the larger standing-dead trees began to soften with decay and break more easily in wind events. The House Wren clearly benefitted from tree blowdown and breakage events that occurred in the first 6 years after fire; its abundance grew steadily to the point that it was the most abundant species in severely burned forest patches 6–11 years after fire.

Several authors (e.g. Hutto 1995; Gentry and Vierling 2007; Saab *et al.* 2007) have recognised a delayed increase in populations of Lewis's Woodpecker following severe fire, and it is clear that the benefit of severe fire to this woodpecker species would not have been detected in this study had the data collection period been restricted to the first few years after fire. The delayed positive responses of other species (e.g. Williamson's Sapsucker and White-breasted Nuthatch) have not been described previously and they are notable, as we discuss more fully below. Williamson's Sapsucker is noteworthy in that its distribution was nearly restricted to edge conditions between unburned forest patches (where it fed on sap of living trees) and adjacent burned forest patches (where it nested in relatively large, decayed or broken-topped trees that became abundant after wind events in severely burned patches created them a decade or so after fire). The delayed increase in White-breasted Nuthatch detections may be related to the increase in availability of insects beneath thick bark after the bark began to peel away from, or slough off, dead Ponderosa Pine, Western Larch, and Douglas-fir. Finally, the delayed increase by Western Bluebird represents a pattern that has received considerable attention (Duckworth and Badyaev 2007; Duckworth 2008, 2009; 2010, 2012; 2014); Western Bluebirds are not as well adapted as Mountain Bluebirds are to colonise burned forests early on after fire, but once they colonise, they are better suited to outcompete the Mountain Bluebird in severely burned hot spots.

The pattern of a rapid increase in abundance illustrated by species listed under the first response pattern is well described and well appreciated, but the second pattern of a delayed increase in severely burned forest patches is not widely appreciated because the use of a single 'after fire' category in many previous fire effects analyses may have hidden the influence of a delayed response. For example, of the 11 published studies of bird communities in burned and unburned forests reviewed by Kotliar *et al.* (2002), none separated the effects of fire severity or time-since-fire. Since that time, numerous authors (Hannah and Hoyt 2004; Saab *et al.* 2004; Smucker *et al.* 2005; Kirkpatrick *et al.* 2006; Schieck and Song 2006; Kotliar *et al.* 2007; Saab *et al.* 2007; Hutto 2008; Kotliar *et al.* 2008; Pons and Clavero 2010; Nappi and Drapeau 2011; Saracco *et al.* 2011; Stephens *et al.* 2015) have included either fire severity or time-since-fire in their analyses of fire effects, and all of these authors concluded that it is important to do so. This study serves to reinforce the idea that fire effects cannot be accurately assessed in the absence of knowledge about the context surrounding a particular sample location and that includes, but is not limited to, fire severity and time-since-fire.

Response Pattern 3

This pattern is illustrated by species that revealed a fairly abrupt or slightly delayed increase in abundance within the first year or two following fire, but the positive response is limited to locations that burned at lower severities. This group includes species that have previously been labelled as 'mixed' responders in meta-analyses of fire effects (e.g. Kotliar *et al.* 2002) because they respond positively to fire in some studies and negatively to fire in others. Our results suggest instead that these species do not respond to fire unpredictably; it is just that their response depends on fire severity. For example, some of these species maintained an elevated occurrence rate in locations that burned at a lower severity throughout the duration of the 11-year study (e.g. Brown-headed Cowbird, Red Crossbill, Red-breasted Nuthatch, Western Tanager and Ruby-crowned Kinglet), while others showed a positive but brief response to low-severity fire only for a brief period following fire (e.g. Common Raven; Brown Creeper, *Certhia americana*; Pileated Woodpecker; Evening Grosbeak; Yellow-rumped Warbler; Hammond's Flycatcher; Hermit Thrush). For those species that showed a sustained (albeit small) positive response to low-severity fire, a predominantly green-tree forest that burned recently at low severity might actually provide a forest condition that is more suitable than a long-unburned green-tree forest. The species that revealed a brief and limited increase in abundance only during the first year or two following fire may reflect an influx of individuals returning from wintering locations only to find many of their previously occupied locations too severely burned, so they then proceeded to squeeze into unburned or lightly burned forest near previously occupied forest patches. If true, their relatively high abundance in mildly burned forest may not reflect suitable conditions, but may be a reflection of birds making the best of a bad situation. A key question for future fire research would be to determine whether increases in numbers of detections after recent fire in mildly burned green-tree stands reflect conditions that are better in quality, even if only briefly, than long unburned green-tree stands for this group of species.

Response Pattern 4

This pattern is illustrated by the six species that were less abundant in burned than in unburned forest immediately following fire and appeared to become even less abundant across the 11-year time span. For each of these species, there was no combination of fire severity and time-since-fire that resulted in detection rates that were as great as they were in unburned forest. Perhaps these species do not benefit in any way from fire. Indeed, many authors would be quick to classify the species associated with this group of six species as 'negative responders.' However, we know that the abundances of each of these six species will be lower in recently burned forests only in the short-term. This was only an 11-year study, and we know that these same species will reach their highest occurrence rates in the years to come. If those occurrence rates reach a peak at some point in the future before falling again when the forest reaches an even older age, then those species would also be more accurately classified as positive responders; it is just that the time required to show a relatively high abundance is longer than the length of the present study. Even a rudimentary knowledge of natural history suggests that many bird species (e.g. Orange-crowned Warbler, MacGillivray's Warbler, Calliope Hummingbird and Lazuli Bunting) benefit from high-severity fire, but the greatest positive response cannot usually be detected until somewhere between 10 and 30 years following severe fire when plant succession produces a high density of conifer seedlings and shrubs. Similarly, the six species in this last group could be said to benefit from fire if forest conditions associated with, hypothetically, a 300-year-old forest are not as good as forest conditions associated with a 100-year-old forest because the only way optimal conditions for these species could be 'restored' would be through a severe fire event that creates their preferred habitat 100 years later. Interestingly, Taylor and Barmore (1980; Table 2) showed precisely that pattern for Ruby-crowned Kinglet, Yellow-rumped Warbler and Hermit Thrush in Yellowstone and Grand Teton National Parks – those species that are more abundant in century-old than in either younger or older forests. Using the same logic, even a species that is more abundant at year 300 than at year 700 following fire would mean that severe disturbance (disturbance severe enough to trigger ecological succession) is necessary to 'restore' appropriate forest conditions for that species too (see also Imbeau *et al.* 1999, Schieck and Song 2006; Zhao *et al.* 2013). The important point is that we cannot assess the effects of fire without data on bird abundances from a more extended series of forest ages following fire.

Caveat and management implications

Because the bird occurrence rate at unburned points was based on a different set of years than the occurrence rate for any one of the two-year post-fire samples, it is possible that a significant difference in the abundance of any one species between the unburned and a particular fire severity/time-since-fire category resulted from a temporal change in abundance independent of fire effects. However, even though our results emerged from a single fire event that may be best considered a case study, a previous study of bird occurrence patterns across hundreds of fires over the past 30 years (see Hutto *et al.* 2015, and references therein) suggests that the individual species' responses make good biological sense and are not artefacts of unusual increases or declines in bird abundance independent of fire effects. Nonetheless, these findings should serve as hypotheses to be tested through replication of independent samples drawn from different time-since-fire categories. Only through use of a chronosequence approach (Hutto and Belote 2013) will we be able to generate the replication of burned-forest conditions necessary to evaluate fire effects in a manner that will allow us to appreciate how both fire severity and time-since-fire create the conditions needed by birds that respond positively to fire.

The take-home lesson here is that we cannot rely on traditional 'burned vs unburned' comparisons presented in most published reports on fire effects to assess whether species respond positively or negatively to fire. Fire severity, time-since-fire and other forest conditions matter to organisms that respond positively to disturbance, therefore we will have to consider the kind of forest, tree sizes and densities, fire severity and time-since-fire if we want to investigate fire effects in a biologically meaningful way. By considering the effects of just two of those forest conditions here, it is clear that the majority of species increases in abundance during part or all of the first dozen years after fire, as evidenced by significant differences in rates of detection inside vs outside burned forests. Thus, we cannot gain a thorough understanding of fire effects through results that emerge from short-term, before-and-after studies; we need to know more about the occurrence rates of species across very long time spans if we are to speak knowledgably about the effects of fire on any particular species. Our finding that 60% of the bird species surveyed are most abundant in some stage of forest succession following fire than they are in mature forest is undoubtedly a conservative estimate of the proportion of species that benefit from fire because our data cover only a relatively short 11-year period following fire.

Not only are there unambiguous responses by the majority of bird species to fire, but the responses are also highly dependent on a spatial component associated with fire severity. As suggested by earlier work that included fire severity as an independent variable (e.g. Smucker *et al.* 2005; Kotliar *et al.* 2007, 2008; Vierling and Lentile 2008; Stephens *et al.* 2015), fire severity has a dramatic influence on the probability of occurrence of bird species. The same pattern was true here for nearly all species that were detected on at least 10 points. Some species are clearly most abundant in the less severely burned forest patches, while others are clearly most abundant in the more severely burned patches. This result is important because it implies that mixed-severity fire effects are necessary for the creation of conditions needed by the variety of bird species that respond positively to fire. Even more importantly, the variety of burned-forest conditions favoured by different bird species may be difficult to create through a prescription of low-severity understory fire applied outside the normal fire season, because such fires do not generate the higher-severity patches needed by the species that are relatively restricted to forests that have burned severely (Hutto 2008). In addition, land management practices designed to prevent or eliminate severe fire will also eliminate the very conditions required by many of the species highlighted here, as will post-fire salvage logging, which has been documented to have overwhelmingly negative effects on birds – effects that are among the strongest and most consistent scientific results ever published on any wildlife management issue (Hutto *et al.* 2015).

Ecologists have long known that severe, stand-replacement fires are characteristic of some (mostly subalpine) forest types (Brown and Smith 2000), but many forest managers and most politicians along with the public at large are still remarkably uninformed about the naturalness and necessity of severe fire in vegetation types born of, and maintained by, severe fire (Hutto *et al.* 2016). Even more striking is the near absence of an appreciation for the naturalness and necessity of severe fire in the low- to mid-elevation mixed-conifer forest types, even though severe fire events that create patches of severely burned forest are a natural and important part of those forests too (Baker *et al.* 2007; Hessburg *et al.* 2007; Baker 2009; Margolis *et al.* 2011; Perry *et al.* 2011; Baker 2012; Heyerdahl *et al.* 2012; Marlon *et al.* 2012; Veblen *et al.* 2012; Williams and Baker 2012*a*, 2012*b*; Odion *et al.* 2014; Sherriff *et al.* 2014; Williams and Baker 2014; Baker 2015*a*, 2015*b*; Baker and Williams 2015; Yocom-Kent *et al.* 2015; Hutto *et al.* 2016). For the lower-elevation mixed-conifer forest types, such as the forest studied here, it is quite clear that some amount of severe fire is natural and that large numbers of bird species benefit from the severe-fire component. Given these results, the challenge is to educate land managers, politicians and the public at large about the importance of maintaining severe fire on the landscape and to design fire-safe communities that can withstand the effects of severe fire disturbance events (Hutto *et al.* 2016). Only then will we be in a position to manage for appropriate amounts and sizes of severely burned forest patches that occurred historically and to celebrate the creation of mosaics of different fire severities and post-fire ages that follow directly from severe fire as a natural disturbance process.

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Composition of Bird Communities Following Stand-Replacement Fires in Northern Rocky Mountain (U.S.A.) Conifer Forests

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Abstract: *During the two breeding seasons immediately following the numerous and widespread fires of 1988, I estimated bird community composition in each of 34 burned-forest sites in western Montana and northern Wyoming. I detected an average of 45 species per site and a total of 87 species in the sites combined. A compilation of these data with bird-count data from more than 200 additional studies conducted across 15 major vegetation cover types in the northern Rocky Mountain region showed that 15 bird species are generally more abundant in early post-fire communities than in any other major cover type occurring in the northern Rockies. One bird species (Black-backed Woodpecker,* Picoides arcticus*) seems to be nearly restricted in its habitat distribution to standing dead forests created by stand-replacement fires. Bird communities in recently burned forests are different in composition from those that characterize other Rocky Mountain cover types (including early-successional clearcuts) primarily because members of three feeding guilds are especially abundant therein: woodpeckers, flycatchers, and seedeaters. Standing, fire-killed trees provided nest sites for nearly two-thirds of 31 species that were found nesting in the burned sites. Broken-top snags and standing dead aspens were used as nest sites for cavity-nesting species significantly more often than expected on the basis of their relative abundance. Moreover, because nearly all of the broken-top snags that were used were present before the fire, forest conditions prior to a fire (especially the presence of snags) may be important in determining the suitability of a site to cavity-nesting birds after a fire. For bird species that were relatively abundant in or relatively restricted to burned forests, stand-replacement fires may be necessary for long-term maintenance of their populations. Unfortunately, the current fire policy of public land-management agencies does not encourage maintenance of stand-replacement fire regimes, which may be necessary for the creation of conditions needed by the most fire-dependent bird species. In addition, salvage cutting may reduce the suitability of burned-forest habitat for birds by removing the most important element—standing, fire-killed trees—needed for feeding, nesting, or both by the majority of bird species that used burned forests.*

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Composición de las comunidades de aves luego del reemplazo de rodales a causa de incendios forestales en bosques de coníferas de las montañas Rocosas del norte

Resumen: *Durante las dos últimas temporadas de cría immediatamente despúes de los numerosos y extensos incendios de 1988, estimé la composición de la comunidad de aves en cada uno de los sitios de bosques incendiados, en el oeste de Montana y el norte de Wyoming. Detecté un promedio de 45 especies por sitio y un total de 87 especies en todos los sitios combinados. Una recopilación de estos datos con otros de conteo de aves a partir de más de 200 sitios adicionales, conducido a lo largo de 15 tipos principales de cobertura de vegetación en las montañas Rocosas del norte mostró que 15 especies de aves eran en general más abundantes en las comunidades tempranas posteriores al incendio, que en cualquier otro tipo principal de cobertura presente en las Rocosas del norte. Una especie de ave (el pájaro carpintero de espalda negra,* Picoides arcticus*) parece estar restringida en su distribución a los árboles muertos en pie, que quedan a causa del reemplazo de rodales a partir de los incendios. Las comunidades de aves en los bosques recientemente incendiados, son diferentes en composición de aquellos que caracterizan otros tipos de cobertura de las montañas Rocosas (in-*

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for a very limited number of habitat types, however (for example, low-elevation ponderosa pine forests). Most of the forested landscape in the northern Rockies evolved under a regime of high-intensity, large fires every 50-100 years (Fischer & Bradley 1987), not under a regime of low-intensity, frequent understory burns. A study of fire history in the Selway-Bitterroot Wilderness showed that less than 10% of the forested land experienced nonlethal fire; most of the forest types experienced partly to completely lethal fires every 100-200 years (Brown et al. 1994). Although some might argue that all forest types have been subjected to fire suppression for too long and that unnaturally dense understory buildups are leading to unnaturally severe fires, the stand-replacement fires that currently consume forests that evolved under that regime (for example, the 1988 Yellowstone fires) are not at all unusual in intensity or extent (Romme & Despain 1989).

Second, current human population and human settlement trends allow for the retention of very few areas large enough to allow free-ranging fire, and almost none of those areas have prescriptions allowing stand-replacement fires to occur (Agee 1991). Even when there is plenty of space to let fires burn, the general response is to expend enormous resources to eradicate fire because of the damage it does to timber resources, the danger it poses to humans and their buildings, and—despite ample evidence to the contrary—the damage it may do to tourism because of the visual impact. Brown and Arno (1991) have addressed this growing predicament of putting fire back into the landscape while still operating within the economic, social, and political constraints that society continues to impose: It will not be easy.

Third, there is a lack of public education about the benefits of stand-replacement fires. The biological naivete surrounding the 1988 fires was astounding and did more to muster opposition than support for "let it burn" wilderness policies. The lack of understanding demonstrated by the public, especially prominent politicians, generated a good bit of the conflict over policy (Cutler 1988). Simple facts—for example, there exists a strong distributional association between some bird species and burned forests—should be used to garner support from the public for liberal prescribed-fire policies.

Fourth, forests are not being managed in ways that mimic natural processes. One could argue that the loss of burned forest acreage due to fire control has been compensated for, at least in part, by timber harvesting. Many people believe that the conditions present after a clearcut are basically the same as those present after a severe fire (Kohrt 1988; Maschera 1988; Eggleston 1989; Swift 1993). But conditions created by a stand-replacement fire are biologically unique, at least in terms of the biomass of standing, dead trees that remain and, to a much greater extent, in terms of ecosystem structure and function. Clearcutting is, in general, a poor substitute for fire because such timber harvesting does not retain some of the most important elements, such as standing, dead trees, that are integral components of the post-fire ecosystem and that probably contribute to unique successional pathways (Agee 1991; Hansen et al. 1991) and wildlife communities.

Stand-replacement fires should not be viewed as unnatural disasters that can (and should) be prevented (Kipp 1931). As Heinselman (1985) has argued, plans to maintain stand-replacement fire regimes are justified in at least the more remote of our public lands, and prescribed-fire regimes should not be limited to periodic, mild, understory burning in lower-elevation ponderosa pine forests. Managers must also be careful to mimic all aspects of natural disturbance (such as timing, frequency, and intensity) and not just introduce disturbance as such (Hobbs & Huenneke 1992). Finally, because the pattern of relative bird abundances differs among burns, managers probably need to provide a diversity of burned cover types, intensities, and maybe even a variety in landscape contexts of burns to provide for the variety of species that may depend on fire.

Post-fire Timber Harvesting

On public lands, managers should leave an adequate amount of standing, dead trees after a fire because of the species that depend on that forest element. The current tendency to expedite timber "salvage" sales on burned forest lands needs to be re-examined. Already, as much as 60% of all timber sales on some forests in the Northern Region of the U.S. Forest Service come from salvaged timber (Schwennesen 1992). These sales, which are often exempt from public notice or comment, are generally supported by a well-meaning but misguided public that believes "dead and dying timber ought to be harvested and put to use" (Schwennesen 1992).

If some bird species require burned forests for the maintenance of viable populations (which is strongly suggested by this study), then post-fire salvage cutting may be conducted too frequently to be justified on the basis of sound ecosystem management. In instances where a salvage cut is deemed necessary, managers who wish to mitigate such effects by leaving some of the standing dead trees should be aware that bird species differ in the microhabitats they occupy within a burn. Therefore, methods that tend to "homogenize" the stand structure (such as selective removal of all trees of a certain size and/or species) will probably not maintain the variety of microhabitats and, therefore, bird species that would otherwise use the site. Selective tree removal also generally results in removal of the very tree species (Table 4) and sizes (Table 5) preferred by the more fire-dependent birds. It may be best, instead, to take trees from one part of the burn and leave another part of the burned area untouched. That way, some of the guess-

Table 4. The numbers of seven species of conifers (>10 cm diameter at breast height) encountered along a series of transects in the Grant Village, North Fork, Canyon Creek, and Blackfoot-Clearwater sites, and the percentages of those used by woodpeckers for feeding purposes.

Tree Species	*(n)*	*Woodpecker use (%)**
Ponderosa pine, *Pinus ponderosa*	297	80.5
Western larch, *Larix occidentalis*	100	64.0
Douglas-fir, *Pseudotsuga menziesii*	593	47.9
Engelmann spruce, *Picea engelmanni*	109	2.3
Lodgepole pine, *Pinus contorta*	647	0.2
Subalpine fir, *Abies lasiocarpa*	172	0.0

**Percentages differ significantly among tree species (G = 1081, p = 0.000).*

Table 6. Number (%) of cavity and open-cup nests in each of six classes of potential nest sites.

Nest Site	*Cavity Nests*	*Open-Cup Nests*	*Available (%)**
Broken-Top Conifer	15 (31)	3 (14)	6
Intact-Top Conifer	12 (25)	9 (44)	92
Broken-Top Aspen	2 (4)	0 (0)	0
Intact-Top Aspen	18 (38)	0 (0)	2
In Bank, On Ground	1 (2)	8 (38)	n/a
In Shrub	0 (0)	1 (5)	n/a

**Based on a sample of 200 trees along a single, 10-m-wide transect in the Canyon Creek site.*

species are not the same as those that best predict the presence of another. Accordingly, the single variable that shows the best partial correlation with bird abundance varies widely among species (Table 7).

Discussion

Contrary to what one might expect to find immediately after a major disturbance event, I detected a large number of species in forests that had undergone stand-replacement fires. Huff et al. (1985) also noted that the density and diversity of bird species in one- to two-year-old burned forests in the Olympic Mountains, Washington, were as great as in adjacent old-growth forests. These numbers are not an artifact of birds simply passing through on their way from one adjacent unburned area to another. Most species we detected were feeding in the burned forests, and at least a third (36%) of those detected were nesting therein as well. If the birds were merely feeding while passing through, I should have detected more species and individuals in small burns and fewer in large burns because the probability of passage should decrease with increased isolation from unburned source areas. In fact, the presence of a species was largely independent of burn size; in only two cases (Townsend's Solitaire [*Myadestes townsendi*] and Solitary Vireo [*Vireo solitarius*]) was bird abundance significantly negatively correlated with burn size, and those species may indeed have been present in the smaller burns because of the proximity of unburned forest to some of the census points.

Several bird species seem to be relatively restricted in distribution to early post-fire conditions. These include Olive-sided Flycatcher, Three-toed Woodpecker, Black-backed Woodpecker, Clark's Nutcracker [*Nucifraga columbiana*], and Mountain Bluebird [*Sialia currucoides*]. Although none of these species may be considered an early post-fire obligate in the strictest sense, few strict obligates are associated with any habitat (Niemi & Probst 1990). I believe it would be difficult to find a forest-bird species more restricted to a single vegetation cover type in the northern Rockies than the Black-backed Woodpecker is to early post-fire conditions. Although it is possible that Black-backed Woodpecker populations are maintained by source refuges of low numbers in unburned forests, it is equally likely that their populations are maintained by a patchwork of recently burned forests. The relatively low numbers in unburned forests may be sink populations that are maintained by birds that emigrate from burns when conditions become less suitable 5–6 years after a fire.

5-6

Table 5. The sizes of each of three species of trees used by woodpeckers for feeding purposes in the Blackfoot-Clearwater site.

	Tree Diameter at Breast Height (cm)					
Tree Status	*0–10*	*10–20*	*20–30*	*30–40*	*>40*	*Significance**
Douglas-fir, *Pseudotsuga menziesii*						
not fed upon	269	180	77	9	0	
fed upon	10	70	123	24	10	0.0000
Ponderosa pine, *Pinus ponderosa*						
not fed upon	261	39	17	1	1	
fed upon	72	175	48	7	9	0.0000
Western Larch, *Larix occidentalis*						
not fed upon	16	4	0	0	0	
fed upon	11	30	3	0	0	0.0001

**Based on G-test of independence between tree size and signs of feeding activity.*

100 mi

ern Flicker [*Colaptes auratus*], Steller's Jay [*Cyanocitta stelleri*], Orange-crowned Warbler [*Vermivora celata*], and Chipping Sparrow [*Spizella passerina*]) were most abundant in the slightly older burned forests (10–40 years after fire) (Table 3). Three species (American Robin, Yellow-rumped Warbler [*Dendroica coronata*], and Dark-eyed Junco) were detected in both early- and mid-successional burned forest studies 100% of the time. Thus, burned forests may be of critical importance to a large number of Rocky Mountain bird species that are either relatively restricted to or relatively abundant in such forests.

The picture I paint of bird communities in burned forests contrasts sharply with that painted by other authors (Emlen 1970; Bendell 1974; Lyon et al. 1978; Niemi 1978; Lyon & Marzluff 1985), who have stated that bird communities change little after fire. After a careful review of those papers and the papers that those authors summarized, however, it is clear that the no-effect conclusions have emerged, in part, from studies of low-intensity fires or nonforested habitats and almost always from comparisons of one or two study sites and one or two controls—far too little replication to draw general conclusions about fire effects. Most important, however, the no-effect conclusions are based on composite statistics such as total bird density, species richness, and within-guild abundances, which hide more than they reveal in terms of biological effects of fire on specific species.

Bird species that use burned forests occupy a variety of feeding guilds and most rely heavily on the standing dead trees for food acquisition. For example, several bird species detected in recently burned forests may be taking advantage of the increased availability of conifer seeds after cones open in response to fire. Seed eaters that feed on conifer seeds (especially Clark's Nutcracker, Cassin's Finch [*Carpodacus cassinii*], Red Crossbill [*Loxia curvirostra*], and Pine Siskin [*Carduelis pinus*]) were more abundant in early post-fire habitat than in any other cover type, and they were significantly more abundant (Mann-Whitney $U = 29{,}568$, $p < 0.001$) in the first year than in the second year following a fire, when conifer-seed resources would have been more depleted. Another feeding group that seems to depend on food provided by the burned trees includes the bark-probing woodpeckers, which eat primarily wood-boring beetles (Beal 1911). Woodpeckers are clearly responding to the increase in availability of cerambicid and buprestid beetle larvae (Evans 1964; Komarek 1969; Bock & Bock 1974; Fellin 1980; Harris 1982; Amman & Ryan 1991), which in some cases are themselves responding to the increase in availability of unburned wood that lies beneath the bark of fire-killed trees (Amman & Ryan 1991). Adult beetles in the genus *Melanophila* are, in fact, specialized to feed on fire-killed trees and are capable of using infrared sensors to detect and colonize burned forests more than 161 km distant (Evans 1964, 1966). Finally, aerial insectivores (flycatchers, swallows) relied on standing dead trees as perch sites from which they sallied into the open air space for their prey.

Because the pattern of relative bird abundances differed among sites, the relative suitabilities of sites probably also differed among bird species. The same conclusion is suggested by results of the partial correlation analysis, in which the specific elements associated with bird abundance differ among species.

Most (77%) of the bird species I detected in burned forests were migrants. With concern about declining populations of migrants (Askins et al. 1990), perhaps conservation biologists should be devoting more attention to the loss of early successional habitats born of "natural" disturbance by investigating the extent to which such habitats are necessary for the maintenance of viable populations.

Conservation and Management Implications

The Importance of Stand-Replacement Fires

Fires are clearly beneficial to numerous bird species and are apparently necessary for some. The same case has been made for plants, in which some species germinate and flower only within 1–3 years after a fire and then bank their seeds for storage until the next fire (Heinselman 1981). Fire is such an important creator of the ecological variety in Rocky Mountain landscapes (Arno 1980; Gruell 1983) that the conservation of biological diversity is likely to be accomplished only through the conservation of fire as a process. Fire is in fact ". . .the only natural agent that is sufficiently widespread, abundant, fast, and regular to hold plant successions in seral stages on a vast scale and, therefore, to maintain the diversity of animal life that is so dependent upon such successional vegetation" (Komarek 1966). Efforts to meet legal mandates to maintain biodiversity should, therefore, be directed toward maintaining processes like fire, which create the variety of vegetative cover types upon which the great variety of wildlife species depend (Hansen et al. 1991).

Unfortunately, we are not currently managing the land to maintain the kind of early successional seral stages that follow stand-replacement fires and, hence, many fire-dependent plant and animal species. Why not? First, prescribed fires in conifer forests are most often low-intensity, understory burns that are justified by the argument that, with past fire prevention, forest composition is now "unnatural" and that we need to reintroduce a native fire regime of frequent, mild, understory burns to restore forests and to prevent catastrophic crown fires, which are "destructive" and "unnatural" (Biswell 1968; Alexander & Dube 1982). This justification holds only work associated with choosing what to leave is avoided. This is clearly an area that deserves additional research attention.

Implications for Live-Tree Harvesting Methods

It is unfortunate that the effect of a timber harvesting method on birds (and other vertebrates) is nearly always evaluated in terms of how much the bird community composition changes from before to immediately after harvest (Hutto et al. 1993; Hejl et al. 1995). The method that best mitigates immediate harvest effects (that produces the least change) is generally viewed as the best alternative. Instead, maybe managers should favor methods that minimize deviation not from the bird communities typically associated with the pre-cut forest, but from those associated with the series of post-fire successional communities anticipated to have eventually occurred on that particular plot of land. In this light, many of the "new forestry" thinning practices, which appear favorable in terms of mitigating the immediate effects of cutting, may not represent the best strategy in terms of minimizing the impact of timber harvesting on natural patterns and processes. This is because many of the newer harvesting practices in mid- to high-elevation conifer forests create structurally artificial stands of thinned trees, which may bring "unnatural" combinations of bird species together, eliminate the full range of seral stages, and, perhaps worst of all, reduce the prospect of fire in the future (Gruell 1980). Recent full-page ads by the timber industry in the northern Rocky Mountains (for example, Missoulian, 24 August 1994, p. A-10), have, in fact, emphasized the fire-prevention "benefit" of forest thinning. Such a consequence may be fine at the urban-forest interface. It may be a well-intentioned but misplaced goal, however, for forested wildlands.

Most selective harvesting and thinning methods also result in the loss of large trees, many of which are otherwise destined to become the kind of snags that many primary and secondary cavity nesters depend on for nesting purposes should a stand-replacement fire occur. The predominant use of already existing snags by cavity nesters in burned forests (Table 6) implies that excavation is much easier in those than in the plentiful but otherwise less suitable (sometimes case-hardened) standing, dead trees. Because the most suitable nest trees for cavity excavation are snags that are themselves old-growth elements, one might even suggest that many of the fire-dependent, cavity-nesting birds depend not only on forests that burn, but on older forests that burn. Clearly, the relationship between pre-fire forest structure and post-fire bird communities deserves more attention.

A comparison of the bird communities in recent clearcuts and recent burns (Fig. 1) reveals a fair amount of similarity in the face of some important differences between the two cover types (Table 3), due primarily to the presence of standing dead trees in the burned sites, which are used for feeding and/or nesting purposes by a large number of bird species (see also Davis 1976). I found an even greater overall similarity between clearcuts and burns that are in mid-successional stages, suggesting that, when considered over all post-harvest successional stages, clearcutting may come closer to matching the natural patterns of bird occupancy on a patch of land than do many (or most) other cutting practices. I must reiterate, however, that the relative abundances of many species differ quite markedly between recently burned and recently cut forests. Even in mid-successional burns and clearcuts, which showed a greater relative similarity in bird-community composition than the earlier stages did, there were still significant differences in the absolute abundances of a large number of individual species (for example, compare the two abundance estimates for Calliope Hummingbird, Red-naped Sapsucker [*Sphyrapicus nuchalis*), Clark's Nutcracker, and Cedar Waxwing [*Bombycilla cedrorum*]). Therefore, even though the bird communities in clearcuts begin to look similar to those in fire-disturbed forests after a decade or two (Fig. 1), the bird communities are still quite different (in an absolute sense) from those that occur after a natural fire. Perhaps the best alternative to traditional harvesting methods in forests that evolved under standard-replacement fire regime may be to conduct some sort of partial harvest, after which the remaining forest would be burned lethally.

Fire (and its aftermath) should be seen for what it is: a natural process that creates and maintains much of the variety and biological diversity of the Northern Rockies. Most current cutting practices neither create large amounts of standing dead timber nor allow forests to cycle through stages of early succession that are physiognomically similar to those that follow stand-replacement fires. Unless managers begin to couple lethal burning with their cutting practices in those forests that evolved under stand-replacement fire regimes, traditional land-management practices will not achieve the goals of ecosystem management.

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PROCEEDINGS OF A SYMPOSIUM ON

ELK VULNERABILITY

Lonner & Cada - 82
137

Hillis - 38
Lyon - 99

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David W. McCleerey

DEFINING ELK SECURITY: THE HILLIS PARADIGM

ABSTRACT

Elk vulnerability may be reduced, and hunter opportunity may be increased, by providing security areas for elk during the hunting season. We define security area requirements for land managers so that timber harvest decisions can reflect elk security needs.

To provide a reasonable level of bull survival, each security area must be a nonlinear block of hiding cover ≥ 250 acres in size and ≥ one-half mile from any open road. Collectively, these blocks must equal at least 30% of the analysis unit. Vegetation density, topography, road access, hunter-use patterns and elk movements are variables that must be considered when applying these guidelines. Examples are provided that illustrate how the security guidelines are applied in the field.

INTRODUCTION

Timber harvest affects elk vulnerability by changing the structure, size, juxtaposition and accessibility of security areas. Biologists have recently provided working definitions of "security," "security area," and "elk vulnerability" (Lyon and Christensen 1990). However, elk and timber managers still await research answers to current questions such as: "How large must a cover block be to provide effective security, how far must a security area be from a road, and how much of the area should provide security to meet elk vulnerability objectives?"

We developed guidelines for retaining elk security areas west of the Continental Divide in Montana. We suggest that the concepts presented here could assist managers in providing security areas elsewhere. We also hope this stimulates constructive criticism and research that improve the guidelines.

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STUDY AREA

We devised guidelines applicable to the situations we know in the Clark Fork River drainage (excluding the Flathead River drainage). The area is characterized by steep slopes extensively forested by ponderosa pine, Douglas-fir, lodgepole pine, western larch and subalpine fir. Average elk populations and hunter numbers are at 30-year highs in the area, while the average bull/cow ratio observed by Montana Department of Fish, Wildlife and Parks (MDFWP) biologists in early spring has declined during the same period (MDFWP, Missoula, unpubl. data). The majority of elk habitat in the area is managed by the Lolo, Bitterroot and Deerlodge National Forests; although substantial portions are owned by Plum Creek Timber Company, Champion International Corporation, other private landowners, Montana Department of State Lands and USDI Bureau of Land Management.

MANAGEMENT OBJECTIVES

Lonner and Cada (1982) proposed that, "hunting recreational opportunities are good when hunting season lengths are relatively long, harvest rates are uniform, and rules and regulations few. The present 35-day general elk-hunting season in Montana permits a diversity of choice [for hunters] with regard to time, weather conditions, hunter density and area. A lengthy hunting season has little meaning if the majority of the harvest occurs in the first few days."

Nine years since Lonner and Cada's (1982) contribution, MDFWP and the three national forests within the study area formally share the following objectives: 1) maintain the current, relatively unregulated, 5-week hunting season; 2) distribute the bull harvest evenly over the entire hunting season; and 3) maintain a desired level of mature bulls in the post-hunting season population (For. Plan, Lolo Natl. For., 1986; For. Plan, Bitterroot Natl. For., 1987; For. Plan, Deerlodge Natl. For., 1987; Draft Elk Manage. Plan, MDFWP, Helena, 1991). We developed guidelines to meet these objectives.

The agencies have decided to maintain habitat security levels that allow desired numbers of bull elk to escape harvest, rather than impose more restrictions on hunters (e.g., shorter hunting seasons, antler-point restrictions, limited licenses). The recreational opportunities resulting from this type of management are becoming increasingly rare nationwide (Anon. 1988).

DOCUMENTATION

We developed guidelines from the following background of knowledge:

1. Elk behavior changes in response to the hunting season (Marcum 1975; Morgantini and Hudson 1979, 1985; Canfield 1988; Lyon and Canfield 1991).

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2. Elk avoid areas adjacent to roads with vehicular traffic, especially during the hunting season (Marcum 1975, Perry and Overly 1976, Lyon 1979, Irwin and Peek 1983, Lyon 1983, Lyon et al. 1985, Lyon and Canfield 1991).

3. Elk spend more time in dense cover during hunting season than they do before the hunting season (Marcum 1975, Irwin and Peek 1983, Canfield 1988). Large cover-blocks contribute to security more than small blocks (Canfield 1988, Lyon and Canfield 1991).

4. Elk movements generally are confined to habitats within a traditionally used home range (Edge et al. 1985, Lyon et al. 1985, Edge et al. 1986).

5. Road closures may either increase or decrease elk vulnerability depending upon the influences of cover, topography and hunting pressure, both within and adjacent to a security area (Basile and Lonner 1979, Lyon et al. 1985).

SECURITY-AREA GUIDELINES

How Large Must a Cover Block Be?

Larger is better—To meet the hunting opportunity objectives outlined here, managers should strive to retain, perpetuate, or replace the largest security areas possible. We assume that as security areas increase in size, elk become harder for hunters to find, and liberal hunting opportunities become less costly in terms of elk vulnerability.

Minimum size—In the lower Clark Fork drainage, conditions are favorable for elk to elude hunters: cover is dense, terrain is steep, and forest communities are largely unfragmented. Lyon and Canfield (1991) found that elk in this area selected for large, connected, vegetation communities (i.e., forest blocks of similar canopy structure). All other factors held constant, 236-acre unfragmented communities met minimum security requirements for 60% of the radioed elk. For the purposes of these guidelines, 250 acres appears to be the minimum-sized area for providing security under favorable conditions; under less favorable conditions, the minimum must be >250 acres.

Variables to consider—Effective security areas may consist of several different cover-types if the block is relatively unfragmented. For example, regenerated cutting units that provide reasonable cover might be found within an effective security area (Canfield et al. 1986). Among security areas of the same size, one with the least amount of edge and the greatest width generally will be the most effective. Rugged topography may increase security if it substantially decreases the accessibility of the area to hunters. Wallows, springs and saddles may require more cover than other habitats because both hunters and elk recognize and target these destinations.

How Far Must a Security Area Be from a Road?

Minimum distance—Generally, security areas become more effective the farther they are from an open road. Considering documented road-avoidance by elk (Lyon 1983, Lyon et al. 1985), the minimum distance between a security area and an open road should be one half mile. The function of this ≥ one half mile "buffer" is to reduce and disperse hunting pressure and harvest that is concentrated along open roads (Daneke 1980). Failure to accomplish this function will reduce the effective size of the security area and may render it ineffective.

Road design considerations—Road-design features may inadvertently turn designated security areas into hunter destinations. For example, trailheads, turnouts and/or parking areas in close proximity to security areas will concentrate hunting pressure in the vicinity and increase elk vulnerability. Similarly, open roads located both above and below a security area on a slope will encourage hunters to walk through the security area.

Cover and terrain—When cover is poor and terrain is gentle, it may require a distance >one half mile from open roads before security is effective. In such situations, hunters may identify the security area from the road, and the gentle terrain will deter few hunters from hiking. Conversely, if the security area is hidden or difficult to reach from a road, elk may find security in situations < one half mile from an open road.

Closed roads—Roads may be closed (to motorized travel) to provide security and a buffer between security areas and open roads. However, the minimum distance between open roads and security areas increases as closed-road densities increase within both the security area and buffer. Closed roads located within security areas may increase elk vulnerability by providing hunters with walking and shooting lanes. Use of horses and increasing use of mountain bikes by hunters on closed roads allows them better access and increases elk vulnerability, compared to unroaded habitats. Therefore, roads within security areas should be kept to an absolute minimum.

How Much of the Area Should Provide Security?

Analysis unit—First, a standardized "habitat analysis unit" (Lyon and Christensen 1990) must be described. To be biologically meaningful, analysis unit boundaries should be defined by the elk herd home-range (Edge et al. 1986), and more specifically by the local herd home-range during hunting season. Typically, the hunting season home-range includes the local herd transitional-range and at least the upper edge of winter range. These boundaries should be verified in advance by radio telemetry, particularly where elk vulnerability is at issue. Without telemetry data, biologists should test their home-range predictions against the experience of reliable local hunters and outfitters. Analysis units should not be adjusted for land ownership; instead, they should reflect the cumulative habitat conditions perceived by elk.

Minimum amount of security—Our collective experience suggests elk vulnerability increases when less than 30% of an analysis unit is comprised of security areas (Canfield 1991). Where bull survival objectives are high, it may be necessary to retain greater than 30% of the analysis unit in security.

Spatial arrangement—In conjunction with considering "how much security," it is critical to consider spatial arrangement of security areas across the landscape. The arrangement should provide for the habitat needs of elk through the 5-week hunting season (e.g., forage and water). Providing security only on dry, harsh, steep slopes may allow elk to avoid hunters early in hunting season; however, it is unlikely that elk will stay in harsh sites for extended periods (Marcum 1975). Further, security areas should cover a wide elevational range so they are available to elk under various weather conditions (e.g., security areas at high elevations may be unusable by elk during periods of deep snow).

A few large, or several minimum-sized, security areas may comprise the same combined proportion of an analysis unit. The best balance between security-area sizes and numbers for an analysis unit will result from creative thinking firmly based on knowledge of local elk-movement and hunting patterns.

APPLICATION OF THE MANAGEMENT GUIDELINES

We suggest that security areas should be ≥ 250 acres in size, ≥ one half mile from an open road, and should comprise ≥ 30% of a valid analysis unit. Unquestioning adherence to these guidelines may lead to serious misapplications and should be avoided. We believe the guidelines are properly applied when used to compare relative security levels in an analysis unit over time or to compare and evaluate the cumulative impacts of various timber-harvest alternatives on security. These guidelines represent minimums and do not necessarily justify reducing security to meet these levels (i.e., if 50% of an analysis unit is security, do not assume that 20% of the unit is excess security).

Inferences from detailed knowledge of a local elk herd—such as that typically obtained by radio telemetry—should override these management guidelines whenever discrepancies occur. For example, radioed elk have shown us site-specific exceptions where security is provided along highways or in small cover-blocks that hunters do not find. Similarly, traditional migration corridors and other elk concentration areas, if known, may deserve special considerations that are not covered by these guidelines (USDA 1991).

A comprehensive, sustained timber-management planning effort is required to obtain the greatest benefits from these guidelines. Radio-telemetry data should be collected ≥ 1 before year preparing alternative management strategies, and it may take ≥ 1 year to budget and prepare for a projected telemetry effort. Future timber harvest rotations, and recruitment of new security areas, should be projected to evaluate the best options for any proposed timber sale. Proposed timber harvests in remote and heavily forested analysis units should be carefully approached because the rare opportunity exists to retain elk security by design in these units, rather than by default as dictated by past logging practices.

In analyzing security requirements for a specific area, interpretation of the guidelines is needed to ensure that the result makes biological sense for local conditions. The point of designating elk security areas is not to meet some generalized guidelines, but to provide functional habitat.

We present examples of actual management problems we have addressed, to illustrate: 1) guideline adjustments that made designated security areas reflect reality, and (2) provisions for meeting present and future security needs.

Example 1

The Sequoia-Brewster area lies about 20 miles from Missoula. The area's entrance road ends at a gate on level terrain (Fig. 1). The ease of walking in the area and the concentration of hunters at the end of the road suggested to us that an area only one half mile from the parking area would not provide adequate security. Therefore, the buffer between the parking and security areas was increased to 1 mile.

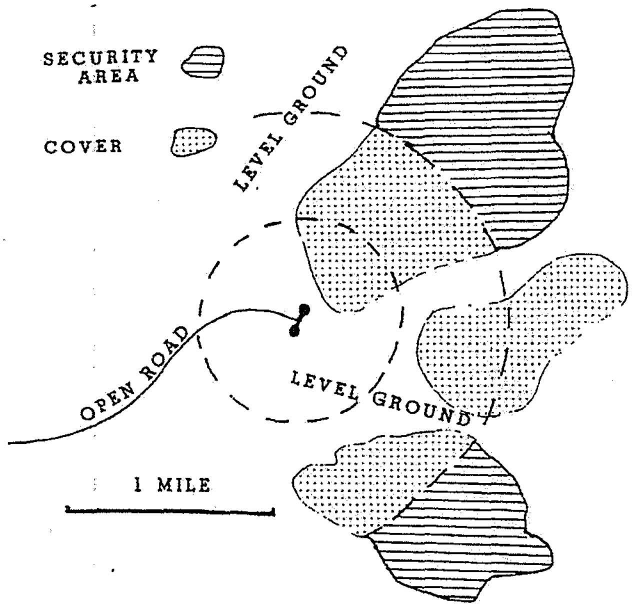

Figure 1. *Sequoia-Brewster area, showing the relationship of security to an open road, recognizing the trailhead, level ground, and concentrated use. The one half mile zone was increased to 1 mile.*

wrong place. This made bulls especially vulnerable.

Predictably, the bull/cow ratio of this elk population is extremely low. After the 1989 hunting season, this ratio had declined to 3:100 (J.E. Firebaugh, MDFWP, Missoula, pers. commun.), suggesting that bull mortality due to hunting is very high.

To recover security in this analysis area, we first proposed to decrease hunter access to the high-elevation basins by closing entry roads near the points where they cross the divide from the west (Fig. 3). Second, to allow recovery of large cover-blocks in the productive, high-elevation basins, we developed a long-term strategy for the spatial arrangement of timber harvest: deferring timber harvest in designated large blocks (Fig. 3) to allow contiguous areas to regain cover at the same time, and reduce the area's fragmentation. Third, future timber harvests will be designed to minimize fragmentation by concentrating logging in small areas not currently providing security. The initial logging entries will revisit previously logged land, joining (in effect) the scattered, recovering units (Fig. 4). This will create a block of recovered cutting-units that will provide the next generation of security, totalling about 25% of the analysis unit by the year 2000.

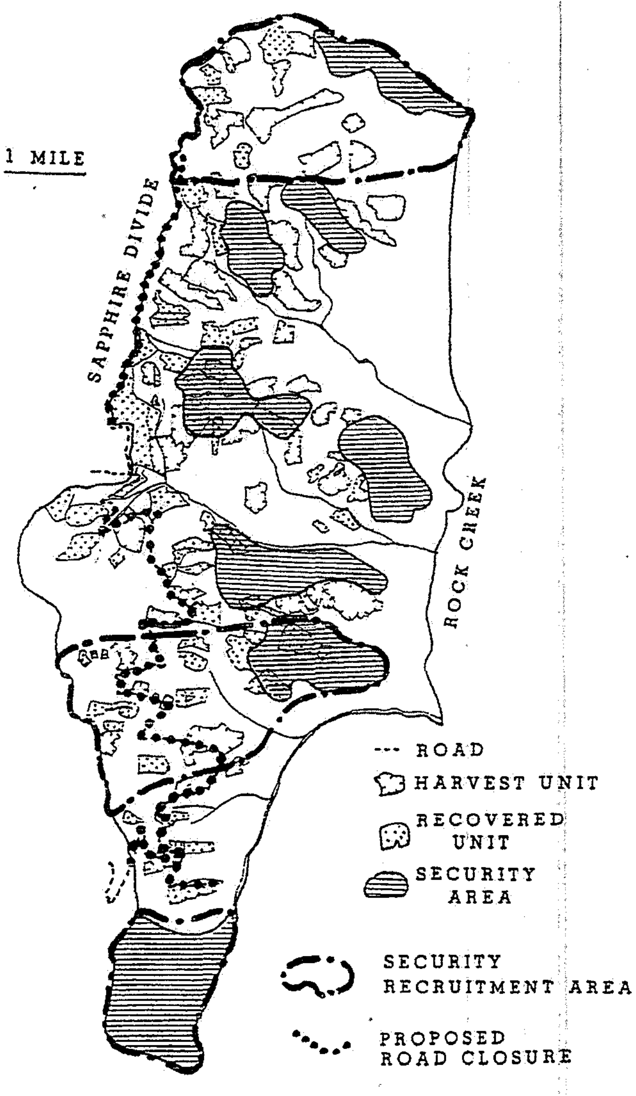

Figure 3. *Sapphire Divide area, showing designated security recruitment areas and roads proposed for closure. Note how the past harvest pattern has fragmented cover.*

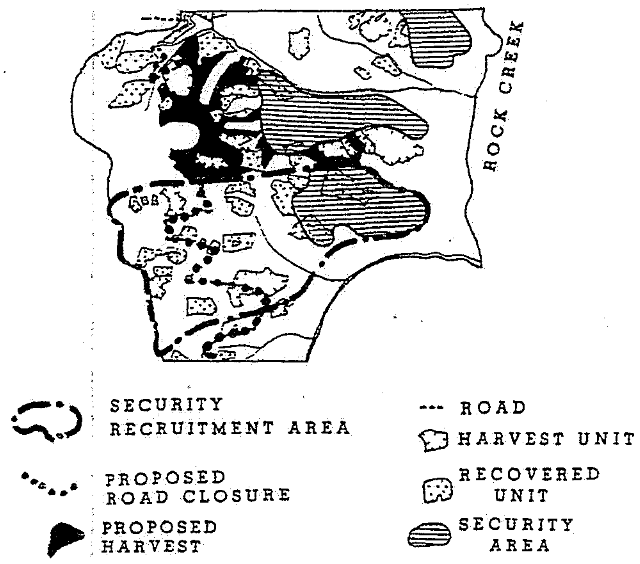

Figure 4. *South Sapphire Divide area, showing a clustered timber harvest strategy designed to create a large block of future security. Note how proposed cutting units are adjacent to recovered harvest units.*

CONCLUSIONS

During the last year, these guidelines were applied to nine elk herd-units involving 14 timber sales. Two disturbing trends were discovered. First, most herd units already had less than the minimum 30% security due to past timber harvest; in many of these cases, there were strong indications that bull survival was declining or at risk. Second, even in situations where security was substantially less than 30%, all remaining security stands were targeted for timber harvest. This indicates that timber harvest decisions made over the next few years will potentially severely impact remaining security and, ultimately, hunter opportunity.

Additional research is needed to test and refine these guidelines. However, based on the rapid, apparent decline of security, it is critical that we begin applying these guidelines immediately. Planning must not only address the quality and spatial arrangement of existing security areas, but also must provide for the regeneration of replacement security areas where a sustained timber harvest is desired.

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Liz Cheney: The House Jan. 6 committee has uncovered a 'broad,' 'well-organized' conspiracy
by Aldous J Pennyfarthing
★262 188

One More Dead Russian General in Ukraine
by GaryNaham
★156 92

Tweets of the Week May 29-Jun

Bald eagle chick, dead in nest tree after Cal Fire control burn next to the nest in 2021. Cal Fire has not committed to stop burning by the nest this year. Who would think it would be a problem for public agencies to adhere to laws that protect wildlife?

Climate change impacts have been worsening for years, raising temperatures and exacerbating fire danger in California and the world. In many cases though, trees and other plants are being treated as enemies to be annihilated, rather than as the ecosystems that enable life on earth to exist.

In California, both Cal Fire and PG&E are being given exemptions from any environmental review for their "fuel reduction" or "vegetation management" programs. In this time when thousands of scientists worldwide are screaming and waving red flags about biodiversity and climate catastrophe, the impacts of these projects are being ignored, particularly to wildlife and habitat.

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4 2022
by oldhippiedude
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"School lunch isn't cost effective for taxpayers."
by SquireForYou
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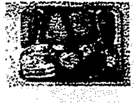
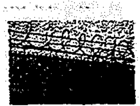


And this is freedom? Are they completely insane? Time to lock THEM up.
by Vetwife
★155 💬114



A Dad's Abortion Story...
by ExPatDanBKK
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The Voting System WAS Tampered With In Georgia And Raffensperger Covered It Up
by deltopia
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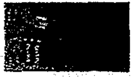

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Political Christianity has killed biblical Jesus and replaced him with the Fascochrist.
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Let's start assigning blame for mass shootings where it really belongs: "A well regulated Militia.."
by integrate
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I think Russia is being enveloped right now in that triangle
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★229 💬81

PRESIDENT JOE BIDEN—DAY501--SECOND YEAR DAY138—Evening Shade-Sunday
by hpg
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Is the fertilized egg, blastocyst, zygote, embryo a person, a separate self? I argue it cannot be.
by novapsyche
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CRYBABY TRUMP Bashes FOX News, 'Perverts' and 'RINOS' Who 'Didn't Have the Guts' to Impeach Him
by News Corpse
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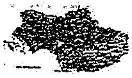

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Cal-Fire-burns-n

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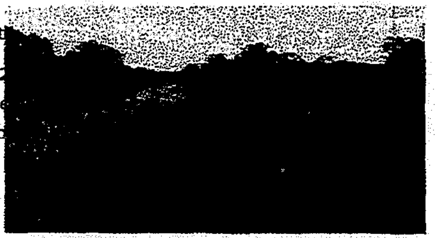
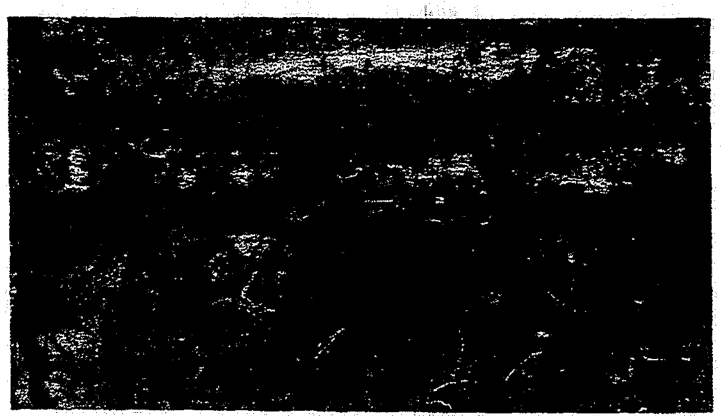
What has happened over the past several years to a bald eagle nest east of Red Bluff, California illustrates what is occurring because of these exemptions from any oversight. The fear of fire is being exploited to the detriment of the natural world, rather than substantive actions being implemented to reduce the emissions which are causing climate disaster.



Hwy 36, east of Red Bluff. The eagle nest is to the right (south side). This is the roadside Cal Fire has burned in 2020 and 2021 when the eagle nest was occupied. There is little reason to burn here, and many reasons not to.

The eagles' nest to the south of the highway, circled in red.



Parent eagle with young eaglet in front (little grey head) in nest tree, April 2022.

Local residents have been watching this nest since 2020. A photographer from Red Bluff was going out to the nest every day in 2021. At the end of May, the photographer saw a notice that there was going to be a control burn by the nest in a few days. She contacted a local eagle group, who

"We are breaking them like nuts."
by annieli
★145 59

A Ukrainian military expert confirms much of the analysis you've read here and looks to the future
by Charles Jay
★294 108

Packard Museum: The 1920s (photo diary)
by Ojibwa
★37 11

called Cal Fire to tell them about the nest which was occupied by two young eaglets. The eagle group left a message and received a message back from Cal Fire saying their biologist said it was fine to be burning near the nest. The eagle group called back to get the biologist's name, but received no answer then or later. It wasn't "fine".

The burn was done on June 1st. This nest is approximately 100 feet down a ravine from the highway. The eaglets were probably only 6-7 weeks old, 4 or 5 weeks from being able to fly.

The photographer was standing next to the nest during the burn and taking photographs. The Cal Fire people were slightly to the east of the nest. The smoke and flames can be seen on the south side of the highway, on the same side as the nest.

Cal Fire burning next to eagle nest, 2021. How much extra CO2 is being emitted by extra equipment use and burning unnecessarily?

The photographer went to check the nest a few days later and saw one adult perched above the nest, but could see no eaglets.

The next morning, the photographer took a photo which shows a dead eaglet hanging from the nest. The photographer contacted me (Marily Woodhouse from Defiance Canyon Raptor Rescue). We went to search for the other eaglet, in the hope it was still alive.

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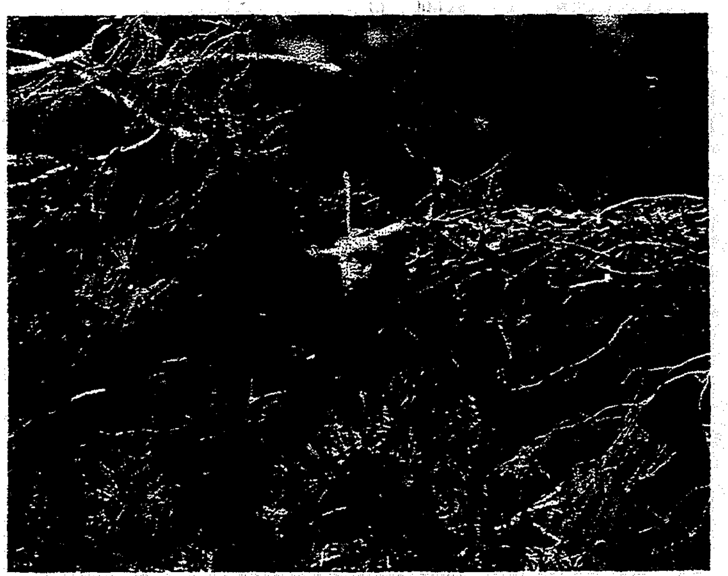
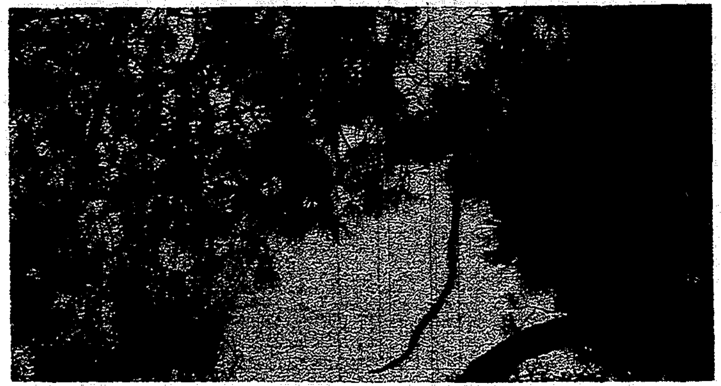


Dead eaglet hanging from the nest.

Both the adults were at the top of the tree, above the nest. I went down the ravine to the nest tree to walk around beneath it to search for the second eaglet and the adults stayed in the tree top. I walked to the southeast side of the tree and looked up and saw the other eaglet hanging dead in the tree, below the nest about 10'.



The second eaglet, dead below the nest.

We reported the deaths to US Fish and Wildlife and CA Department of Fish and Wildlife, but never received any notification of any action taken.

A State Wildlife Health Lab biologist wrote to us later that:

"A bird's respiratory system is more sensitive to toxins, including smoke, than a mammal's respiratory system. This is because birds have a higher oxygen demand than mammals and a bird's lungs are 10 times more efficient at capturing oxygen. The rapid efficiency of gas exchange in bird lungs makes them more susceptible to inhaled toxic agents, including smoke. Inhaled toxins, such as smoke, can cause irritation and damage the respiratory system. It also can

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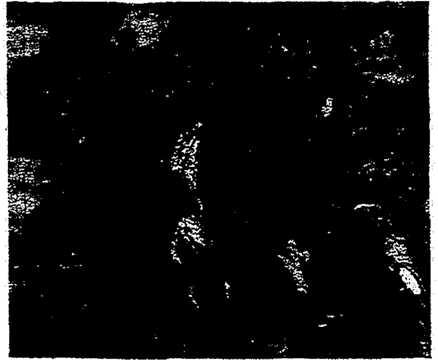
compromise the immune system, making the bird more susceptible to infections. This is especially true in young birds in the nest that are unable to escape the smoke. Smoke inhalation toxicity in birds is caused by irritant gases (aldehydes, hydrogen chloride, and sulfur dioxide), particulate matter, and nonirritant gases (carbon monoxide, carbon dioxide, and hydrogen cyanide) released by combustion."

There was a burn done next to the Dales Station nest in 2020 also. I was called upon to rescue an eaglet who got out of the nest before he could fly that year. It was several days before the burn was done that year, so he was away from the nest when the burn occurred. His sister was still in the nest during the burn. I received a call from Dales Station, less than a mile from the nest, in August, 2020 about an eaglet who had been on the ground for 3 days, standing next to a shallow pool of Paynes Creek. My determination was that it was the female from the nest. She was open-mouthed breathing with a raspy noise. She died a few hours after she was caught and transported. The Wildlife Lab report said: "This was a juvenile female in poor nutritional condition with no fat reserves and minimal pectoral muscle development. Internally, there was evidence of an extensive infection. The visible infection resembled avian tuberculosis which is caused by the bacterium Mycobacterium avium. It's widespread in the environment in soil and dust and is usually an opportunistic infection. Depending on where the lesions are in the bird, gives an idea of how it entered the body. The lesions in this bird were primarily in the air sacs suggesting it was inhaled."

The male who had been in care was released in 2020. A first year eagle was seen back at the nest in 2021. Judging by his and the adults' behavior, it was the male who was in care away from the nest during the burn in 2020.



2020 bald eaglet being released. He was away from the nest in rehab care during Cal Fire's control burn in 2020, which probably saved his life.

I had occasion to contact Cal Fire in February 2022 about another issue. I had just been informed that the Dales Station bald eagle nest was occupied, so mentioned it in the hope of preventing another burn next to the nest. Cal Fire and its employees are public servants. It is their job to uphold state and federal laws, which include protection of wildlife, but the answer from a Cal Fire employee contained only dismissive, condescending remarks, clearly refusing to take steps to ensure any protections were implemented.

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Daily Kos

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Let's start assigning blame for mass shootings where it real...

Daily Kos

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Many letters, calls, and emails have ensued since February (most unanswered). I made maps from Cal Fire's own fire database showing how rarely the area around the nest has burned. The ravine area there is extremely rocky and is grazed by cattle. And then there are the State and Federal laws that protect nesting birds. Still, Cal Fire will not commit to refrain from burning by the nest again this year.

Last week a biologist from a PG&E contractor company working in Greenville (a town that burned in the Dixie fire last year) called Raptor Rescue because they wanted us to take eggs from a nest in a tree they wanted to cut down. I explained the multitude of reasons that was a bad idea, along with it being illegal for them to do. The man said "We have an exemption". How many nesting birds are being destroyed in California due to these stupid, thoughtless exemptions and the complete lack of oversight which is occurring?

There have got to be protections enforced. Apparently that won't happen without widespread public outrage.

Here are some state employees to contact if you will help tell them there is a problem with their practices:

George Morris, Cal Fire Northern Region Unit Chief (530) 224-2445 (They would not give out his email address)

Dave Russell, Cal Fire Tehama/Glenn Unit Chief (530) 528-5199 dave.russell@fire.ca.gov

Tina Bartlett, Regional Manager CDFW, (530) 225-2300 tina.bartlett@wildlife.ca.gov

People often focus on individual species, but we believe every species is important, whether it is on a man-made list or not. Habitat fragmentation and loss have significant impacts on wildlife. Defiance Canyon Raptor Rescue works to rescue, rehabilitate, and return raptors to their wild lives, along with our work to protect watersheds and forests of California.

www.thebattlecreekalliance.org (http://www.thebattlecreekalliance.org/)

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Petition to List the Pinyon Jay (*Gymnorhinus cyanocephalus*) as Endangered or Threatened Under the Endangered Species Act

Photo: Mike Lewinski, Taos, NM

Submitted to the U.S. Secretary of the Interior
acting through the U.S. Fish and Wildlife Service

April 25, 2022

Defenders of Wildlife

NOTICE OF PETITION

April 25, 2022

Deb Haaland
Secretary of the Interior
U.S. Department of the Interior
1849 C Street NW
Washington, DC 20240

Martha Williams
Director
U.S. Fish and Wildlife Service
martha_williams@fws.gov
fws_director@fws.gov
via email

Dear Secretary Haaland:

Pursuant to the Endangered Species Act ("ESA"), 16 U.S.C. § 1533(b), the Administrative Procedure Act, 5 U.S.C. § 553(e), and the ESA's implementing regulations, 50 C.F.R. § 424.14, Defenders of Wildlife formally petitions the Secretary of the Interior to list the Pinyon Jay as an endangered or threatened species and to designate critical habitat concurrent with the listing. 50 C.F.R. § 424.12.

This Petition sets in motion a specific process, placing definite response requirements on the Secretary of the Interior and the U.S. Fish and Wildlife Service ("FWS"), by delegation. Specifically, FWS must issue an initial finding as to whether the Petition "presents substantial scientific or commercial information indicating that the petitioned action may be warranted." 16 U.S.C. §1533(b)(3)(A). FWS must make this initial finding "[t]o the maximum extent practicable, within 90 days after receiving the petition." *Id.* Petitioners need not demonstrate that listing or reclassification is warranted; rather, petitioners must only present information demonstrating that the petitioned action may be warranted. While petitioners believe that the best available scientific and commercial data demonstrates that listing of the Pinyon Jay as endangered is in fact warranted, there can be no reasonable dispute that the available information indicates that listing this species as either endangered or threatened throughout all or a significant portion of its range may be warranted. FWS must promptly make an initial finding on the Petition and commence a status review as required by 16 U.S.C. § 1533(b)(3)(B).

As required by 50 C.F.R. § 424.14(b), Defenders provided written notice (via email) to the state agencies responsible for the management and conservation of the Pinyon Jay on March 16, 2022, more than 30 days prior to the submission of this Petition. A copy of the notice accompanies this Petition. *See* 50 C.F.R. § 424.14(c)(9). We anticipate that, in keeping with 50 C.F.R. § 424.14(f)(2), FWS will acknowledge the receipt of this Petition within a reasonable timeframe. As fully set forth below, this Petition contains all the information requested in 50 C.F.R. § 424.14(c)–(e) and 16 U.S.C. § 1533(e). All cited documents are listed in the Literature Cited section; electronic copies of these documents accompany this Petition; and pinpoint citations to these have been provided where appropriate. *See* 50 C.F.R. § 424.14(c)(5)–(6).

Petitioner Defenders of Wildlife ("Defenders") is a non-profit conservation organization dedicated to the protection of all native animals and plants in their natural communities. Defenders' 2019–2028 Strategic Plan identifies keystone species as one of several key groups of species whose conservation is a priority for our organization's work,[1] and has been working to protect the Pinyon Jay for years. Defenders uses science, education, litigation, and research to protect wild animals and plants. Known for our effective leadership on endangered species issues, Defenders also advocates for new approaches to wildlife conservation to protect species before they become endangered. Our programs reflect the conviction that saving the biodiversity of our planet requires protecting entire ecosystems and ensuring interconnected habitats. Founded in 1947, Defenders of Wildlife is a 501(c)(3) membership organization with nearly 2.2 million members and supporters.

If you have any questions, please feel free to contact us via the information contained in the signature blocks below.

Sincerely,

Patricia Estrella
New Mexico Representative
pestrella@defenders.org
(505) 395-7334

Bryan Bird
Southwest Program Director
bbird@defenders.org
(505) 395-7332

Petitioner
Defenders of Wildlife
1130 17th Street NW
Washington, DC 20036

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[1] More information on Defenders' work is available at https://www.defenders.org and Defenders' 2019–2028 Strategic Plan is available at https://defenders.org/sites/default/files/2019-06/Defenders-of-Wildlife-2019-2028-Strategic-Plan.pdf.

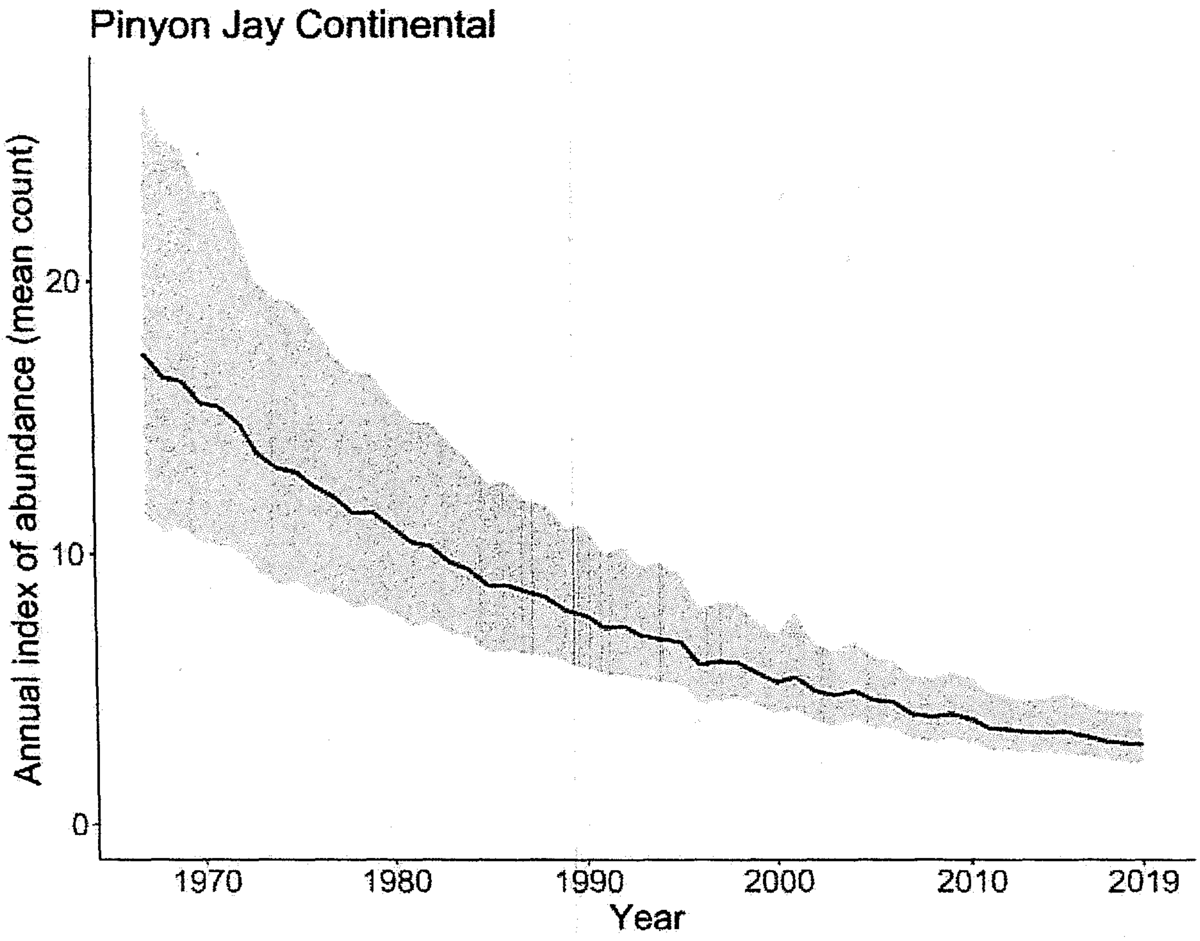

Figure 4. Survey-wide population trajectories for the Pinyon Jay estimated from the BBS using the standard regression-based model (SLOPE) used for BBS status and trend assessments since 2011.

species of fungi, 165 known species of lichen, and 25 species of bryophytes. Soil microorganisms and over 10,000 insect species, 64 species of mammals, and at least 113 species of birds have been described in Mesa Verde's pinon-juniper woodlands (Floyd 2021, at 7–8). In addition to supporting high biodiversity, piñon-juniper woodlands make significant contributions to carbon sequestration (Floyd 2021, at 8).

At least 73 bird species breed in piñon-juniper woodlands, and over half are Neotropical migrants (Balda and Masters 1980, at 150–51). In one study in Utah, piñon-juniper bird communities ranked second in the percentage of obligate and semi-obligate species, third in total number of individuals counted, and fourth in species richness and diversity (Paulin et al. 1999, at 242). Total bird numbers and species were higher in every season in Rocky Mountain juniper stands than in grasslands (Sieg 1991, at 2–3). Piñon-juniper habitats also support high mammal, herpetofauna, and invertebrate diversity (Bombaci and Pejchar 2016, at 36).

In addition to Pinyon Jays, several other bird species of conservation concern breed in piñon-juniper habitats, including declining high priority obligates such as the Juniper Titmouse (*Baeolophus ridgwayi*) and Gray Vireo (*Vireo vicinior*). USFWS and PIF list several piñon-juniper species of conservation concern, and PIF conservation plans in several western states list priority species which breed in piñon-juniper. Because of the role of the Pinyon Jay as a long-distance seed disperser for piñon pines, the jay is crucial for the establishment and maintenance of piñon-juniper woodlands, and it is therefore key to the conservation of other birds and wildlife of these habitats.

Species	**USFWS BCC**	**PIF Red**	**PIF "R" Yellow**	**PIF "D" Yellow**
Gray Vireo			x	
Pinyon Jay	x			x
Woodhouse's Scrub-Jay	x			
Juniper Titmouse				
Mountain Chickadee				
Bushtit				
Bendire's Thrasher	x	x		
Virginia's Warbler	x			x
Black-throated Gray Warbler	x			
Black-chinned Sparrow	x			x

Table 5. USFWS Birds of Conservation Concern (USFWS 2021) and PIF priority species (Partners in Flight 2021a) breeding primarily in piñon-juniper habitats.

a long-term, rangewide monitoring program, and therefore provides the most reliable and only long-term population trends available for Pinyon Jay.

Pinyon Jay populations have declined rangewide, in every Bird Conservation Region ("BCR"), and in every state (Table 3). These declines are clear even when considering only the highest-credibility data (blue highlighted, Table 3). The two BCRs with the highest credibility ranks, Southern Rockies/Colorado Plateau and Great Basin, also harbor 90% of the Pinyon Jay population and have declined at approximately 2% per year since 1967. States having the highest proportion of the Pinyon Jay global population show similarly high yearly declines over the long term (Table 3). However, within the Pinyon Jay range, BBS data suggest that some areas have more severe declines, while some areas may show population increases (Figure 5. **Geographic variation in Pinyon Jay population trends. Data from BBS; trend map ending in 2019 not available. Numbers refer to Bird Conservation Regions listed in Figure 1. CC BY Defenders of Wildlife 2021. See Appendix 1 for metadata.**).

Note that the annual population trends presented here are the most recent compiled by BBS. Some readers may note that earlier trend estimates for 1967-2015 indicated larger annual decline rates for Pinyon Jay than the ~2% rangewide estimate in Table 3. This discrepancy occurred when BBS changed the analysis methods it uses to calculate trends for some species, including Pinyon Jay, starting in 2019 (J. Sauer pers. comm. to C. Beidleman, 16 August 2021). Partners in Flight used the older trend numbers for the population decline and half-life estimates provided below. Hence, these estimates would be different if the latest trends from BBS were incorporated. Using the newer analytical methods, J. Sauer (pers. comm. to C. Beidleman) estimates that the Pinyon Jay population declined by 66.8% from 1967–2019, rather than 85%, as projected by Partners in Flight, below (Rosenberg et al. 2016, at 52).

Partners in Flight

Partners in Flight (PIF) finds the Pinyon Jay long-term (1970-2014) population has declined by 85%, and the short-term (2004-2014) population change has declined by 3.7% (Rosenberg et al. 2016, at 52). The population half-life is estimated at 19 years, meaning that an additional 50% loss of the global population is expected by 2035. PIF therefore considers the Pinyon Jay as a species with a short "half-life" and high urgency (Rosenberg et al. 2016, at 3, 34, 52).

The Partners in Flight Avian Conservation Assessment Database (Partners in Flight 2021a) provides ranks based on several component scores, which are added to produce a risk ranking. A total score for each landbird species then places each at-risk species in one of three categories: Red Watch List, Yellow Watch List, or Common Birds in Steep Decline. Species are included in the Watch List if they have a maximum combined score of ≥14, or 13 in combination with a population trend score of 5. Red Watch List species have a combined score >16 and are considered highly vulnerable and urgently in need of special attention. Yellow Watch List species are considered to have restricted ranges and small populations and are in need of constant care. These species are further divided into "R" Yellow Watch and "D" Yellow Watch species. "R" Yellow Watch species have high vulnerability scores for restricted ranges and small populations, with moderate threats and stable or increasing trends. "D" Yellow Watch species have declining populations, with high trend scores, moderate to high threats, and low vulnerability scores for range. Common Birds in Steep Decline are still numerous or widely distributed enough that they do not warrant Watch List status but are experiencing long-term declines. They have lost from 50%-90% of their populations since 1970 and most are projected to lose another 50% within 20-25 years. For detail on how these scores are calculated, see Panjabi et al. (2021, at 7–21).

USFWS BCC

Species	**Priority Species, State PIF Plan**						
Ferruginous Hawk			ID		NV	UT	
Black-chinned Hummingbird		CO					
Gray Flycatcher	AZ	CO	ID		NV	UT	WY
Ash-throated Flycatcher							WY
Cassin's Kingbird		CO					WY
Gray Vireo	AZ	CO		NM	NV	UT	WY
Plumbeous Vireo			ID				
Pinyon Jay	AZ	CO	ID	MT	NE	NM	NV
Woodhouse's Scrub-Jay				NM			WY
Juniper Titmouse	AZ	CO		NM	NV		WY
Mountain Chickadee				NM			
Bushtit				NM			WY
Western Bluebird					NV		WY
Bendire's Thrasher			ID	NM	NV	UT	
Virginia's Warbler							
Black-throated Gray Warbler	AZ	CO	ID	NM	NV	UT	
Black-chinned Sparrow				NM			
Scott's Oriole		CO			NV	UT	

Table 6. Piñon-juniper priority bird species, from PIF state conservation plans.

III. IDENTIFIED THREATS TO THE PETITIONED SPECIES: FACTORS FOR LISTING

As demonstrated below, substantial scientific and commercial information indicates that listing the Pinyon Jay as endangered or threatened in all or in any significant portion of its range may be warranted. *See* 16 U.S.C. § 1533(b)(1)(3)(A). The species is declining throughout its range and faces threats including habitat loss and degradation, climate change, and more. Existing regulatory mechanisms have proven inadequate to protect the Pinyon Jay. Without adequate protections, the species' limiting life history characteristics, in combination with the other threats discussed, cause the Pinyon Jay to be in danger of extinction throughout all or a significant portion of its range or likely to become so within the foreseeable future.

A. Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range

1. Historical Woodland Dynamics and Disturbance Regimes

To assess, understand, and manage the condition of forests and woodlands, scientists and managers wish to know their pre-historical/historical range of variation ("HRV"), which is influenced by

Farmington Daily Times

NEWS

'Hundreds of thousands, if not millions': New Mexico sees massive migratory bird deaths

Algernon D'Ammassa Las Cruces Sun-News

Published 2:41 p.m. MT Sep. 12, 2020 | **Updated 2:56 p.m. MT Sep. 12, 2020**

LAS CRUCES - Biologists from New Mexico State University and White Sands Missile Range examined nearly 300 dead migratory birds Saturday at Knox Hall on the university's main campus.

Over the past few weeks, various species of migratory birds are dying in "unprecedented" numbers of unknown causes, reported Martha Desmond, a professor at NMSU's Department of Fish, Wildlife and Conservation Ecology.

"It is terribly frightening," Desmond said. "We've never seen anything like this. ... We're losing probably hundreds of thousands, if not millions, of migratory birds."

In August, large numbers of birds were found dead at White Sands Missile Range and at the White Sands National Monument in what was thought to be an isolated incident, Desmond said.

After that, however, came reports of birds behaving strangely and dying in numerous locations in Doña Ana County, Jemez Pueblo, Roswell, Socorro and other locations statewide.

The affected birds have included warblers, sparrows, swallows, blackbirds, flycatchers, and the western wood pewee.

"A number of these species are already in trouble," Desmond said. "They are already experiencing huge population declines and then to have a traumatic event like this is – it's devastating."

On Saturday, Desmond was joined by Trish Cutler, a wildlife biologist at WSMR, and two NMSU students for an initial evaluation of the carcasses.

Desmond said her team also began catching and evaluating living specimens on Friday as residents find birds behaving strangely and gathering in large groups before dying.

"People have been reporting that the birds look sleepy ... they're just really lethargic," Cutler said. "One thing we're not seeing is our resident birds mixed in with these dead birds. We have resident birds that live here, some of them migrate and some of them don't, but we're not getting birds like roadrunners or quail or doves."

On the other hand, numerous migratory species are dying rapidly and it is not immediately clear why, although the cause appears to be recent. Desmond said the birds had moulted, replacing their feathers in preparation for their flight south, "and you have to be healthy to do that; but somewhere after that, as they initiated their migratory route, they got in trouble."

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The biologists guessed the cause might involve the wildfires ravaging the western U.S. and dry conditions in New Mexico.

"They may have been pushed out before they were ready to migrate," Desmond said. "They have to put on a certain amount of fat for them to be able to survive the migration. These birds migrate at night and they get up in the jet stream, and they might migrate for three nights in succession, they'll come down and they'll feed like crazy, put on more fat and go again."

The biologists noted that the majority of the dying birds are insectivores, but that seed eaters were sickening and dying as well.

The birds will be sent to the U.S. Fish and Wildlife Service Forensics Laboratory in Ashland, Ore. for further analysis. Desmond it could be weeks before results come back, and the findings could bear serious ecological implications.

"Over 3 billion birds have died since 1970. Insect populations are crashing, and this is just an unprecedented mortality," she said. "Climate change is affecting the abundance of insects, it's affecting the volatility of the fires, and the scary thing is this may be an indication of the future."

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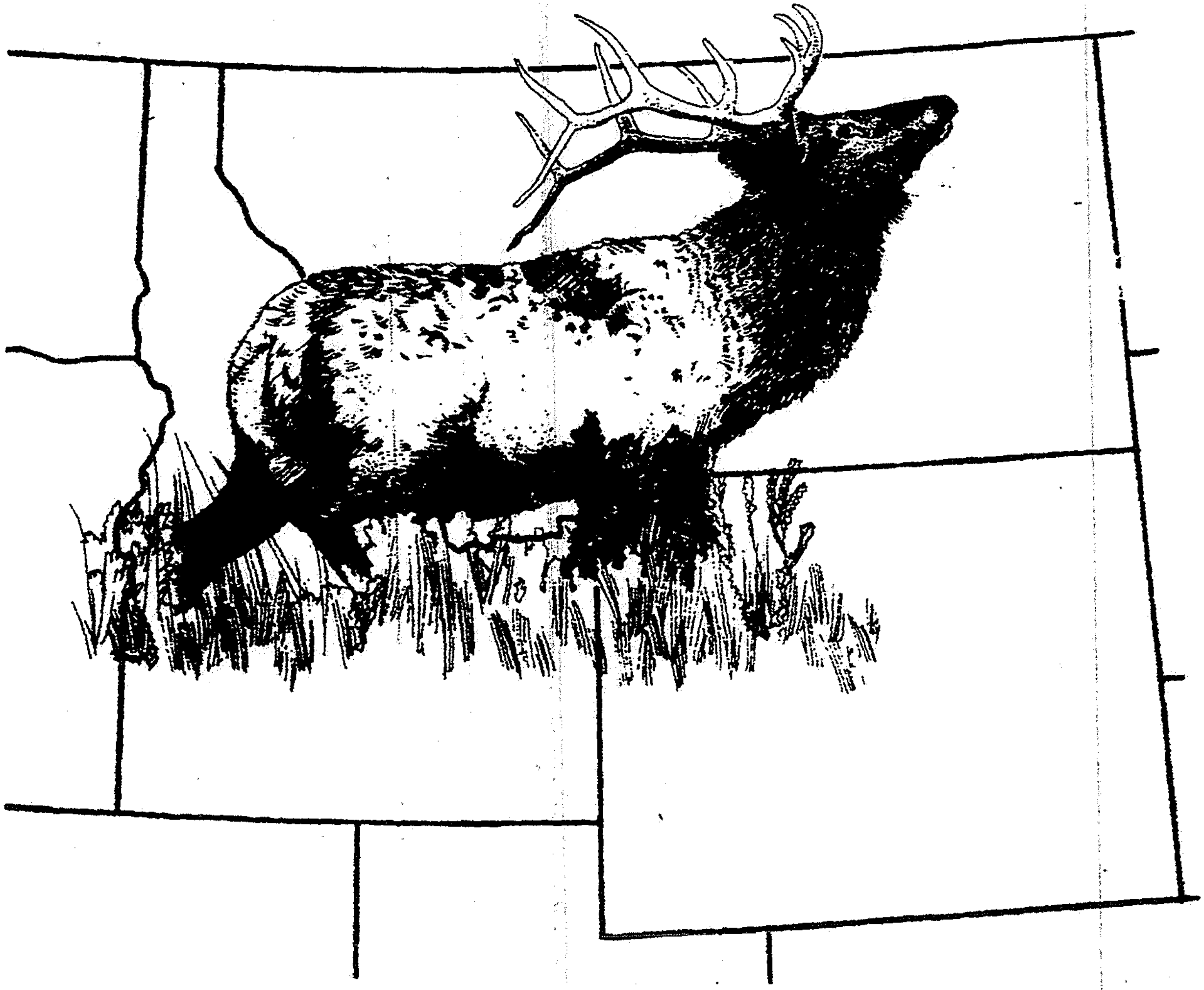

Elk Management in the Northern Region: Considerations in Forest Plan Updates or Revisions

Alan G. Christensen
L. Jack Lyon
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INTRODUCTION

In the West, a large percentage of elk habitat is managed by the Forest Service, U.S. Department of Agriculture. Elk are a giant economic factor in Montana and Idaho, easily accounting for over $100 million annually for hunting alone. This activity is especially important to many small, rural communities. At the same time, elk management can be controversial where it conflicts with other resource activities such as grazing, logging, and public access. As the habitat manager for this valuable resource, the Forest Service must develop management programs based on the best available information, work closely with State game managers, fully inform the public and disclose the effects of management actions, and embrace implementation of an ecological approach to elk management.

When the initial forest plans were developed in the early 1980's, there was no cohesive direction identifying a common set of elk management standards. As plans were written, generally recognized key pieces of elk management information were creatively modified by virtually every forest in an attempt to meet local needs. This often resulted in adjacent forests having startlingly different goals, objectives, standards, guidelines, and terminology.

In these forest plans, the approach to elk was usually narrow and focused. Because elk was a regional indicator species, managers established population targets, habitat standards, and monitoring goals. We recognize now that elk are part of a bigger picture and that elk habitat management must be placed within the context of ecosystem management, biodiversity, State management strategies and goals, and shifting public demand and interest that now embrace nonconsumptive and consumptive interests.

This problem has been recognized and a solution proposed for the Northern Region of the Forest Service. Common terminology, a new perspective on elk vulnerability, and a better understanding of the application of habitat effectiveness have created the opportunity for forests to be more consistent and in tune with State management objectives. In the interest of better elk management, it is imperative State plans and forest plans address the same issues. Elk vulnerability is the framework issue.

We present an initial overview under which individual forests can creatively address elk management and yet retain consistent and cohesive approaches within regional and State boundaries. Emphasis has been placed on process, content, and implementation of new information rather than on numerical standards, although these remain important for measuring success. Specific process guidance for biologists in the Northern Region of the Forest Service is provided in the appendix.

KEY COMPONENTS OF ELK MANAGEMENT

The relationship between National Forest lands and elk needs to be recognized for the following key components:

1. Habitat in which elk grow, reproduce, and exist as elements of biological diversity.
2. The basis upon which State management programs depend. While hunting mortality accounts for upward of 90 percent of elk mortality, the States depend on habitat availability and condition for their programs to exist.
3. Sites for the public to have the opportunity to hunt and view elk. Recreation is an important product of National Forest lands. In most areas, use of forests peaks during fall hunting seasons, but in other areas wildlife viewing is a year-around product. The setting needs to be considered along with other habitat issues.
4. Maintenance of elk as a part of the natural community and recognition of elk habitat in a landscape context and in response to natural processes.

These key components can be recognized and evaluated in the following three types of habitat considerations:

Habitat effectiveness: This is a measure to be applied to nonhunting, summer and fall habitat situations. It was developed from research related to the ability of habitat to meet elk needs for growth and welfare requirements. It has been consistently misapplied as a measure of security during hunting season.

Elk vulnerability: This deals with security for elk during the hunting season. There is a rapidly expanding body of new information relating to this management concept that will be available for inclusion in forest plans as they are updated, revised, or amended.

Winter range: This has been a collective term referring to elk habitat during the nonsummer and fall, nonhunting season. However, during some years elk will move to winter habitat during the fall hunting season and, in most situations, become vulnerable. In updates, revisions, or amendments we must recognize and deal with this possibility as well as deal with traditional considerations.

HABITAT EFFECTIVENESS

Summer range includes the habitat used by elk from about late green-up (May) until they move to winter ranges, but prior to the hunting season. Summer range is the complete matrix upon which elk herds depend for growth, reproduction, and thrift. Management focus is on maintaining the ability of the habitat to meet elk needs for forage, water, seclusion, and special features (such as licks and moist areas). Forest Service lands that support summer range are the basis for State elk management; specifically, if habitat is degraded or poorly managed, the elk population will be degraded and, thus, directly influence State elk population management programs.

Habitat effectiveness is defined as the percentage of available habitat that is usable by elk outside the hunting season (Lyon and Christensen 1992). This is the measure of success in meeting elk needs on summer range. Based on years of research from various sites in Montana and Idaho, relatively sophisticated technologies exist for calculating habitat effectiveness. In forest plan revisions, updates, and amendments, this term should be used as a measure of summer range ability to support elk. Sources of information for habitat effectiveness and the major factors that influence it are included in Irwin and Peek (1979), Leege (1984), Lyon (1983, 1987), Lyon and others (1985), Thomas and others (1979), and Wisdom and others (1986). (See the References section at the end of this publication.)

Considerations for Forest Plans Related to Habitat Effectiveness

The following list is not inclusive but does cover the main issues managers need to consider.

1. *Roads*—density (miles per square mile), construction standards, seasons of use, method of closure.
2. *Special features*—wet sites, riparian habitat, licks, movement corridors.
3. *Cover*—extent, shape, size, connectiveness.
4. *Scale of analysis*—site specific, herd unit, habitat analysis unit.
5. *Spatial relationships*—intermingled ownerships, adjacent administrative units, district or forest "averaging."
6. *Domestic livestock*—forage and spatial competition.

Recommendations

Roads—Roads are undoubtedly the most significant consideration on elk summer range.

1. Use figure 1 (Lyon 1983) road model for determining habitat effectiveness related to roads. Avoid classifying roads as primitive and downgrading their effect unless they really are.
2. Discuss methods of closure. For elk, physical closure with "trashing" is desirable for year-long closure. Area closures are needed where terrain features and cover characteristics do not favor closure with gates or barriers. Honor systems of closure have been only moderately successful, at best.
3. Discuss construction standards. Where roads will be system roads, strive for construction and design features that lay lightly on the land. Identify temporary roads where they are an option. Avoid "tie through" systems where possible. Strive for minimum miles of new construction in summer range. Identify logging technology that reduces road construction. Avoid key habitat features when locating roads.
4. Any motorized vehicle use on roads will reduce habitat effectiveness. Recognize and deal with all forms of motorized vehicles and all uses, including administrative use.
5. Levels of habitat effectiveness:

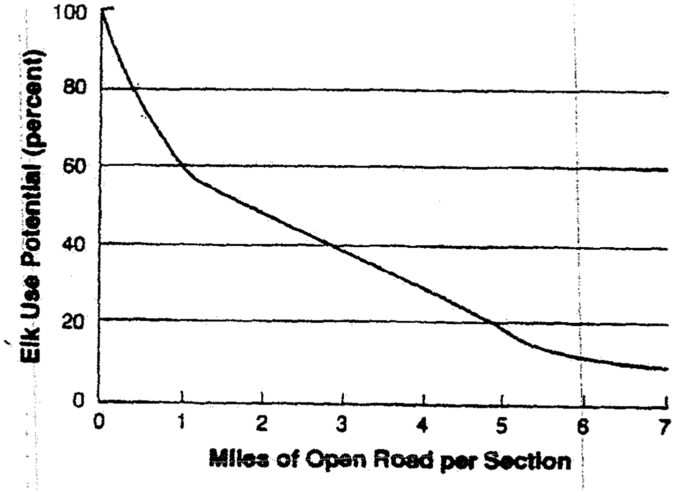

Figure 1—Habitat effectiveness for elk determined by road density (Lyon 1983).

0.75? 2.2 mi ~50%

a. For areas intended to benefit elk summer range and retain high use, habitat effectiveness should be 70 percent or greater.

b. For areas where elk are one of the primary resource considerations habitat effectiveness should be 50 percent or greater.

c. Areas where habitat effectiveness is retained at lower than 50 percent must be recognized as making only minor contributions to elk management goals. If habitat effectiveness is not important, don't fake it. Just admit up front that elk are not a consideration.

d. Reducing habitat effectiveness should never be considered as a means of controlling elk populations. A population over target is not a Forest Service habitat problem. Remember that in most situations, populations can be reduced through hunting.

Special Features—Wet drainage heads, saddles, riparian habitats, shadowed draws with cool air movement, and wet meadows are some examples of special features. In many areas these features support a disproportionate level of elk use and contribute significantly to overall elk use of a larger area. Generally, these sites are highly desirable for forage, water, temperature regulation, movement, or a combination. Such sites should be recognized and protected in prescriptions that deal with elk summer range. Logging activities, road locations, and siting of structures or activities should all be evaluated. Avoid damaging these features where elk are a benefiting resource (Lyon and others 1985).

Cover—Early guidelines greatly emphasized analysis of cover, specifically thermal and hiding cover (Thomas and others 1979). Today, detailed analyses of hiding and thermal habitat components are not considered as essential except in habitats with high natural levels of openings or where conifer cover is at a premium. Some approaches have created the classification "optimal cover" (Wisdom and others 1986) as an aid in analyzing cover from aerial photographs. Another approach, where stand analysis data are available, is provided by the HIDE2 hiding cover computer model (Lyon 1987).

While we still need to recognize the importance of maintaining cover blocks and movement corridors, a more meaningful approach to cover analysis includes maintenance of security, landscape management of coniferous cover, and monitoring elk use with radio telemetry or other means. Recognition that summer cover blocks are also fall hunting season security areas is an important coordinating consideration.

Cover unit size, patterns on a landscape basis, connectiveness with other cover, the amount of cover available to elk, and known use patterns by elk should be considered in prescriptions.

Scale of Analysis—Early guidelines tended to be project specific in scale; often 3,000 to 10,000 acres was recommended. However, while road locations, special features, and the location of cover or cutting units still need project-level analysis, such analysis also needs to recognize the project in a broader context of herd units (where known), habitat analysis units, or other meaningful, larger scale perspectives. Herd units need to be identified in cooperation with State biologists. Consideration of project-level effects may necessitate analysis in light of influences on adjacent herd units, adjacent forests, or even adjacent States over landscape units from 30,000 to 150,000 acres.

Another consideration in establishing factors for scales of analysis are known movement patterns. If your management of summer range may influence elk in terms of their movement to adjacent fall or winter ranges, the scale of analysis should be appropriate.

Spatial Relationships—This criterion has to do with habitat features, values, or project analyses that have a relationship to intermingled ownerships, concurrent and adjacent activities, or adjacent features that are significant to your concerns for elk habitat.

When elk habitat crosses intermingled ownerships, activities that reduce habitat effectiveness on intermingled lands require the Forest Service to decide how they will be dealt with in prescriptions. Adjacent and concurrent activities beyond Forest Service control, such as logging and grazing on private land, should be recognized in prescriptions, and courses of action for the Forest Service should be identified. Federal managers need to coordinate with State biologists on these activities.

Internally controlled activities that affect elk summer range should also be recognized. An example is the relationship of herd units or analysis areas to each other and, collectively, to forest elk habitat. Each individual unit should have an identified role for elk and a level of habitat effectiveness. In this way, the whole area or forest can achieve an expected level. It is undesirable to play off one unit against another. For example, recognizing high habitat effectiveness values in adjacent wilderness areas should not be a justification for excessive reductions in habitat effectiveness in managed areas, even if some average level for the forest is met.

Significant reductions in habitat effectiveness in areas identified as benefiting elk cannot be recovered at a pace equal to our ability to move activities around a forest. In addition, patterns of recreational activity related to elk can be significantly affected by this type of management.

Domestic Livestock—Current perspective is that cattle on elk summer range are not as significant a conflict as formerly thought and probably only warrant analysis where local understanding indicates a problem may exist. Elk appear to avoid areas where cattle are present if other options exist. Where no other options exist, elk will tolerate some cattle use.

Major points of conflict are wet sites and gentle terrain with succulent vegetation. Season-long cattle occupation of these types of sites undoubtedly reduces their value to elk.

Forests where cattle are a concern need to work with State biologists on standards and guidelines for cattle and elk relationships.

Of equal concern is the perception that elk herd expansion is causing cattle use reductions on National Forests. In developing management guidance, forests should address this issue and strive to gather habitat use data that will help clarify this situation.

Summary for Summer Range

1. Habitat effectiveness is the method of measurement.
2. The presence and motorized use of roads is the major impact on elk habitat effectiveness.
3. Detailed cover: forage analysis is important only when cover is at a premium.
4. Landscape levels of analysis are necessary.
5. Recognition of adjacent activities, intermingled ownerships, and cumulative effects is needed in plan revisions and updates.
6. Analysis of elk and domestic livestock conflicts is probably warranted where it is considered a problem locally.
7. Forests should set standards for habitat effectiveness that are congruous with goals for a prescriptive unit. Specific prescriptive guidelines should reflect the level of habitat effectiveness desired.
8. Close coordination with State biologists and recognition of identified State management goals for elk are necessary in all aspects of summer range management.
9. Forests should recognize traditional uses of elk as well as burgeoning nonconsumptive interest in elk.

ELK VULNERABILITY ANALYSIS

The primary source of elk mortality is hunting. While the State manages hunters, the Forest Service management of access and cover are extremely influential in affecting the ability of hunters to kill elk. Therefore, it is important that in forest plan revisions or updates, prescriptive guidance is identified for elk vulnerability analysis. This procedure applies during the hunting season and is not to be confused with habitat effectiveness. Vulnerability is a separate issue that forests need to recognize in elk management and write into prescriptions. Vulnerability results from an extremely complex relationship involving access, cover, topography, hunter density, and weather. A great deal of intercorrelation among these factors exists, and a great deal of cooperation between agencies will be necessary to achieve the goals of elk vulnerability management.

The measure of success for elk vulnerability is the level of compatibility between Forest Service and State management plans. Often, this will be the number of bulls per hundred cows surviving the hunting season or some expression of the quality of the recreation experience provided.

Considerations for Forest Plans Related to Elk Vulnerability

The following list is not inclusive but does include the main issues managers need to consider:

1. *Roads*—season of use, density.
2. *Security areas*—distance from roads, size, cover characteristics, closures (area), topographic characteristics.
3. *Cover management*—description, connectiveness, scale, terrain relationships.
4. *Mortality models*—demonstrated predictors of elk mortality based on habitat quality, hunter density, or other factors.

Recommendations

Roads—As with habitat effectiveness, access to and use of roads appear to be the most significant factors in vulnerability analysis.

Two studies in Idaho have demonstrated direct relationships between levels of road access and bull mortality (Leptich and Zager 1991; Unsworth and Kuck 1991). In Montana, Youmans (1991) implicated "road densities as the key factor in increased elk vulnerability."

Concerning open roads during hunting season, forests should develop criteria that meet State management goals for elk. Information on the relationship between roads and elk vulnerability is so new that specific criteria are scarce. However, the studies in northern Idaho provide initial guidance. Unsworth and Kuck (1991) found bull survival more than doubled in situations comparing road densities in excess of 4 miles per section with densities under 0.5 mile per section. In a different study area, Leptich and Zager (1991) reported bull mortalities of 62, 45, and 31 percent in study areas with 4.5, 2.6, and 1.0 miles of open road per section. In both these studies, cover during the hunting season was not considered limiting.

1. In areas with heavy cover, road management can be extremely influential in meeting desirable postseason bull:cow ratios.
2. Where heavy cover is not available, reduced open road densities contribute to maintaining some level of quality hunting opportunity through the season and to meeting postseason bull:cow ratios. In areas of

more open cover and, perhaps, gentler terrain, roads speed up the harvest of available bulls and make bulls more vulnerable throughout the season. Increased emphasis should be placed on security where poor cover conditions exist.

3. Even primitive roads that see little summer use are often used extensively during the hunting season. Area closures with open routes designated will most likely provide better security than individual closures. Area closures should address all motorized vehicles including all-terrain vehicles.

Security Areas—Security is the result of a combination of factors that allow elk to remain in a specific area while under stress from hunting. In Forest Service management, such areas are defined by cover blocks and road management. Specifically, these are areas of coniferous cover large enough and far enough away from open roads to provide security. There have been efforts on the Lolo and Deerlodge National Forests to develop criteria for managing security. The "Hillis paradigm" (Hillis and others 1991) provides these criteria and, with careful consideration, may be appropriate for other forests to use as a general guide. Briefly, this model identifies the size (250 or more acres), shape (nonlinear), and distance from open roads (over 0.5 mile) for security areas as well as how much of the area (over 30 percent) should be dedicated to security.

In discussions with biologists in Idaho and Montana, there appears to be a gradient from west to east regarding the significance of cover in this equation. In northern Idaho, it appears that open road density, hunter numbers, and topographic roughness are the major considerations (Unsworth and others 1993). Cover is so ubiquitous that security can be controlled with road management alone. As you move east into Montana and over the Continental Divide, cover considerations become more important because cover is less abundant and less contiguous. It is extremely important for forest biologists to work with their State counterparts in developing criteria for security areas, including their size, extent, distance from roads, and vegetative characteristics. Data from radio telemetry studies are the best source for developing such criteria.

Cover Management—This criterion is directed mostly at the more naturally open elk habitat in central and southwestern Montana and southern Idaho where care must be taken to recognize and retain adequate coniferous cover. In developing this criterion, a landscape-level perspective is absolutely necessary. Size, location on the landscape, connectiveness with other cover, and vegetative composition are important considerations (Hillis and others 1991). Data from Montana hunting seasons suggest that elk are less selective about the specific vegetative characteristics of coniferous cover and more responsive to size of units, connectiveness with adjacent units, and the scale of cover on the landscape (Lyon and Canfield 1991). A strong relationship exists between maintaining cover for summer range habitat effectiveness and maintaining the same cover for security during fall hunting. Where coniferous cover may be a limiting factor, it will be important to develop long-term perspectives (rotation length) on cover management that address condition, quantity, location, and configuration.

Mortality Models—Models that link habitat, hunter density, and elk mortality can provide guidelines to coordinating habitat condition and State management objectives. Unsworth and others (1993) have developed a model for northern Idaho that predicts bull elk hunting season mortality using open road density, circular standard deviation of aspect, and hunter intensity (density for the length of the season). This model virtually requires a computerized Geographic Information System for calculating the aspect variable. But the effect can be estimated based on the fact that greater topographic relief reduces elk vulnerability. The more moderate the topography (fig. 2), the more impact road density and hunter density have. If we assume average topography and around 10 hunter days per section spread over a 26-day season, the probability of mortality for a bull elk is 60 percent greater in an area with 1 mile of road per section than in an unroaded area. Likewise, 2 miles of road per section will more than double the mortality probability, and at higher road densities bulls usually do not survive the hunting season.

Using a different measure of hunting intensity, Vales and others (1991) and Vales (1993) presented data from northeastern Oregon indicating that the ratio of hunters to available elk can also provide an estimate of probable mortality; basically, there is a consistent increase in harvest rate as the number of hunters per elk increases (fig. 3). These data are important because they indicate that excessive hunting pressure can, in the end, overwhelm all other provisions of elk vulnerability management.

Summary for Elk Vulnerability

1. Roads appear to be the single most important variable that the Forest Service manages. Roads not only directly affect elk mortality but also affect hunter opportunity by accelerating bull mortality. Forests must work closely with State biologists to identify acceptable levels and locations of motorized access to meet postseason bull:cow ratios and maintain optimum hunter opportunity.

2. Security area definition is variable across the region. Some forests have developed criteria. It is essential that cooperation and coordination with State biologists be used to formulate criteria.

3. Elk vulnerability analysis, a new concept, will be further defined. Hunter density and opportunity

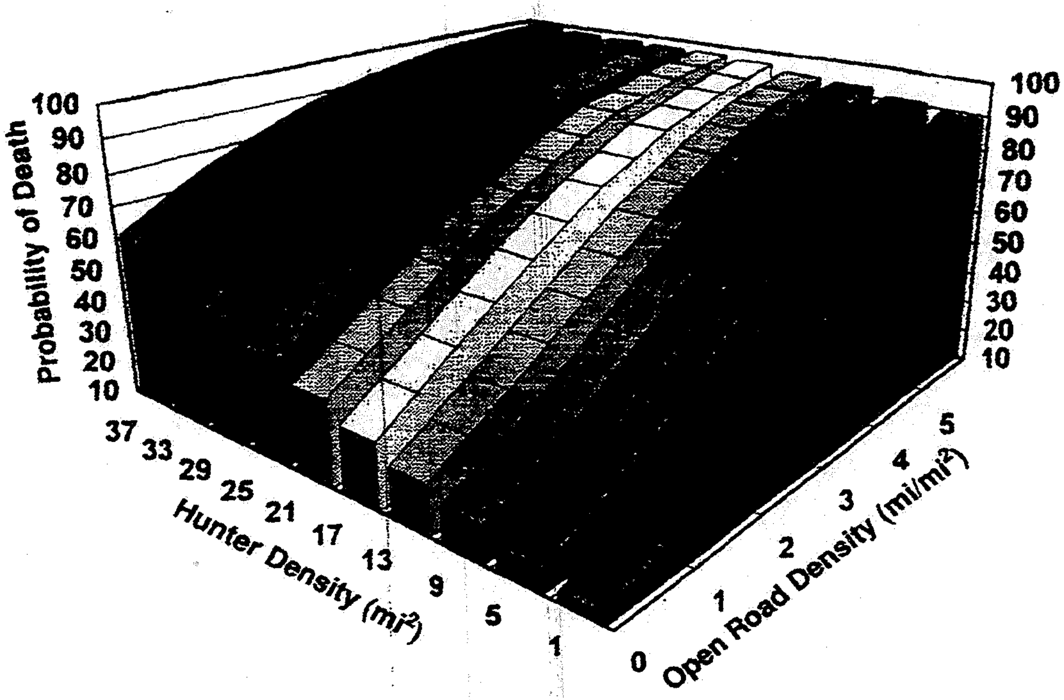

Figure 2—Elk vulnerability influenced by hunter density and road density (Unsworth and others 1993).

afforded by State regulations are also major components. It is essential that forest biologists and planners and their State counterparts communicate and coordinate extensively on this topic as forest standards and guidelines are developed.

4. Recently available mortality models can establish numerical standards for elk mortality. Local data bases may exist to help tailor mortality models to specific geographic areas. Numerical standards for elk mortality can be established through coordination with State biologists.

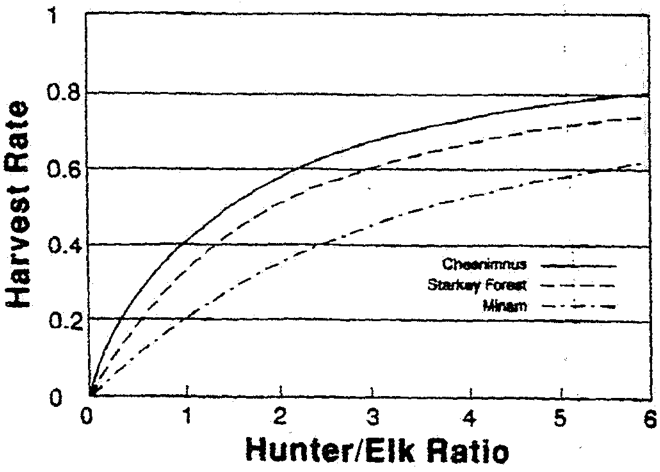

Figure 3—Elk vulnerability influenced by hunter to elk ratio (Vales 1993).

WINTER RANGE

Management of winter range remains the single most site-specific consideration for elk habitat. Each winter range is unique in some way. In this section, we briefly address the traditional considerations that already appear in the majority of forest plans. We again mention, however, that winter range should be evaluated as a part of the vulnerability assessment where appropriate to do so.

Traditionally, winter ranges for elk have been viewed as geographic sites on which animals concentrate seasonally because of snow depths. Heavy utilization of available plants, and animal die-off in severe winters, have been commonly recorded. For many years, the primary objective of management was to improve, or at least prevent deterioration of, existing vegetation.

In recent years, our understanding of animal physiology on winter ranges has modified this view. Forage is important, but in severe weather many animals substitute an energy-conservation strategy for forage intake. Thus, management of winter range to improve thermal cover and prevent harassment may be as important as anything done to change forage quantity or quality.

Considerations for Forest Plans Related to Winter Range

The following list is not inclusive but does include the main issues managers need to consider:

1. *Forage quantity and quality*—methods for improvement.
2. *Thermal cover*—energy conservation considerations.
3. *Roads and other disturbances*—energy conservation considerations.
4. *Livestock management*—forage allocation management.

Recommendations

Forage Quantity and Quality—In the majority of situations, actually modifying forage quantity or quality on the winter range is a difficult management challenge. Encroaching vegetation can sometimes be removed mechanically or with fire, and large or decadent shrubs can be burned to produce resprouting.

Thermal Cover—Some winter ranges lack thermal cover, which does not mean thermal cover serves no purpose where it is available. Where behavior patterns have been recorded, elk select resting and feeding sites based on control of energy transfer rather than forage availability. We recommend selective retention of larger trees where possible.

Roads and Other Disturbances—Disturbance and harassment result in tremendous energy costs to wintering animals. Selective road closures and restrictions on recreational use have proved effective in reducing these costs.

Livestock Management—Appropriate management of domestic livestock can, in some cases, be an important consideration in management of elk winter ranges. Local range specialists should be consulted about grazing techniques designed to leave adequate winter forage for elk.

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INT-GTR-86.

AVIAN COMMUNITIES IN THE PINYON-JUNIPER WOODLAND:

A DESCRIPTIVE ANALYSIS

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ABSTRACT

The expansive range and elevational distribution of the pinyon-juniper woodland in the western United States contributes to the wide variety of forms of this habitat type. Similarily, the breeding-bird community expresses this variety. A total of at least 73 different bird species are known to breed here. About 31 of these species breed with regularity in pinyon-juniper woodlands. Only about 5 of these species are restricted to this habitat type. Usually less than half of the breeders are permanent residents. A high proportion of the breeding birds forage for seeds or insects on the ground. The number of species that breed in cavities and/or forage on trunks and branches is positively correlated with pinyon pine density. Seasonal densities of breeding birds vary greatly depending on annual fluctuations in precipitation and seed and berry production. Winter diversity and density is strongly correlated with juniper berry production. Both junipers and pinyons show an adaptive suite of characters for dispersal by birds.

KEYWORDS: pinyon pine, juniper, avifauna, guilds, diversity, density, breeding-birds, winter birds.

The pinyon-juniper woodland could be labeled the characteristic habitat-type of the southwest because of its expansive range.

Extending over large areas of Arizona, Colorado, Nevada, New Mexico, and Utah it occupies somewhere between 43 and 76 million acres of land in the West. In New Mexico alone the woodland covers over 32,000 square miles or 26 percent of the state (Pieper 1977). The woodland stretches from the east slope of the Sierras to Oklahoma and from Oregon to Texas and into Mexico. It is the common vegetation-type of the foothills, low mountains, escarpments, and mesas of the southwest (Fig. 1). Throughout its range this "pygmy forest" shows broad tolerance limits ranging in elevation from a high of 10,000 ft. in the Sierras to a low of 3200 ft. in the four corners area, with junipers alone extending even lower in many areas (West et al. 1975). It is found on a variety of soils derived from granite, basalt, limestone, and mixed alluvium (Hurst 1975).

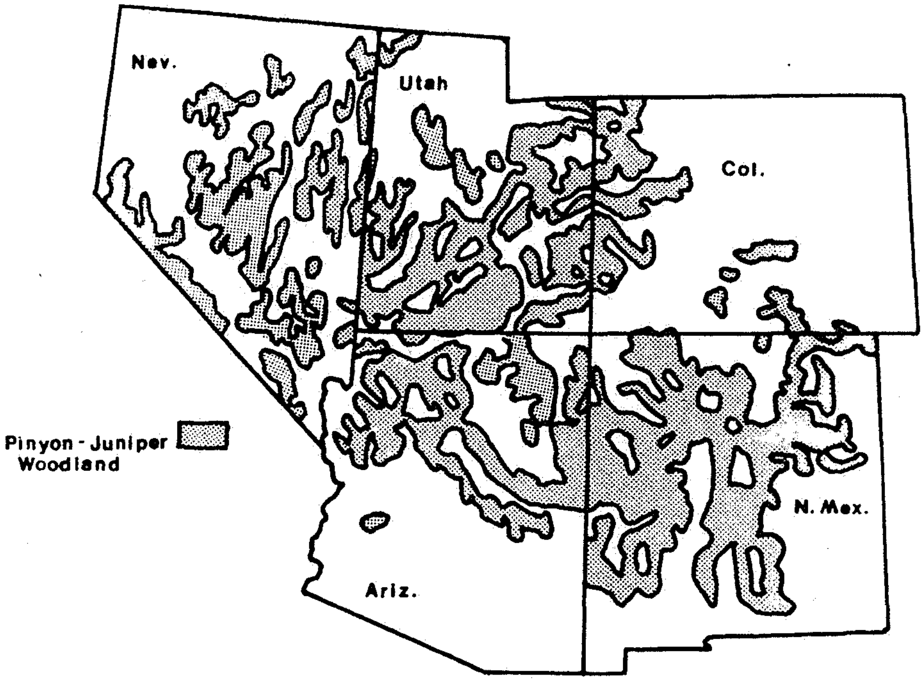

Figure 1. The distribution of pinyon-juniper woodland in the five western states where it is most abundant (From Clary 1975).

4 sps.

The major trees of this woodland consist of four species of junipers, Juniperus occidentalis, J. deppeana, J. monosperma and J. osteosperma. The latter species is the most wide-spread of the junipers. The two most common pinyon pines are Pinus monophylla and P. edulis with the latter species having the most extensive distribution. The dominant trees of the area are relatively small (hence the name pygmy forest) ranging in height from 15-40 ft. with individual trees having dense foliage. In general the junipers are more drought-resistant than pinyon pines and therefore occur in highest densities at lower elevations, whereas, pinyon pines become more abundant at higher elevations in this woodland (Short and

McCullock 1977). The density of these trees varies from very sparse to very dense depending on elevation, climate, and soil type. Total plant cover increases with elevation up to about 6600 ft. (Tueller et al. 1979).

The understory vegetation of the pinyon-juniper woodland is highly variable depending on soil type, exposure, and climatic pattern. Tueller et al. (1979) lists 240 positively identified species of vascular plants from the Great Basin pinyon-juniper woodlands. The list includes 67 species of shrubs and succulents, 46 grasses, and 122 forbs that grow under pinyon and juniper trees. Major shrubs include sagebrush (*Artemisia* sp.), bitterbrush (*Purshia tridentata*), rabbitbrush (*Chrysothamnus* sp.), and various species of oaks (*Quercus* spp.). Few of these species are found growing in association with one another, as the understory is reasonably depaupered. None of the shrubs, succulents, grasses or forbs are listed as rare and endangered and none are restricted to this vegetation type. Most woodlands contain only a few of these species. Thus, plant species diversity (as well as density) is reasonably low compared to other vegetation types in the southwest.

The climate of this vegetation-type can be summarized as being rather severe with hot summers, cold winters, low amounts of precipitation in the form of rain and snow, low relative humidity and high winds. Mean daily maximum temperatures for the hottest month of the year vary from 26°C to 36°C. Total yearly precipitation varies between 8 and 18 inches (West et al. 1975).

The lower limits of this woodland now mingle with grassland, desert scrub, Great Basin Desert or shrublands in different parts of its range. Because of climatic cycles (cool, moist to hot, dry) this lower boundary has been very active during the last 10,000 years (Martin and Mehringer 1964, Wright et al. 1973, Wells and Berger 1967). Evidence from pollen deposits, sloth dung, and wood rat middens indicate a considerable lowering of this boundary. This depression caused isolated areas of the woodland to come into contact with other such areas thus increasing the potential for redistribution of the flora and fauna. The return of a warmer, drier climate caused an upward retreat leaving behind isolated relict pockets of pinyon-juniper woodland, with its faunal components.

Even though early settlers heavily used pinyons and junipers for mine props, fence posts, and fuel, during the last 130 years the vegetation type is undergoing an expansion into low shrublands, grasslands and Great Basin Deserts (West et al. 1975). At the same time the density of trees in more permanent stands is also increasing. Numerous causes have been proposed to explain this increase, but the major culprit seems to be overgrazing by cattle and sheep (Aro 1971). Improper grazing has reduced forage production thereby releasing the trees from competition with the herbs and shrubs. Johnsen (1962) believes the spread of juniper in northern Arizona is due to the increased spread of seeds by livestock, lack of periodic fire, overgrazing which reduces competition of grasses with juniper seedlings, and a gradually changing climate which favors the spread of juniper. La Marche (1974) presents evidence that the period from 1850 to 1940 was wetter and warmer than the period before or after this.

It is evident that this woodland as a whole is an extremely complex, variable community. As stated by West et al. (1975), "Early attempts to explain distribution, composition, successional changes, and management responses in terms of single factors were overly simplistic. These variations can be better explained in terms of a complex of environmental patterns, historical events, and successional mechanisms. The relative importance of each factor of the environmental complex varies with the synecological context."

2-3 yrs Juniper

A major characteristic of this woodland as far as birds are concerned is the periodic production of vast quantities of pinyon pine seeds and juniper berries. Large crops of pine seeds are produced once every five or six years whereas juniper berry production occurs every two to three years. In many years neither tree forms reproductive propagules. Both life-forms appear to have intra-specific synchrony. For example, in a year of a good berry crop, one hectare contains between 19 and 38 million berries. A cubic meter of foliage holds 20,000 berries. The number declines steadily through the late fall and winter as birds and mammals consume them. The flesh of a single berry has about 315 calories making it a desirable source of energy. The berries are a shiny blue in color making them conspicuous; they ripen in the fall when insects are sparse and bird densities are high due to migration (Salomonson 1978). Thus junipers have adaptations favoring zoochory (Morton 1973). The pinyon pine also has a constellation of adaptations that favor dispersal by animals, especially birds (Table 1) (Vander Wall and Balda 1977). This pine not only allows animals easy access to its seeds but may entice dispersal agents. This means the seeds are easily located, extracted from the cones and eaten or cached for future use (Vander Wall and Balda 1977, Ligon 1978). More Pinus edulis seeds are cached in dry, exposed soils than can be used by the birds in years of high cone crops. In some years, pinyon pines produce absolutely no cones per hectare (Balda, unpubl. data), whereas in other years they may produce as many as 1800 cones/tree (Ligon 1971). These seeds are extremely nutritious, containing about 7400 cal/g (Little 1938). A pinyon pine seed contains 14.5 percent protein, 60 percent fat, and 18.7 percent carbohydrate (Botkin and Shires 1948). The large size, high energy content, and high protein level makes this seed a highly desirable food stuff.

Management of pinyon-juniper woodlands since the mid-40's has largely consisted of control of the spread of junipers (and in some cases pinyon) into grasslands and type-conversion of pinyon-juniper woodlands into grazing lands. Both eradication of the type and control has been justified on public lands because the trees are generally considered as of low commercial value relative to other harvestable trees of the West. During the period 1950 to 1964 Box et al. (1966) estimate that approximately three million acres of pinyon-juniper woodland were converted to grazing lands. Between 1950 and 1961, more than one million acres were converted in Arizona alone (Arnold et al. 1964).

The major objective of most type-conversion projects, often referred to as "Range Improvement Projects" is to produce additional forage for livestock (Terrel and Spillett 1975). These conversions represent "a change from multiple use to one use, grazing" (Little 1977). Land managers today are going through a period of cautious soul-searching about how to proceed with management on these lands. This treatment has not been popular, leaves the area an aesthetic diaster, has questionably proven long range benefits and "Most questions concerning wildlife and pinyon-juniper range conversion are unanswered and probably will remain so." (Terrel and Spillett 1975). The best synopsis of pinyon-juniper management and guidelines for future use can be found in a symposium edited by Gifford and Busby (1975).

TABLE 1. Features of pines with different dispersal strategies

Characteristic	*P. ponderosa* (wind)	*P. edulis* (animal)
Seed size	small	large
Seed conspicuous	no	yes
Seed quickly released	yes	no
Seed coat labelled	no	yes
Synchronous cone opening	yes	no
Position of cone	down, out	up, out
Cone scales	present	absent

BREEDING BIRDS

A total of 73 different species of birds are reported to breed in pinyon-juniper woodlands (Table 2). Undoubtedly more records are known but these will most often be rare or unusual occurrences. These 73 species are taxonomically aligned in 8 orders and 25 families. Because of the geographic area span by this plant community and the wide physiognomic variety (over its range), no one area contains near this breeding diversity. For example, in north-central Arizona 5 pinyon-juniper plots were sampled intensively during two breeding seasons (Grue 1977, Masters 1979) and the number of breeding species per 40 ha plot ranged from 12 to 24 and averaged 19 species. Rasmussen (1941) reports 43 species inhabiting the pinyon-juniper woodland on the Kaibab Plateau in summer but has good evidence for breeding by only 12 species. Hardy (1945) lists 22 species as regular breeders in Utah pinyon-juniper woodlands whereas Hering (1957) reports 15 breeding species.

Relatively few of the 73 species are restricted to pinyon-juniper woodland. Table 2 lists 5 obligates and 13 semi-obligates. An obligatory species is defined for purposes of this presentation as one which nests only in pinyon-juniper woodland within a geographic area that contains other habitat types. A semi-obligatory species may nest in one additional plant community. This definition is knowingly broad as most of these species nest in different habitat types in portions of their range where pinyon-juniper woodland is absent. Hardy (1945) mentions only the Piñon Jay and Plain Titmouse as being obligatory and the Bushtit as a semi-obligatory species in this woodland type.[1/] But, the Piñon Jay often nests and forages in ponderosa pine forest (Balda and Bateman 1971) and the Bushtit is also known to use other habitats.

[1/] Scientific names for all birds mentioned in the text or tables are given in Appendix I.

TABLE 2. Breeding birds of pinyon-juniper woodlands1)

Species	Status2)	Distribution3)	Niche Width 4)
Turkey Vulture ✓	S	4	
Cooper's Hawk ✓	P	2	
Red-tailed Hawk ✓	P	4	
Swainson's Hawk ✓	S	1	
Ferruginous Hawk ✓	P	2	
Golden Eagle ✓	P	3 (Sp.)	
Prairie Falcon ✓	P	2 (Sp.)	
American Kestrel ✓	P-S	4	
Gambel's Quail	P	2	
Mourning Dove ✓	P-S	7	
Screech Owl ✓	P	3	obligatory
Great Horned Owl ✓	P	3	
Long-eared Owl ✓	S	1	
Saw-whet Owl ✓	P	1	
Poor-will ✓	S	2	
Common Nighthawk ✓	S	3	
Lesser Nighthawk	S	2	
White-throated Swift ✓	S	2 (Sp.)	
Black-chinned Hummingbird	S	4	semi-obligatory
Costa's Hummingbird	S	1	
Broad-tailed Hummingbird	S	3	
Common (red-shafted) Flicker ✓	P	6	
Hairy Woodpecker ✓	P	4	
Ladder-backed Woodpecker	P	1	
Western Kingbird	S	1	
Cassin's Kingbird	S	2	
Ash-throated Flycatcher	S	7	semi-obligatory
Say's Phoebe	S	2	
Gray Flycatcher	S	2	obligatory
Western Wood Pewee ✓	S	1	
Violet-green Swallow ✓	S	2	
Cliff Swallow ✓	S	1 (Sp.)	
Scrub Jay	P	6	obligatory
Black-billed Magpie ✓	P	2	
Common Raven ✓	P	6	
Piñon Jay ✓	P	5	semi-obligatory
Mountain Chickadee ✓	P	3	
Plain Titmouse	P	7	obligatory
Bushtit	P	6	semi-obligatory
White-breasted Nuthatch ✓	P	3	
House Wren ✓	S	1	
Bewick's Wren	P-S	3	semi-obligatory
Cañon Wren ✓	P-S	3 (Sp.)	
Rock Wren ✓	P-S	4	
Mockingbird	S	4	semi-obligatory
American Robin ✓	P-S	1	
Western Bluebird ✓	P-S	1	
Mountain Bluebird ✓	P-S	4	
Blue-gray Gnatcatcher	S	5	semi-obligatory
Loggerhead Shrike ✓	S	2	

36+
15
51

TABLE 2. (cont.)

15

Species	Status[2)]	Distribution[3)]	Niche Width[4)]
Gray Vireo	S	3	obligatory
Solitary Vireo ✓	S	2	
Black-throated Gray Warbler	S	5	semi-obligatory
Scott's Oriole	S	3	
Brown-headed Cowbird ✓	S	4	
Hepatic Tanager	S	1	
Black-headed Grosbeak ✓	S	5	
Lazuli Bunting ✓	S	2	
Cassin's Finch ✓	S	1	
House Finch ✓	P-S	5	semi-obligatory
Lesser Goldfinch	S	2	
Red Crossbill ✓	S	1	
Green-tailed Towhee ✓	S	2	
Rufous-sided Towhee ✓	P	6	semi-obligatory
Brown Towhee	P	2	semi-obligatory
Vesper Sparrow ✓	P-S	2	
Lark Sparrow ✓	S	4	semi-obligatory
Black-throated Sparrow	P-S	3	
Sage Sparrow	S	1	
Dark-eyed Junco ✓	S	1	
Gray-headed Junco	P-S	1	
Chipping Sparrow ✓	P-S	6	
Brewer's Sparrow ✓	S	1	
Black-chinned Sparrow	P	2	semi-obligatory
Total: n = 74	P = 23 (32%) S = 38 (51%) P-S = 13 (17%)	s-o = 13 o = 5	

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1) Data from Rasmussen (1941), Hardy (1945), Miller (1946), Hering (1957), Grue (1977), Masters (1979)
2) P = permanent resident; S = summer resident
3) The number indicates the number of census plots or study areas used for breeding. The maximum is 7. Sp. = special landscape required.
4) Obligatory = in a given geographic area the species breeds only in the pinyon-juniper woodland; semi-obligatory = same as above but breeds in one additional plant community.

Few, if any other natural habitat-types in North America have so few truly obligatory species. The reason(s) such should be the case is not clear but may relate to the great physiognomic diversity found in the pinyon-juniper woodland. Just as there is no typical pinyon-juniper woodland there are few obligate pinyon-juniper birds.

Just as the number of breeding species varies between woodlands so does breeding bird density. In southwestern Arizona where many oaks are found in the woodlands breeding bird density may reach 250 pairs per 40 ha (Balda 1967). This density is seldom if ever reached in the pinyon-juniper woodland where densities vary between 30 and 190 pairs per 40 ha (Table 3). Ninety-five pairs is about an average figure. Grasslands usually have fewer breeding pairs and ponderosa pine forests more than the pinyon-juniper woodland.

TABLE 3. Characteristics of the avian woodland breeding birds

Study	Habitat	No. of Breeding Species	No. of Breeding Pairs/40 ha
Balda, 1967	oak-juniper	36	224
Balda, 1967	oak-juniper-pine	36	267
Grue, 1977	juniper-parkland	17-23	54-179
Beatty, 1978	juniper-grassland	11-12	35-40
Grue, 1977	juniper-pinyon	24-26	66-130
Masters, 1979	pinyon-juniper I	9-10	90-87
Masters, 1979	pinyon-juniper II	18-21	191-138
Masters, 1979	pinyon-juniper III	19-19	122-133
Hering, 1957	pinyon-juniper (?)	15	33
Beidleman, 1960	pinyon-juniper	2	30
Hardy, 1945	pinyon-juniper	22	--
Miller, 1946	pinyon dominated	55	--

Breeding bird densities in a single location show rather large annual fluctuations that appear to be linked to biotic and physical factors. In very dry years the breeding bird populations may be reduced between 50 and 70% (Grue 1977). Possibly pinyon pine seed crops may attract breeding birds the next spring. Masters (1979) found a 28% increase in populations after a large cone crop (Table 4).

Table 4. Changes in breeding bird densities (pairs/40 ha) and diversities between years

Study	First Year Density/Diversity	Second Year Density/Diversity	%Change	Reason
Masters, 1979	191/21	138/18	28/14	Pinon seed crop before first year
Grue, 1977	130/26	66/24	49/8	Annual fluctuation in precipitation
Grue, 1977	179/23	54/17	70/26	Same as above

Masters (1979) attempted to explain the relationship between various habitat parameters and characteristics of the breeding bird fauna. At the level of the community, she found that the number of breeding bird species was significantly correlated with a) the density of pinyon pine, b) total tree density and c) pinyon pine foliage volume. Foliage height diversity (as measured in two-meter height classes) was a significant predictor of bird species diversity. Breeding bird density was significantly correlated with pinyon pine density when the bird population figures following a large pinyon pine cone crop are ignored.

A "typical avifauna" of the pinyon-juniper woodland thus appears to be as simplistic an approach as trying to describe a typical vegetation for this woodland type. Never-the-less we have selected from the list of 74 breeding species a group that has a distribution score (Table 2) of four or higher and/or is listed as obligatory or semi-obligatory in niche width. A major danger here is that two closely related species may be sympatric and thus neither would have achieved the criteria for inclusion. Such could have been the case for nighthawks, kingbirds, hummingbirds, bluebirds, medium-billed sparrows and a few other cases. In these instances the most common of the dyad or triad was added to the list to make it as representative as possible. From Table 2, 29 species met the first criteria and the nighthawk and kingbird were added for reasons given above.

Resident Status

Of the 31 species that fit our "typical avifauna" criteria 14 (45%) are summer residents and 11 (35%) are permanent residents. Six species show variable patterns of residency either based on geographic considerations (i.e. summer residents in the northern portion of their range and permanent residents in the more southern areas) or variable weather conditions (i.e. migrate in harsh winter, remain stationary in mild winters). Hardy (1945) in eastern Utah described 36% of the nesting species as permanent residents and 64% as summer residents, almost identical to our typical avifauna if one includes the "switchers" in the summer category.

Data from intensively censused plots in central Arizona over a two year period showed about the same split as does the Utah data (Grue 1977). The proportion of permanent resident species ranged from 35 to 40%.

In north-central Arizona however, Masters (1979) censusing three pinyon-juniper plots for two years found a range of permanent resident breeders from 32 to 56% (Table 5), and Hering (1957) near Mesa Verde, Colorado had 53% permanent residents. One could expect permanent residency to increase in the woodlands with decreasing latitudes but such an increase is not apparent from either the proportion of the breeding population that is permanent or the absolute number of species that do so. On both of Masters' (1979) plots with proportion of permanent residence above 50% the ratio of pinyon to juniper trees was better than 2:1. (Hardy's 1945 ratio was 0.36 to 1). Hering (1957) did not provide the necessary data to assess this habitat feature but the general area of her study contains high densities of P. edulis (pers. obs. R. P. Balda). Of the 55 species of breeding birds (a woodland high!) listed by Miller (1946) in a southern California woodland predominated by pinyon pine, 27 species or 49% were apparently permanent residents. Two areas without pinyons had 33 and 35%

permanent resident breeders. Thus, we suggest with caution that a positive correlation may exist between the proportion of permanent residents in the community and the proportion of trees in the woodland that are pinyon pines. In all probability no one factor will answer the question, but this one does deserve future investigation. Of the 18 species listed as either obligatory or semi-obligatory in this woodland 8 are permanent residents.

Table 5. Residence status of breeding birds from specific sites in north-central Arizona (Masters 1979)

Status	Number of Species (%) Sites I		II		III	
Permanent Resident	3.0*	(32)	8.0	(53)	7.5	(56)
Summer Resident	6.5	(68)	7.0	(47)	6.0	(44)

* 2 yr. average

Foraging Guilds

An instructive way to look at avian communities is the use of foraging guilds (Root 1967). A guild is defined as one or more species in a community that use similar foraging techniques. Guilds can be defined as broadly or narrowly as the observations and data base permit. Here for the sake of simplicity and accuracy (but sacrificing specifics) I define foraging guilds only by substrate-type. This is done because very little information is known about the species under consideration to allow for finer distinctions. Foraging guilds used include ground, foliage, air, bark, and flowers. If a species used two of these substrates I assigned half the value to each guild.

The descriptive analysis from nine different intensively studied woodland sites shows few trends. The number of ground foragers varied from 6 (Hering 1957) to 16 (Grue 1977) species. Relative proportions of ground foragers varied between 40% (Hering 1957) and 57% (Grue 1977). No significant correlation (Spearman Rank Correlation) between the density of pinyon pine or juniper and either the number or proportion of ground foraging species was found.

The number of foliage foragers in the breeding community varied from a low of three in a juniper-grassland (Beatty 1978) to a high of 12 in a predominantly pinyon pine stand. The mean number of species that used foliage as a substrate where both pinyon and junipers were represented was 5. The number or proportion of foliage foraging species showed no significant correlation with pinyon or juniper density.

The number of species of hummingbirds (nectar feeders) also shows no correlation with tree species density. Hummingbirds most likely respond more to the species composition and flowering patterns of the shrub and forb strata which may be limited by physical factors (temperature, moisture, etc.).

There is also no trend for aerial feeders. Aerial foraging species number between 1 (Hering 1957) and 9 (Miller 1946). On areas containing both pinyon and junipers the mean number of aerial feeders was 4.

In some woodlands a small group of breeding species forage extensively on trunks and large branches. In no intensive study area analyzed for this report where the ratio of pinyon to junipers was less than 1:1 did any of these species breed. Where pinyons outnumbered junipers by 2:1 or better two species appeared. Almost invariably these two species were the Hairy Woodpecker and White-breasted Nuthatch. The former species obtains insects by hammering holes through the bark or flaking layers of bark off in small plates. The latter species probes the crevices in the bark to obtain insects. Both species reach higher densities in ponderosa pine forests (Szaro and Balda 1979) than in the woodlands. Either there are more insects in, under, and on pinyon pine bark than juniper or the bark pattern is such that insects are easily extracted.

The "typical avifauna" for pinyon-juniper woodlands has a slightly higher number of ground and foliage foragers than the studies described above (Table 6). This probably occurred because our selected sample of birds is slightly larger than would be found in any one woodland area.

Table 6. Foraging Guilds for a "typical pinyon-juniper woodland"

Guild	*Number of Breeding Species (%)
Ground	14.5 (52)
Foliage	7.0 (24)
Aerial	4.5 (16)
Bark	1.0 (4)
Flower	1.0 (4)

*Carnivores not included

The above analysis has dealt solely with numbers of species because of the high year-to-year variability in densities. Master's (1979) regression models to predict characteristics of the bird populations included foraging guilds. Eight independent foliage variables were used. Pinyon density was significantly correlated with densities of aerial feeders, bark feeders, and total density of all insectivorous birds (Table 7). No variable contributed solely by junipers was important as a predictor of any of the breeding bird characteristics measured. Why the above result should occur is not immediately obvious but suggests pinyon pine may provide a more suitable foraging substrate than juniper.

Only fragmentary data exists to support the contention that juniper is less attractive as a foraging substrate than is pinyon pine. In an oak-juniper-pine (_Pinus cembroides_ and _P. leiophylla_) woodland in southeastern Arizona, Balda (1969) studied foliage use by the 36 breeding species. The number of observations in each tree species were compared to the foliage volume contributed by each tree species. Based on foliage volume an expected number of bird observations per tree species was calculated. Actual foraging observations in juniper were far less than expected, whereas foraging observations in pines were much greater than expected. At that time Balda proposed that the breeding birds may simply have not yet learned to use juniper as it is known that juniper is presently spreading into new areas and increasing in areas where it was once sparse. The Black-throated Gray Warbler, Chipping Sparrow, Bridled Titmouse and Common Bushtit utilized juniper more than any other species. Three of the four species listed above are members of our "typical woodland avifauna." In a pinyon-juniper-ponderosa pine ecotone Laudenslayer and Balda (1976) found that pinyon pine was selected more intensely than predicted by expected numbers generated from foliage volume. Juniper was selected approximately as often as expected. We explained this difference by using the relative proportion of foraging surface within both trees. Although both species have their green foliage concentrated on the outer edges of the branches, needles of pinyon pine are found growing farther inward than in juniper. Thus, if the growing areas and areas of green vegetation on these trees are used as prime foraging surfaces then pinyon provides more of this surface per tree than does juniper.

Table 7. Percent variability explained (r^2) of breeding bird parameters by vegetation factors which are significantly correlated (Masters 1979)

Factor	Density of Feeding Guilds		
	Aerial Feeders	Bark Feeders	Insectivores
Pinyon Pine Density	.980	.781	.949
Total Tree Density	.979	.776	.947
Pinyon Foliage Volume	.902	NS	.834

Insect densities in pinyons and junipers may also be a reason why pinyon density is a good predictor of density of insect eating birds. Masters (1979) found, however, that junipers had a higher number of insect taxa than did pinyon. Insect abundance (as measured by total length) was about the same in both trees. The similarity coefficient (a measure of community similarity) indicated that pinyon and juniper have different arthropod faunas associated with them.

Nesting Guilds

The classification of the avian community by nesting habits may also provide clues as to how breeding birds interact with the structure of the vegetation. Of the 31 species used as a "typical avifauna" 60% (18.5) nested in foliage (the 0.5 is for the Mourning Dove that uses both foliage and ground for a nest substrate), 23% (7) used cavities and the remainder nest on the ground. Hardy's data (1945) fits well with 61% of the breeding birds nesting in the foliage, 21% in cavities, and 18% of the species nesting on the ground.

On two intensively studied plots in central Arizona Grue (1977) found an average of from 60 to 68% foliage nesting species in the breeding bird community. From 15 to 20% of the species nested in cavities. Again the fit is reasonable with what a "typical avifauna" would show (Table 8).

Table 8. Nesting guilds of breeding birds from specific sites in central Arizona (Grue 1977)

Nesting Guild		Number of Species (%) Pinyon-Juniper Woodland	Juniper Parkland
Foliage		16.5* (66)	14.0 (68)
Cavity		5.0 (20)	3.0 (15)
Ground		3.5 (14)	3.5 (17)
	Total	25.0	20.5

* 2 yr. averages

In north-central Arizona Master's (1979) found cavity nesters to make up almost half of the breeding species on areas where pinyons outnumbered junipers (Table 9). Hering (1957) found cavity nesting species made up 47% of the breeding species on an area of presumable high pinyon densities. Both studies had 7 to 8 cavity nesting species present. The pinyon dominated woodland in California (Miller 1946) contained 11 cavity nesting species.

Table 9. Nesting guilds of breeding birds from specific sites in north-central Arizona (Masters 1979)

Nesting Guild	Number of Species (%) Sites I	 II	 III
Foliage	7.0* (74)	7.5 (50)	6.5 (48)
Cavity	2.0 (21)	7.0 (47)	6.5 (48)
Ground	0.5 (5)	0.5 (3)	0.5 (4)

* 2 yr. averages

The emerging pattern is more than suggestive that cavity nesting species will occur with higher probability in woodlands containing large numbers of pinyon pines. On three study sites in north-central Arizona Masters (1979) found that 79% of the variability (r^2) in density of the combined cavity nesting species (not species numbers as discussed above) was explained by the density of pinyon pines.

Both density and diversity of cavity nesting species may be related to pinyon pine in some manner. Since cavity nesters depend on weakened or diseased trees to excavate cavities in, it is possible that pinyon pine are more prone to attack by insects and other disease causing organisms. Also, it may be that pinyon branches are more brittle and are therefore more prone to breaking thus allowing disease agents entry. Dead junipers are hard whereas dead pinyon pines contain soft wood (pers. obs.).

The question that remains deals with tree-type selection by the foliage breeding birds: Do foliage breeders select for either juniper or pinyon when choosing a nest site? Both Hardy (1945) and Short and McCulloch (1977) make unsubstantiated comments that foliage nesting birds prefer junipers over pinyons for nest-sites. Based on the amount of data presently available it is not possible to answer that question and more research is required to show if any preference is shown (Table 10). The two species that showed regular use of juniper were the Black-chinned Hummingbird and Black-throated Gray Warbler whereas the Chipping Sparrow showed no preference for either tree (Masters 1979, Balda 1969).

Table 10. Nest sites of foliage nesting birds in western woodlands

Study		Number of Nests in		
		pinyon	juniper	other
Balda, 1967	oak-juniper 67 1	(not present)	1	12
Balda, 1967	juniper-oak-pine 46 46 29	--	11	10
Laudenslayer and Balda, 1976	pinyon-juniper-ponderosa pine 48 46 27	3	--	--
Masters, 1979	pinyon-juniper I 32 33	--	1	--
Masters, 1979	pinyon-juniper II 87 33	10	5	--
Masters, 1979	pinyon-juniper III 87 34	6	2	--

WINTERING BIRDS

Winter bird populations of the woodland have been studied in central Arizona by Grue (1977) and in north-central Arizona by Shrout (1977). A total of 32 species have been recorded as wintering in these woodlands. These 32 species belong to five orders and 14 families. Of these, 18 are permanent residents, 10 are winter residents, and 4 are switchers. The most regular winter residents are the two species of juncos, White-crowned Sparrow, and Ruby-crowned Kinglet. Three of these four species are seed eaters. Prominent "switcher" species are the Mourning Dove, American Robin, the two bluebirds, and the House Finch. Only the Bushtit, kinglet and wren are insectivorous (Table 11).

Species numbers vary considerably from year-to-year. Shrout (1977) reported a diversity of 10 species in one winter and 20 the next on the same 40 ha plot. Mean number of wintering species in Arizona woodlands is about 15 (Grue 1977, Shrout 1977).

Table 11. Birds wintering in pinyon-juniper woodlands

Species	Status*	Distribution (max = 3)
Rough-legged Hawk	W	1
Merlin	W	1
Prairie Falcon	P	(pers. obs.)
Gambel's Quail	P	1
Mourning Dove	P-S	2
Hairy Woodpecker	P	1
Common (red-shafted) Flicker	P	3
Horned Lark	P	1
Common Raven	P	3
Piñon Jay	P	1
Scrub Jay	P	3
Mountain Chickadee	P	1
Plain Titmouse	P	3
Common Bushtit	P	3
White-breasted Nuthatch	P	(pers. obs.)
Red-breasted Nuthatch	W	(pers. obs.)
Bewick's Wren	P	2
Ruby-crowned Kinglet	W	2
American Robin	P-S	1
Townsend's Solitaire	W	1
Western Bluebird	P-S	1
Mountain Bluebird	P-S	1
Sage Thrasher	W	1
Evening Grosbeak	W	1
House Finch	P-S	2
Cassin's Finch	W	(pers. obs.)
Rufous-sided Towhee	P	1
Vesper Sparrow	P	1
Dark-eyed Junco	W	3
Gray-headed Junco	W	3
Chipping Sparrow	P	3
White-crowned Sparrow	W	3
n = 32	P=16, W=11 and P-S=5	

Winter densities vary greatly from place-to-place and from year-to-year. For example during the winter of 1973-74 Grue (1977) reported 318 individuals per 40 ha in a pinyon-juniper woodland and 251 wintering birds in a 40 ha juniper parkland. This is a 21% difference.

Year-to-year variations are even more striking. In some years the woodland supports huge flocks (too large to count) ~~of bluebirds~~, American Robins and mixed flocks of juncos. In other years one can walk for hours seeing only a very few birds (Vaughan pers. comm., R. P. Balda pers. obs.). Shrout (1977) found 293 wintering birds per 40 ha in the winter of 1973-74 and 75 individuals during the winter of 1974-75 on the same plot. Using a conservative calculation this is a 74% change in population density between years.