Terrestrial Wildlife in the Post-mined Appalachian Landscape: Status and Opportunities

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Abstract Coal mining is an anthropogenic stressor that has impacted terrestrial and semi-aquatic wildlife in the Appalachian Plateau since European settlement. Creation of grassland and early-successional habitats resulting from mining in a forested landscape has resulted in novel, non-analog habitat conditions. Depending on the taxa, the extent of mining on the landscape, and reclamation practices, effects have ranged across a gradient of negative to positive. Forest-obligate species such as woodland salamanders and forest-interior birds or those that depend on aquatic systems in their life cycle have been most impacted. Others, such as grassland and early-successional bird species have responded favorably. Some bat species, as an unintended consequence, use legacy deep mines as winter hibernacula in a region with limited karst geology. Recolonization of impacted wildlife often depends on life strategies and species' vagility, but also on altered or arrested successional processes on the post-surface mine landscape. Many wildlife species will benefit from Forest Reclamation Approach practices going forward. In the future, managers will be faced with decisions about reforestation versus maintaining open habitats depending on the conservation need of species. Lastly, the post-mined landscape currently is the focal point for a regional effort to restore elk (*Cervus canadensis*) in the Appalachians.

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1 Setting the Context for Terrestrial Wildlife

The Appalachian region covers approximately 530,138 km^2 and extends more than 1,600 km from southwestern New York to northeastern Mississippi encompassing the Blue Ridge, Ridge and Valley, and Appalachian Plateau provinces, along with adjacent portions of the Interior Low Plateau provinces (Boettner et al. 2014). Although the region includes 13 states, the majority of coal recently was or is currently being produced from the 211,372 km^2 Appalachian Plateau portions of Alabama, Tennessee, Kentucky, Virginia, West Virginia, Ohio, and Pennsylvania, save for the Anthracite Region in the Ridge and Valley of northeastern Pennsylvania (Averitt 1975).

Although heavily logged during the nineteenth and early-to-mid-twentieth centuries following European settlement, second- and third-growth forests represent 65% of the current land base (Yarnell 1998); however, some counties in the region have landcover that exceed 90% forested (Boettner et al. 2014). Highly variable elevational and climatic conditions found in the region result in diverse growing conditions and habitats that support a wide range of forest types: montane red spruce (*Picea rubens*)-eastern hemlock (*Tsuga canadensis*) and northern hardwood beech (*Fagus grandifolia*)-birch (*Betula* spp.)-maple (*Acer* spp.)-cherry (*Prunus serotina*) types in the northern and higher elevation portions of the Plateau. Appalachian cove hardwoods and oak forests, dominated by yellow-poplar (*Liriodendron tulipifera*) and a variety of oaks (*Quercus* spp.), respectively, exist throughout the region. The botanically rich mixed-mesophytic hardwood forests occur in eastern Kentucky, southwestern Virginia, and southern West Virginia whereas oak-pine (*Pinus* spp.) types become common in Tennessee and Alabama (Braun 1950). The amount of early-successional lands present prior to European settlement is open for debate, particularly those shaped by Native American use of fire, but invariably these areas occurred throughout the landscape (Harper et al. 2016). For example, early accounts by explorers and surveyors in the mid to late eighteenth century described vast fields of native grasses and clovers (*Trifolium* spp.) in the region (Ford et al. 2003). Accordingly, the region supports tremendous habitat diversity that contributes to the high faunal richness found therein. Pickering et al. (2002) estimated that the Appalachian Region, including the Appalachian Plateau, contained 76 amphibian species, 58 reptile species, 255 bird species, and 78 mammals.

Wildlife communities and their habitats in the Appalachian Plateau have been modified and influenced by humans for thousands of years, beginning with late Pleistocene-early Holocene arrival of Native Americans through European settlement (Van Lear and Harlow 2002). Habitat change, especially the early conversion of river-bottom forests and switch cane (*Arundinaria gigantea*) to agriculture (level to moderately sloping tillable land is uncommon locally) by the early nineteenth century and then industrial logging and early coal mining before the twentieth century, combined with unregulated subsistence hunting profoundly affected wildlife abundance. Similarly, by 1940, the loss of the American chestnut (*Castanea dentata*) from the chestnut blight (*Cryphonectria parasitica*) removed a keystone mast source for regional wildlife (Hepting 1974). American bison (*Bison bison*) were extirpated by the 1820s and the eastern elk (*Cervus canadensis*) by the 1870s. The passenger pigeon (*Ectopistes migratorius*), whose sheer abundance constituted an important component of forest disturbance, was largely absent on the landscape shortly before the twentieth century (Ellsworth and McComb 2003). Predators such as the eastern mountain lion (*Puma concolor cougar*) and eastern wolf (*Canis lupus lycaon*) were gone by the 1890s, persisting longer than bison and elk due to the introduction of free-range livestock production. Species considered common today, following decades of closed hunting seasons, localized re-introduction, and regulated harvest, were typically present only in areas of low human population density. These species included: wild turkey (*Meleagris gallopavo*), beaver (*Castor canadensis canadensis*), white-tailed deer (*Odocoileus virginianus*), river otter (*Lutra canadensis*), and black bear (*Ursus americanus*) (Trefethn 1975). The remote, large post-mined landscape has facilitated the establishment of feral swine (*Sus scrofa*) populations in the region (Gipson et al. 1998). The broad-spectrum ecosystem-level impacts of swine, particularly to native wildlife, vegetation, and water quality, is a concern for managers in the Appalachian Plateau (Campbell and Long 2009).

Other than impacts from locally high human population densities and associated development from initial coal production that provided an impetus for subsistence hunting, wildfire, and logging, deep mining processes historically had few direct terrestrial wildlife impacts per se. However, throughout much of the region, surface mining became a prominent mining method following World War II with the development of larger, earth-moving equipment (Skousen and Zipper 2014) that have substantively altered the Appalachian landscape and impacted its wildlife (Wickham et al. 2013). Larger machinery and increased mechanization allowed large areas to be economically surface mined, with little to no legal requirements for post-mining reclamation (Skousen and Zipper 2014). These unreclaimed sites often contained exposed high walls and large expanses of overburden or spoil. Prior to 1977, if reclamation was performed, it generally included some back filling of excavated areas and the planting of trees, shrubs, or grasses on disturbed areas. These efforts were often only minimally successful, with even hardy volunteer species such as black locust (*Robinia pseudoacacia*) having difficulty establishing in mine soils. Nevertheless, depending on drainage and water quality, borrow pits at the base of highwalls often created permanent standing water in a region with little lentic conditions, creating habitat for Anurans, eastern newts (*Notophthalmus viridescens*), and occasionally shorebirds and waterfowl (Denmon 1998).

Passage of the Surface Mining Control and Reclamation Act (SMCRA) of 1977 established federal control over coal mining, reclamation, and environmental standards (Skousen and Zipper 2014). Among the requirements for a company to receive a mining permit are the following considerations relevant to post-mining wildlife use: (1) regrading (i.e., to approximate original contour) or, by approved variance, an alternate post-mining landscape; (2) establishment of a mine soil suitable for vegetation and selection of appropriate plant species for revegetation; and (3) development of the designated post-mining land use. Federal regulations for meeting these requirements are very explicit and Skousen and Zipper (2014) provide an excellent review of all permit requirements. The SMCRA allows for post-mining land that includes (1) prime farmland, (2) hay land and pasture, (3) biofuel crops, (4) forestry, (5) wildlife habitat, (6) building site development, (7) other suitable uses; and requires that mined lands be returned to a condition that will support their pre-mining land use or higher and better use. What constitutes "higher and better" use is subject to regulatory interpretation, but in economically-depressed Appalachia, economic development often is considered most desirable, followed by managed lands (e.g., farmland, hay, and pasture), and lastly forestry and wildlife habitat. Most mine sites in the Appalachian Region were forested prior to mining but were reclaimed as hay land or pasture (estimated at 80–90%) during the 1980s and 1990s (Skousen and Zipper 2014).

Part of the reason for reclaiming to pastoral use was the initial limited success in reforestation efforts. Implementation of SMCRA regulations inadvertently created physical and environmental conditions (e.g., soil compaction, dense ground cover) that led to poor tree growth on reclaimed mines (Simmons et al. 2008; Burger and Evans 2010). Following decades of reclamation research, the Forestry Reclamation Approach (FRA) was developed as a set of practices to enhance restoration of forest vegetation and ecosystem services (Burger et al. 2005; Zipper et al. 2011b), including: (1) create a suitable rooting medium, (2) loosely graded soil to reduce compaction, (3) use of less competitive ground covers, (4) planting both early successional tree species for wildlife and soil stability, and commercially valuable crop trees, and (5) use of proper tree planting techniques. The goal for adoption of FRA practices is to meet SMCRA guidelines whereby promoting a successional trajectory towards forest vegetation and a faster return of native flora and fauna and their associated ecosystem services (Zipper et al. 2011b).

Nonetheless, surface mining disturbance and reclamation, whether permitted before 1977 or following the SMCRA, has created somewhat non-analog early successional conditions in the Appalachian Plateau that strongly influence wildlife in the post-mined environment. Research on how reclaimed habitats are used by various taxa of wildlife shows varied levels of response: negative, neutral, and positive, depending on the species and landscape/temporal context (Buehler and Percy 2012). Although many of the negative surface mining impacts to forest-associated wildlife have been demonstrated (Wickham et al. 2013; Wood et al 2013), post-mining reclamation can benefit grassland- or early successional shrub-scrub wildlife species as well as generalist species that require a wide diversity of habitats across the large landscape (Zipper et al. 2011a). Of the numerous wildlife species found in the Appalachian Plateau, many rely to some extent on hard mast, particularly that produced by oaks. Within the region, the seasonal availability of oak mast strongly influences the annual survival and recruitment of species ranging from small mammals to white-tailed deer and black bear. Because of the importance of oak mast, the conversion of mature oak forest types to predominantly reclaimed grass-legume and shrub landcover will influence the diversity and abundance of wildlife species occupying the site for potentially decades. Species dependent on forest structure and hard mast, such as tree squirrels, will be absent from the reclaimed site until mature, mast-bearing forest returns. The habitat quality and carrying capacity for other less obligate species will depend on reclamation practices and their success in reaching successional milestones. Early efforts to re-establish oaks and other hard mast-producing species (e.g., black walnut *Juglans nigra*, hickory *Carya* spp.) on reclaimed mine sites were mostly unsuccessful. FRA practices and further research offer optimism that the reestablishment of native mast-bearing species is feasible as part of standard reclamation efforts (Davis et al. 2012). Information on occupancy and use of reclaimed mine sites among taxa varies because of regulatory requirements and recreational and economic development objectives. The amount of information available or lacking for a species group may reflect their regulatory status (Indiana bat *Myotis sodalis*), ecological role as an environmental indicator (amphibians) or as highly adaptable generalist (white-tailed deer). The following is not intended to be exhaustive and focuses mostly on post-SMCRA wildlife status.

2 Herpetofauna

Among all temperate ecosystems, the herpetofauna of the Appalachian region, including the Appalachian Plateau, is characterized by high salamander diversity, particularly the lungless salamanders in the family Plethodontidae (Petranka 1998). Conversely, diversity of other amphibian and reptile groups is low compared to other areas of North America. Anurans are primarily characterized by generalist species within the families Bufonidae, Hylidae, and Ranidae, although some members of these families are more specialized, such as wood frogs (*Lithobates sylvatica*) and mountain chorus frogs (*Pseudacris brachyphona*). Snake species in the region represent a broad array of ecological strategies, although most species are often more abundant in either open areas or forest patches containing suitable basking habitat (i.e., canopy gaps). These species span common generalists such as eastern garter snakes (*Thamnophis sirtalis*) and racers (*Coluber constrictor*), semi-aquatic species such as northern water snakes (*Nerodia sipedon*), and species with more restricted resource needs such as timber rattlesnakes (*Crotalus horridus*). There are also a number of small snakes typically found in forests and edges such as eastern worm snakes (*Carphophis amoenus*) and smooth earthsnakes (*Virginia valeriae*). The turtle species in the region are primarily semi-aquatic, with the exception of the eastern box turtle (*Carolina terrapene*). Finally, lizards are the least diverse of Appalachian taxa and, such as the eastern fence lizard (*Sceloporus undulatus*), generally are associated with edge habitat between forest and open areas. There are three major types of mining impacts that have been investigated with respect to amphibians and reptiles: loss of forest cover and the frequent conversion to grassland or shrubland; creation of ponds or wetlands post-mining; and stream occupancy particularly in valley fill areas following mountaintop removal mining.

2.1 Salamanders

The majority of Appalachian salamanders rely on forests for some aspects of their life history. As a result, minelands reclaimed as grasslands may not be suitable for salamander persistence and/or serve as barriers to a movement among forest patches (Fig. 1). For instance, spotted salamanders (*Ambystoma maculatum*), a widespread Appalachian species, will not move across grassland patches (Rittenhouse and Semlitsch 2006). In southwest Virginia, Carrozzino (2009) found only one salamander detection in reclaimed minelands as compared to 64 observations in either reference or pre-SMCRA mineland forest. Eastern newts successfully reproduced in wetlands at an abandoned contour mine in West Virginia, but there was no evidence of spotted salamander recruitment, which was attributed to reduced water quality (Loughman 2005). Terrestrial salamanders were primarily limited to intact forest or forest fragments around reclaimed mountaintop removal mines in West Virginia, and habitat type was the main factor affecting salamander occupancy (Wood and Williams 2013a, Williams et al. 2017). Similar patterns of low salamander richness and abundance on reclaimed mine sites also have been noted in the Illinois Basin (Lannoo et al. 2009; Terrell et al. 2014; Stiles et al. 2016). In West Virginia, wood frogs only were found in forested areas on a reclaimed mine (Williams et al. 2017) and were absent from an abandoned contour mine (Loughman 2005). However, three species of wetland breeding salamander (e.g., marbled salamander *Ambystoma opacum;* small-mouthed salamander *Ambystoma texanum;* and eastern newt) have successfully recruited in wetlands embedded in reclaimed mines adjacent to unmined forests.

Fig. 1 Slimy salamander (*Plethodon glutinosus*) found in a post-mined landscape. Woodland salamanders in the family Plethodontidae require forested, cool, and moist environments. Surface mining negatively impacts these species in the Appalachian Plateau. Given time, vegetation in-growth, and immigration from surrounding forest habitats, some recovery can occur. (Photo by S.F. Spears)

Unlike post-SMCRA sites, many pre-SMCRA minelands in the Appalachian Plateau reforested to some extent and likely provide a greater opportunity for salamander recolonization than more recently reclaimed areas. Brady (2016) compared pre-SMCRA sites to reference forests in eastern Ohio and found a higher species richness of salamanders in reference sites. However, this difference was due to the complete absence of stream breeding salamanders (e.g., northern dusky salamander *Desmognathus fuscus*), northern two-lined salamander *Eurycea bislineata,* long-tailed salamander *E. longicauda,* and red salamander *Pseudotriton ruber*) from pre-SMCRA sites. Other studies have also found stream-dependent salamanders responding negatively to mining (Merovich et al. 2021).

When only comparing strictly terrestrial salamanders, such as the northern ravine salamander (*Plethodon electromorphus*), eastern red-backed salamander (*Plethodon cinereus*), and northern slimy salamander (*Plethodon glutinosus*), there was no difference in abundance between pre-SMCRA sites and reference forests. Many pre-SMCRA surface mines are also characterized by rock highwalls that could potentially provide habitat for crevice dwelling species. These rock highwalls also typically have to borrow pits with water that likely provide habitat for several pond-breeding amphibian species. With respect to the potential for highwall rock crevices to provide habitat, Hinkle et al. (2018) examined the distribution of the outcrop associated green salamander (*Aneides aeneus*) across sites in three categories in Virginia: highwalls created by surface mining, remnant forested outcrops within a surface mine matrix, and unmined outcrop sites within a national forest. Although they did not find any salamanders on highwalls, they did find presence at 72% of remnant outcrops within mine sites. Furthermore, they found evidence of reproduction at both remnant and unmined outcrops. Highwalls were characterized by fewer crevices within an outcrop and lower forest cover but there were no differences between remnant and unmined outcrops in these categories. As a result, while mining activities likely exclude this species directly, green salamanders appear to persist in a mining landscape if remnant outcrops are preserved.

2.2 Reptiles

Terrestrial reptiles are often found on grasslands of reclaimed mines, but little data exist beyond observations. Snakes tend to occur primarily in the shrubland of post-SMCRA mines, although they do not respond strongly to habitat type (Williams et al. 2017). Species such as racers can be abundant in reclaimed minelands (Myers and Klimstra 1963; Williams et al. 2017), especially in rocky highwall areas (Loughman 2005). The federally threatened eastern massasauga (*Sistrurus catenatus*) has been observed on reclaimed minelands in Pennsylvania, though not believed to be common there (Brenner 2007). Results from research on box turtle use and abundance on reclaimed minelands versus surrounding forests were inconclusive in the Appalachian Plateau as well as the Illinois Basin (Lannoo et al. 2009; Stiles et al. 2016), whereas eastern fence lizards were common (Myers and Klimstra 1963).

2.3 Relationships to Aquatic Habitats

The abundance and heterogeneity of wetlands created from surface mining likely have benefited species such as frogs adapted to breed in small wetlands with a variety of hydroperiods (Stiles et al. 2016). In general, frogs common to the Appalachian Plateau tend to occur on reclaimed minelands when water is present (Lacki et al. 1992; Timm and Meretsky 2004; Turner and Fowler 1981; Loughman 2005; Carrozzino 2009; Lannoo et al. 2009; Terrell et al. 2014; Stiles et al. 2016; Williams et al. 2017). Reptile response is poorly studied in the Appalachian Plateau, but in the Illinois Basin, northern water snakes and painted turtles (*Chrysemys picta*) were common on pre-SMCRA mines (Myers and Klimstra 1963) and post-SMCRA sites (Lannoo et al. 2009; Stiles et al. 2016). Copperbelly water snakes (*Nerodia erythrogaster neglecta*) were observed over a wide array of post-SMCRA conditions with no apparent population impacts (Lacki et al. 2005).

Gore (1983) found the presence of northern dusky salamanders in 16 of 78 streams in surface-mined areas in eastern Kentucky and suggested that increased dissolved solids and lack of shading were reducing populations. Lower occupancy, abundance, and rates of colonization and persistence were observed for a community of stream amphibians (e.g., northern dusky salamander, seal salamander *Desmognathus monticola*, southern two-lined salamanders *Eurycea cirrigera*, spring salamanders *Gyrinophilus porphyriticus*, and red salamanders) in streams within or downstream of reclaimed mountain top removal mines versus streams in unmined second-growth forest reference sites in Kentucky (Muncy et al. 2014; Price et al. 2016; Price et al. 2018). The streams through the reclaimed mine sites were characterized by increased conductivity, fewer in-stream cover rocks, and less surrounding canopy cover, which was correlated with the reduced presence and abundance of salamander species. Bourne (2015) also detected increased selenium levels in salamander tail tips from valley-fill streams relative to those from reference streams. In eastern Tennessee, stream salamander numbers were negatively affected by low pH and high conductivity in streams draining minelands (Schorr et al. 2013). Comparing two reference streams with three valley-fill streams in West Virginia, Hamilton (2002) found reduced salamander abundance at two of the three valley-fill streams, although the oldest (18 years since reclamation) had similar abundances to reference sites, indicating the potential for recovery post-mining. Williams and Wood (2004) noted high rock density and abundant cover objects mitigated some of the impacts to stream salamanders in post-mined valley-fill streams. Further work by Wood and Williams (2013b) found similar salamander species richness between reference and valley-fill streams, but approximately double the number of salamander individuals was observed in reference streams relative to valley fill. Sweeten and Ford (2016)

Fig. 2 Post-mined landscape restored stream. Aquatic and semi-aquatic salamander species in the genus *Desmognanthus* may require > 20 years post-mining and restoration to approximate community composition and population numbers in pre-mined or reference streams in the Appalachian Plateau (Photo by S. Sweeten, Virginia Tech)

assessed occupancy and abundance of stream salamanders with stream habitat and landscape-level covariates in southwest Virginia and found that both occupancy and abundance of *Desmognathus* spp. was best explained by site variables such as stream canopy cover (Fig. 2). In contrast, occupancy of *Eurycea* spp. was related to overall percent mining and forest loss across the watershed but abundance was explained by stream sediments and embeddedness. Based on these results, they recommended that post-mining reforestation may especially benefit *Desmognathus* spp. due to their reduced dispersal compared to *Eurycea* spp. (Sweeten and Ford 2015). Accordingly, successful restoration of mined lands with streams using FRA practices would likely benefit the entire stream salamander community, as species tend to have similar ecological responses to mine disturbance (Price et al. 2018). Although a reduction in stream salamander abundance is seen in forests with timber harvest, the magnitude of this effect tends to be smaller compared to studies from minelands, and stand age is not always the strongest covariate influencing abundance (Moseley et al. 2008).

There are many gaps in knowledge regarding the influence of surface mining on herpetofauna species. Comparative studies examining the abundance and population trends on mined and unmined lands are generally needed for all reptile groups and pond-breeding amphibians in general. Terrestrial salamanders of the genus *Plethodon* can occur in reclaimed forests, but the full suite of factors that influence the ability of this group to recolonize and persist post-mining largely is unknown. Lastly, although stream salamander response is well documented, further work to better understand recovery times and optimal restoration practices to increase populations would be useful.

3 Avifauna

Researchers have long recognized that unprecedented landscape-level changes created by surface mines and resulting reclamation efforts affect avian communities (Riley 1952). Forest bird assemblages are initially altered by the complete clearing of forests for mining, but then effects can extend post-mining if reclamation does not involve reforestation. Clearing of forested areas changes the forest bird community into a grassland-early successional bird community. When large areas of forest are removed it also creates forest edges for the remaining forested areas which affect bird species reliant on core forest areas. However, reclaimed minelands, especially those with residual or created water features and wetlands can add to local avian diversity. For example, over 130 species of birds have been recorded using a large reclaimed mine site in southeastern Kentucky, including waterfowl and wading shorebirds such as the American avocet (*Recurvirosta americana*) that are otherwise rare to absent in the Appalachian Plateau (Cornell Lab of Ornithology 2020).

3.1 Forest Obligate Birds

Generally, forest songbirds are sensitive to changes resulting from surface mining and reclamation. Forest bird species occupancy and abundance decline following tree removal and mining, and core-forest bird species are affected disproportionately. For example, from 1992–2006 in 19 counties in the central portion of the Appalachian Plateau, ~92,500 ha of mature forest was changed to non-forest cover (McDermott et al. 2013). Becker et al. (2015) found that in Kentucky and West Virginia, a landcover transition away from forest after mining activities elicited more negative responses than positive ones for birds and that negative effects on avian species abundance occurred at thresholds lower than (or before) other species responded positively. Specifically, the forest interior guild was most sensitive to landscape changes wherein abundance negatively responded to even small (11%) landscape losses in forest cover (Becker et al. 2015).

Research that focuses on the effects of surface mine reclamation on forest songbirds is limited and usually species-specific (Becker et al. 2015). Cerulean warblers (*Setophaga cerulea*) avoid large-scale disturbances and edges such as large open fields and power lines (McDermott et al. 2013). In West Virginia, there were no clear differences in red-eyed vireo (*Vireo olivaceous*) nest success between forests on reclaimed minelands and non-mined areas (Mizel 2011). However, tree species composition contributed to guild differences, especially on heavily compacted mine benches where American redstart (*Setophaga ruticilla*), rose-breasted grosbeak (*Pheucticus ludovicianus*), and worm-eating warbler (*Helmitheros vermivorum*) were using subcanopy dominated by yellow-poplar for foraging (Mizel 2011). Also, black-and-white warbler (*Mniotilta varia*), ovenbird (*Seiurus aurocapilla*), and worm-eating warbler avoided maple stands because of reduced forest floor leaf litter as compared to oak-hickory (*Carya* sp.) dominant stands (Mizel 2011).

In the Appalachian Plateau, some advocate for minelands to be reclaimed as forests because the contiguous forest is critical for core forest songbirds (McDermott et al. 2013; Wood et al. 2013; Becker et al. 2015; Wood and Ammer 2015). The FRA advocates for science guided reforestation of minelands that outlines an explicit approach dependent on region-specific target species and goals (Wood et al. 2013). Highlighted is the potential for reforested areas to initially provide young-forest habitat for ruffed grouse (*Bonasa umbellus*), golden-winged warbler (*Vermivora chrysoptera*), and American woodcock (*Scolopax minor*; Wood et al. 2013). In West Virginia, American woodcock use reclaimed minelands, which may be the best available habitat in some landscapes, though soil conditions and earthworm biomass is best on reclaimed mines > 5 yr old (Gregg et al. 2001). Also in West Virginia, ruffed grouse utilized reclaimed surface mines, but non-mined areas had greater plant-feeding rates and provided better food resources (Kimmel and Samuel 1984). Overall, there remains a significant lack of information regarding forest reclamation effects on forest birds, in part because it requires decades, at minimum, for forests on reclaimed mines to adequately regenerate.

3.2 Grassland and Early-Successional Birds

In the Appalachian Plateau, reclaimed minelands are dominated by non-native invasive grass and herbaceous vegetation and offer a unique heavily impacted cover type that would otherwise not occur (Stauffer et al. 2011; Sena et al. 2021). Despite the clear disparity in plant species composition and habitat structure between reclaimed mines and surrounding Appalachian Plateau forests, reclaimed minelands provide unique areas for an assemblage of otherwise absent grassland and shrubland bird species (Graves et al. 2010; Ingold et al. 2010; Fig. 3). Reclaimed minelands generally provide adequate habitat for grassland-dependent birds (DeVault et al. 2002), and in limited species-specific examples, do not act as population sinks. Though largely from work in the Illinois Basin, red-winged blackbird (*Agelaius phoeniceus*), grasshopper sparrow (*Ammodramus savannarum*), and eastern meadowlark (*Sturnella magna*) occurred on >95% of reclaimed mineland survey points. Dickcissel (*Spiza americana*; 67%), common yellowthroat (*Geothlypis trichas*; 65%) occurred less frequently, and Henslow's sparrow (*Centronyx henslowii;* 59%) were patchily distributed (Scott et al. 2002). Landscape factors generally did not affect Henslow's sparrow abundance on reclaimed minelands, and Henslow's sparrows were virtually absent from pasture and hayfields in areas around reclaimed mines (Bajema and Lima 2001). Patch-level relationships showed red-winged blackbird abundance was negatively correlated with litter and canopy cover, whereas eastern meadowlark and grasshopper sparrow abundance was negatively correlated with visual obstruction preferring less dense vegetation (Scott et al. 2002). Henslow's sparrows preferred areas dominated by grasses, specifically smooth brome (*Bromus inermis*) and broom

Fig. 3 Common yellowthroat (*Geothlypis trichas*) is an example of an early-successional shrub-scrub obligate bird species that can respond favorably to the post-mined landscape in the Appalachian Plateau (Photo by J. Cox, University of Kentucky)

sedge bluestem (*Andropogon virginicus*). In Tennessee, golden-winged warbler, a critically declining shrubland bird, nest on reclaimed minelands (Bulluck and Buehler 2008). In southeastern Ohio, populations of grassland birds (grasshopper sparrows and Henslow's sparrows) are increasing, whereas in other parts of the state they are decreasing (Ingold 2002). In the Appalachian Plateau, population estimates confirm that reclaimed minelands provide critical habitat in maintaining population densities for grassland birds, specifically Henslow's sparrow (Mattice et al. 2005). In Pennsylvania, the distribution of Henslow's sparrows closely matches the distribution of reclaimed surface mines (Hill 2012). Moreover, there was no support that landscape characteristics affected Henslow's sparrow occupancy; rather local vegetation structure was the most influential factor (Hill and Diefenbach 2014). In fact, typically there are species-specific responses to vegetation structure that are comparable to non-reclaimed mine sites.

Species abundance and nest success are influenced by vegetation structure and disturbances on reclaimed mines, analogous to patterns seen on non-reclaimed mine habitats. Grasshopper sparrow, Henslow's sparrow, savannah sparrow (*Passerculus sandwichensis*), and bobolink (*Dolichonyx oryzivorus*) abundance was negatively correlated with increasing shrub cover on reclaimed mines in southeastern Ohio. Alternatively, eastern meadowlark and dickcissel abundances were not correlated to shrub cover (Graves et al. 2010). In West Virginia, grasshopper sparrows on reclaimed minelands require grass structure similar to that found in native prairie, with ~25% bare ground in their territories (Whitmore 1979; Wood and Ammer 2015). In Pennsylvania, Henslow's sparrow abundance on reclaimed minelands was positively related to increasing grass cover, but the inverse was true for herbaceous cover (Hill and Diefenbach 2014).

A fundamental question for reclaimed minelands is if they act as ecological traps by attracting nesting birds that ultimately are not reproductively successful (Wray et al. 1982). For golden-winged warbler on reclaimed mines, nest sites tended to have more grass and forb cover and less woody cover than random sites, and no micro-habitat variables investigated were significantly correlated with daily nest survival (Bulluck and Buehler 2008). Reclaimed minelands in Pennsylvania provide brood habitat and food resources for imprinted wild turkey poults when compared to unmined areas (Anderson 1980). In southeastern Ohio, random locations had ~2.5 times the amount of woody vegetation, but nest placement was not associated with the number of woody patches or distance to an edge (Graves et al. 2010). Grasshopper sparrow and Henslow's sparrow daily nest survival was negatively correlated with the amount of woody vegetation surrounding nests (Graves et al. 2010). Henslow's sparrow nest survival on reclaimed minelands that are small in size (<100 ha) was comparable to other values reported in the literature; and in Pennsylvania, Henslow's sparrow nest almost always occurred on reclaimed minelands (Stauffer et al. 2011). However, at these sites, Henslow's sparrow produced fewer young, as increasing fledgling production was correlated with decreasing woody vegetation, decreasing plot-level woody vegetation, and decreasing bare ground around the nest (Hill 2012). Another consistent finding is that nesting songbirds on reclaimed minelands have low brood parasitism rates by brown-headed cowbirds (*Moluthrus ater*; Wray et al. 1982; DeVault et al. 2002; Scott and Lima 2004; Hill 2012; Wood and Ammer 2015).

Woody vegetation removal, mowing, prescribed burning (Fig. 4), and grazing are some disturbance tools recommended to improve grassland and shrubland bird habitat on reclaimed sites (Whitmore 1981; Ingold et al. 2009; Hill and Diefenbach 2014; Brooke et al. 2015). Woody vegetation removal on reclaimed minelands correlated with a three-fold decline in Henslow's sparrow survival, but grasshopper sparrow population increased by 15% on woody removal treatments (Hill and Diefenbach 2014). On reclaimed sites in Ohio, Henslow's sparrows and bobolink avoided recently mowed areas for nesting, but savannah sparrow and grasshopper sparrow nested equally on mowed and unmowed areas (Ingold 2002). Site fidelity of Henslow's sparrows was low (~13%) and individuals that returned were only individuals that were banded in unmowed areas (Ingold et al. 2009); no individuals banded

Fig. 4 Prescribed fire being used to maintain and improve grassland and shrub habitat conditions for northern bobwhite (*Colinus virginianus*), white-tailed deer (*Odocoileus virginianus*), and elk (*Cervus canadensis*) in the post-mined landscape in the Appalachian Plateau (Photo by D. Ledford, Appalachian Wildlife Foundation)

in mowed areas returned. Average survival probabilities of returning Henslow's sparrows were greater on mowed than unmowed areas (Ingold et al. 2010) and mowing did not affect grasshopper sparrow or savannah sparrow returns. Prescribed burning reduces litter and encourages pioneer plants, positively affecting northern bobwhite (*Colinus virginianus*) brood habitat (Brown 1981), though prescribed burning is believed to have negatively affected winter survival by reducing the quality of shrub cover in some mined sites (Peters et al. 2015). Northern bobwhite on reclaimed minelands is most limited by the wide interspersion of shrub cover (Brooke et al. 2015).

In the absence of reclamation that focuses on reforestation, grassland and shrubland cover is created and can provide habitat for a community of grassland and shrubland birds that continues to experience dramatic population declines in the eastern United States. Many grassland and shrub bird species not only occupy reclaimed minelands but occur in abundances comparable to un-mined areas. Additionally, these species can nest and reproduce at rates comparable to unmined areas, suggesting that reclaimed minelands are usually not acting as ecological traps. Specifically, for northern bobwhite, reclaimed mines managed or otherwise offer novel opportunities for management directed at maintaining populations that are otherwise lacking in most of the Appalachian Plateau (Brown 1981; Wood et al. 2013; Brooke et al. 2015; Peters et al. 2015).

3.3 Raptors

Post-SMCRA mines can also provide habitat for a variety of raptor species. In northwestern Pennsylvania, reclaimed mines had greater counts of spring raptors than surrounding agricultural areas (Yahner and Rohrbaugh 1998), and the most frequently counted species were red-tailed hawk (*Buteo jamaicensis*; 58%), American kestrel (*Falco sparverius*; 36%), and northern harrier (*Circus cyaneus*; 6%). Red-shouldered hawks (*Buteo lineatus*) tolerated fragmentation created by reclaimed minelands in West Virginia, but were still more likely to occur close to wetland areas (Balcerzak and Wood 2003). On reclaimed mines, these areas were often represented by fill ponds created to control mine erosion. Short-eared owls (*Asio flammeus*) and northern harriers (*Circus hudsonius*) have been documented to nest on reclaimed sites in the Appalachian Plateau, and northern harriers tended to nest in drier areas with denser vegetation with a greater proportion of subadult females comprising the nesting population than expected (Vukovich and Ritchison 2006, 2008).

4 Mammals

4.1 Bats

Bats comprise an ecologically important faunal component with at least 16 species potentially occurring in all or portions of the Appalachian Plateau Coalfields. During spring through fall, outside of hibernation and migratory seasons, all regionally extant species forage in both forested and open upland areas and along all orders of riparian corridors. Most species in the region day-roost in foliage, exfoliating bark, or cavities of live trees or snags including Rafinesque's big-eared bat (*Corynorhinus rafinesquii*), big brown bat (*Eptesicus fuscus*), eastern red bat (*Lasiurus borealis*), hoary bat (*Lasiurus cinereus*), Seminole bat (*Lasiurus seminolus*), silver-haired bat (*Lasionycteris noctivagans*), little brown bat (*Myotis lucifugus*), the threatened northern long-eared bat (*Myotis septentrionalis*), the endangered Indiana bat, evening bat (*Nycticeius humeralis*), and tri-colored bat (*Perimyotis subflavus*). Exceptions to the use of trees as day-roosts are the cave-obligate and endangered Virginia big-eared bat (*Corynorhinus townsendii virginianus*) and endangered gray bat (*Myotis grisescens*). Rafinesque's big-eared bat, big brown bat, little brown bat, and Brazilian free-tailed bat (*Tadarida brasiliensis*) will commonly use anthropogenic structures that mimic hollow trees or snags as day-roosts (Ellison et al. 2007; Fagan et al. 2018). Additionally, Indiana bats and northern long-eared bats will readily use artificial roost-structures designed specifically for bats (Adams et al. 2015; De La Cruz et al. 2018; Hoeh et al. 2018). The eastern small-footed bat (*Myotis leibii*) is the exception in that it day-roosts primarily in emergent rock (i.e., cliff-faces) and talus slopes (Moosman et al. 2015). Many of the cave- or mine-hibernating species (i.e., eastern small-footed bat, little brown bat, northern long-eared bat, tri-colored bat, and Indiana bat) have suffered precipitous declines approaching or exceeding 90% due to White Nose Syndrome (WNS), caused by the novel fungal pathogen *Pseudogymnoascus destructans* (Francl et al. 2012; Powers et al. 2015). Another additive stressor to non-hibernating bats, such as the eastern red bat and hoary bat, in the region has come from additive mortality associated with wind-energy development in Pennsylvania and West Virginia, some occurring on reclaimed surface mines (Arnett et al. 2008).

4.2 Bat Day-Roosting

Surface-mining deforestation removes both day-roosting and foraging bat habitat. Although diurnal roosts are likely not a limiting factor in this largely forested region, the size (i.e., 25–2000 ha) of most surface mines in the Appalachian Plateau increases the likelihood that roosting networks used by active maternity colonies of the social northern long-eared and/or the Indiana bat are altered or destroyed (Menzel et al. 2001; Silvis et al. 2015, 2016). Throughout the 1970s to present, prior to mining, operators are required to survey for Indiana bats and locate day-roosts to determine minimization and mitigation measures (e.g., seasonal tree-clearing) for continued species conservation (US FWS 2020). Migratory and foliage-roosting bats such as the eastern red bat, hoary bat, and Seminole bat do not form large colonies and presumably are not overly impacted by pre-mining tree removal (Menzel et al. 2000). Despite the occurrence of northern long-eared bats and Indiana bats in the Appalachian Plateau, particularly prior to WNS, little or no research has examined post-mining day-roost use in forests surrounding mined lands or implications of colony formation within mined areas themselves. Presumably from research conducted in managed and unmanaged forest landscapes, some pre-SMCRA woody regrowth, particularly cavity bearing black locust trees or snags, may provide quality northern long-eared day-roost habitat (Johnson et al. 2009; Silvis et al. 2016).

Application of the FRA holds promise for improved day-roost habitat for bats as trees mature, particularly species that exhibit exfoliating bark or are prone to cavity formation. Wildlife-friendly tree species provided in FRA guidelines (Wood et al. 2013) such as sugar maple (*Acer saccharum*), shagbark hickory (*Carya ovata*), and American elm (*Ulmus americana*) are preferred by Indiana bats (Menzel et al. 2001; Johnson et al. 2010; Jachowski et al. 2016), whereas black locust and sassafras (*Sassafras albidum*) is preferred by northern long-eared bats in the region (Silvis et al. 2016). Presumably, the lower tree densities and higher snag creation rates than surrounding forests and increased solar radiation will improve future day-roosting conditions (Johnson et al. 2009; Ford et al. 2016). In the interim, young forests on reclamation sites provide bat above-canopy foraging substrate and a source of arthropod prey (Sheets et al. 2013). Similarly, if strategically placed during planting (Wood et al. 2013), these young forest patches create connective corridors between the unmined forest surrounding the mine site as Indiana bats and other forest-adapted foraging species are less likely to traverse open landscapes (Menzel et al. 2005). Additions of artificial day-roost structures which have been used in the region by northern long-eared bats where natural day-roost loss has occurred due to deforestation (De La Cruz et al. 2018) could be used before FRA-planted stands achieve sufficient structure to serve as day-roost habitat. Emergent rock features destroyed by surface mining have impacted eastern small-footed bats, however, this species has been found to readily use crevices in residual highwalls and waste rock, and also will utilized “engineered” rock structures whether designed expressly for the species (Tompkins 2014) or associated with other activities such as rock placement for stream restoration or boulder-lined dam surfaces (Moosman et al. 2013).

4.3 Bat Foraging

Post-reclamation foraging activity by bats in the region also is poorly studied. Research in the Appalachian region generally shows neutral to positive responses from bats following forest harvesting or prescribed burning that reduces forest clutter, particularly among larger-bodied species with lower echolocation call characteristics adapted to forage in more open conditions (Ford et al. 2005; Austin et al. 2018).

Whether these findings are directly transferable to reclaimed minelands is unknown considering the differences in disturbance size between forest management activities (i.e., < 30 ha) versus surface mining, or between natural forest regeneration versus either grass/forb reclamation or FRA-planting practices. Whether in a closed forest or open conditions, overall bat foraging activity is related to proximity to water in the Appalachian Plateau (Ford et al. 2005). Assuming good water quality, creation and maintenance of wildlife-friendly water retention areas (Wood et al. 2013) will benefit bats by providing a source for drinking water and potential foraging areas that mitigate for loss of streams and stream function post-mining. Waterhole creation on xeric ridgetops distant from lentic or lotic habitats to benefit bats has been demonstrated in the Appalachian Plateau (Huie 2002; Maslonek 2010; Johnson et al. 2010; 2013). Research regarding bat response to stream reclamation and restoration is limited (Ciechanowski et al. 2011); however, analogous to the FRA process, as vegetation matures and stream processes stabilize, bats will likely utilize these redeveloping foraging and commuting habitats.

4.4 Bat Hibernacula

As a somewhat positive legacy of coal mining in the Appalachian Plateau, in areas that lack limestone solution caves in karst geology, abandoned underground mines and associated structures such as vent shafts often serve as maternity and bachelor roosts, migratory stopover sites, and hibernacula for bats where prior to mining few had existed before (Krusac and Mighton 2002 Johnson et al. 2005; 2006; Buehler and Percy 2012; Furey and Racey 2015; Fig. 5). In the Appalachians generally, the functional extirpation of bat species such as the northern long-eared bat occurred rapidly with the fungal infection of caves, particularly in settings where long hibernation periods occurred (Johnson et al. 2013; Ford et al. 2016; Austin et al. 2018).

Fig. 5 Legacy deep mines have provided day-roosting and hibernation habitat for bats such as these endangered Indiana bats (*Myotis sodalis*) in the Appalachian Plateau (Photo by Tomas Nocera, Virginia Tech)

However, in the Appalachian Plateau of West Virginia and eastern Ohio, anecdotal evidence suggests some residual populations of bats normally affected by WNS persist perhaps due to lower exposure to WNS in pre-law mines than would occur in karst formation caves. Analogous to persistence being reported on the Coastal Plain to the east where bats overwinter in forests or aberrant hibernacula (Grider et al. 2016; Dowling and O'Dell 2018), the same is observed for northern long-eared and Indiana bats overwintering in small legacy coal adits or emergent rock features on pre-law mines in the Appalachian Plateau (De La Cruz and Schroder 2015). These legacy mines may minimize other bat species' exposure to WNS-vectoring little brown bats, may be utilized only by local bats, and/or provide unsuitable substrates (i.e., substrate pH < 5) for *Pseudogymnoascus destructans,* persistence and growth (Wilder et al. 2011; Raudabaugh and Miller 2013). Underground mines often have microclimates suitable for bat hibernacula or if not, maybe modified in a number of ways to increase habitat suitability including venting, entrance and interior stabilization and modification, gating, and installation of monitoring devices (Carter et al. 2010). Exclusion structures such as gates may help to stabilize the entrance of an underground mine but more importantly will protect remaining bats, that may be somewhat WNS resistant, from unnecessary and costly human-induced arousal (Kunz et al. 2010). Impacts to non-bat cave-obligate biota, including native fungi, have limited the use of antifungal treatments of karst caves, but because mines often lack such species the potential treatment of these areas may enhance use by WNS-susceptible bats and subsequently their use of post-mined landscapes (Carter et al. 2010; Sewall et al. 2016).

4.5 Small and Meso-Mammals

Pre- and post-SMCRA mine restoration that included replanting with a grass-legume mixture created sites that lacked vegetational and structural diversity (McGowan and Bookhout 1986), often dominated by grasses or combination of exotic species (e.g., fescue; sericea lespedeza, *Lespedeza cuneata*; Zipper et al. 2011a, Sena et al. 2021). The persistence of this vegetation can have a decadal impact on successional progression (Holl 2002; Skousen et al. 2006). Small mammal diversity on such sites is low (Lacki et al. 1982; McGowan and Bookhout 1986; Larkin et al. 2008) and the lack of woody cover is a limiting factor in the presence of woodland mice (*Peromyscus* spp.; Sly 1976). Whereas results from these early studies provide information on species richness and relative abundance, their objective was not to compare among reclamation practices or between reclaimed habitat and pre-mining conditions. Chamblin (2002) compared mammal species richness among four treatments (intact forest- pre-mining; grassland—7–21 years post reclamation; shrub/pole—18–28 years; fragmented forest– streamside buffer surrounded by reclaimed lands) and reported no differences in species richness among treatments and no correlation between time since reclamation and species richness on mountaintop-removal mines in southern West Virginia. Use of reclaimed mine sites by fossorial (ground-dwelling) and semi-fossorial mammals depends on depth of soil and degree of substrate compaction

(Lawer et al. 2019). Species requiring exposed rock outcrops such as the Allegheny woodrat (*Neotoma magister*) may decline or be absent from post-SMCRA minelands unless an artificial structure can serve as an alternative to native habitat (Chamblin et al. 2004). Meso-mammals either captured or observed on reclaimed sites include Virginia opossum (*Didelphis virginiana*), eastern cottontail rabbit (*Sylvilagus floridanus*), woodchuck (*Marmota monax*), raccoon (*Procyon lotor*), striped skunk (*Mephitis mephitis*), gray fox (*Urocyon cinereoargenteus*), and red fox (*Vulpes vulpes*) (Yearsley and Samuel 1980; Brenner et al. 1982; Lacki et al. 1982). Species with large home ranges with wide habitat tolerances, such as bobcat (*Lynx rufus*) and coyote (*Canis latrans*) would be expected to opportunistically use portions of reclaimed minelands to meet their habitat requirements, if available.

4.6 White-Tailed Deer and Black Bear

White-tailed deer are found throughout coal-bearing areas of the Appalachian Plateau, from high-elevation forests to bottomland agriculture/urbanized areas, though often at lower densities than elsewhere in the Appalachian region (Kniowski and Ford 2017; but see Campbell et al. 2005). As highly adaptable herbivores, they occupy all suitable habitats including those created during mining and reclamation. However, limited research has examined white-tailed deer use of mined lands (Brenner et al. 1975; Brenner et al. 1977; Knotts and Samuel 1982). White-tailed deer herbivory impacts on tree restoration have been noted (Jacobs et al. 2004; Burney and Jacobs 2018). It should be assumed that white-tailed deer occupied forested lands cleared during mining activities and that they returned to these sites post-mining, although research is needed on changes to carrying capacity, whether positive or negative, resulting from habitat alterations.

Reclaimed surface mines, particularly mountain-top removal sites that encompass thousands of hectares, can provide habitat for black bears within Appalachia, particularly along ecotones where soft mast-producing plants (e.g., *Rubus* spp.) thrive. Oak mast is a primary food source for black bears in the Appalachians, and consequently its availability influences spatial and temporal habitat use by bears (Vaughan 2002), including the relative value of interspersed minelands within an otherwise forested landscape. Because of their large scale and conversion of mature hardwood forest to grasslands, however, reclaimed minelands often contain lower habitat quality for black bears because of reduced mast production (Ryan 2009), thereby representing a net loss in overall habitat. Mineland areas with dense cover by the soft-mast-producing exotic autumn olive (*Elaeagnus umbellata*; as per Oliphant et al. 2017) are readily used by black bears (Allan and Steiner 1972). Regionally, black bears have been frequently observed to den in brushpiles, clearcuts, and even under discarded or idle machinery on or near minelands (J. Cox, University of Kentucky, pers. observation). Bears may also have access to anthropogenic food (e.g., large dumpsters) on operational mines, and limited public access to mines can create refugia from harvest because of reduced hunter access (Ryan 2009); however, these same attractants can condition bears to human foods and lead to poaching or increased risk-taking behavior (e.g. frequently crossing roads) whereas other bears seem to totally avoid mines where disturbance is high.

4.7 Elk

Although early accounts from European settlers indicate the presence of elk throughout the Appalachian Plateau, elk were not found at high densities in more forested areas. Elk most often occurred in more open environments in mixed-species aggregations that included bison and white-tailed deer (Bryant and Maser 1982). Early European settlers, and Native Americans before them, did not primarily rely on elk for food and hides, possibly indicating their relatively low abundance compared to other big game species in the region (McCabe 1982). Nonetheless, in the early twentieth century, a few eastern states, including some in the Appalachian Plateau such as Pennsylvania and Virginia, attempted to reestablish elk, though usually with limited success. Regionally, human development and agriculture had displaced formerly suitable elk habitat on floodplains. Accordingly, in the 1990s, managers turned to reclaimed surface coal mines with mosaics of herbaceous/grass and shrub-scrub habitat on mine benches and surrounding forests as potential analogs to occupied elk habitat in the West. Elk are mixed feeders, spending most of their time in open areas grazing grasses, forbs, and other low-growing herbaceous plants; however, they also spend considerable time within closed-canopy forests and shrub-scrub ecotonal communities that provide browse foods, as well as thermal and security cover. On the Appalachian Plateau, surface mines are often located in relatively remote areas away from human development, and therefore, it was suggested that elk placed in these areas would minimize human-elk conflict. Additionally, it was believed that the vast, grassy "minescapes" would enhance release site fidelity and large herd cohesion, thereby decreasing the possibility of multiple founder effects resulting from scattered, small, low-density populations that overtime be non-viable (Maehr et al. 1999).

Over the past 25 years, elk have been reintroduced to portions of Kentucky, Tennessee, Virginia, and West Virginia within the Appalachian Plateau, with most release sites occurring at post-SMCRA reclaimed surface mines (Fig. 6). The reintroduction of elk into Kentucky provides perhaps the most well-studied example of the relationship between elk and this novel landscape. From 1997 to 2002, ~1550 elk were translocated from western states to a 14-county area "elk zone" in southeastern Kentucky. Elk was released at eight different sites, seven of which were on coal surface mines (some active) with varying levels of ongoing human activity (i.e., off-road vehicle use, blasting, and coal transport). This area was the only region of Kentucky that could support large elk herds while minimizing the potential for human-elk interactions. Human activity, along with poaching, meningeal worm infection (*Parelaphostrongylus tenuis*) that causes neurological lesions in elk, and

Fig. 6 Elk (*Cervus canadensis*) originally from western population sources have thrived on surface mines reintroduction sites in the Appalachian Plateau. Note radio-telemetry collars on left- and right-most cows that allows researchers and managers the ability to track movement patterns and habitat use (Photo by D. Ledford, Appalachian Wildlife Foundation)

poor forage conditions, had the potential to reduce survival, decrease site fidelity and cause human-elk conflict in the surrounding landscape (Lankester 2001; Larkin et al. 2001). Several western states provided what was considered the Rocky Mountain elk subspecies (*Cervus canadensis nelsoni*; Larkin et al. 2001) to Kentucky; however, whether western elk would be a suitable surrogate or not for the extinct eastern elk subspecies was a concern (Polziehn et al. 2000).

In eastern Kentucky, the first few hundred elk released were outfitted with VHF radio-collars, allowing initial assessments of elk habitat use and movement. Reintroduced elk exhibited high rates of annual survival (≥90%; Larkin et al. 2003) and release site fidelity (≥53% within 10 km after one year; Larkin et al. 2004). Release site fidelity was higher where public access was lower and edge habitat was higher (Larkin et al. 2004). Indeed, both survival and natality were very high during the first few years post-translocation and may have reflected a brief period of irruptive growth in a region that, unlike western elk range, lacked harsh winters and large predators such as wolves and mountain lions (Larkin et al. 2003). Despite this early success, research indicated elk could be locally impacted by human-caused disturbances leading to changes in temporal activity and movement patterns, including disruption of herd units (Wichrowski et al. 2005; Olsson et al. 2007). Currently, regulated hunting, where allowed, is the primary mortality agent in adult elk. Meningeal worm infections have occurred, however, parasitic and other stressors have minimally impacted survival, reproduction, and recruitment (Kentucky Department of Fish and Wildlife Resources 2015; Slabach et al. 2018).

With an estimated population of ~ 3,500 (G. Jenkins, Kentucky Department of Fish and Wildlife Resources, pers. communication), elk in Kentucky are now well established, with high densities still occurring at or near some of the original surface-mine release sites. Despite high site fidelity of the Kentucky herd, some natural immigration from Kentucky into Tennessee, Virginia, and West Virginia has occurred over the past 20 years. These reintroductions occurred in Tennessee beginning in 2000 (Kindall et al. 2011), Virginia in 2012 (Virginia Department of Game and Inland

Fisheries 2020), and lastly in West Virginia in 2018 (West Virginia Division of Natural Resources 2020). Combined with reintroduction efforts in the Blue Ridge portion of the Appalachians in the Great Smoky Mountains National Park (Murrow et al. 2009), the larger region has the potential to harbor a much greater interstate population, albeit probably always centered in the Appalachian Plateau mined landscape.

Although the reintroduction of elk in Kentucky and surrounding states has so far been a program success, there may be long-term ecological and economic consequences of having high densities of a large, herd-forming herbivore in the post-mined landscape. These consequences could lead to severe ecological degradation in terms of the composition, structure, and successional trajectories of native plant communities (Bradshaw and Waller 2016), which in turn, affect other wildlife species; however, this early in the reintroduction saga, research on the potential ecological impact of reintroduced elk in the Appalachian Plateau is lacking. Shortly after their reintroduction into Kentucky on post-SMCRA sites, elk established wide (~0.3–0.6 m) trails with visible soil erosion leading from bedding grounds within the forest to reclaimed minelands where they forage during crepuscular and nocturnal periods. Heavily used bedding areas have sparse or no leaf litter, trampled/browsed vegetation (Fig. 7), and large deposits of elk feces and urine-saturated soils (TerBeest 2005). Consequently, these elk-disturbed ridgetops and ravine bottoms had higher soil ammonium, lower soil carbon, and lower soil moisture than non-elk reference forest sites. Although not quantified, Maigret et al. (2019) suggested that elk impacts to unmined forest “islands” could reduce diversity and ecological integrity of native flora and fauna and/or reduced opportunities for natural colonization of reclaimed mine sites by these taxa.

Fig. 7 Pine (*Pinus spp.*) damaged by elk (*Cervus canadensis*) antler rubbing on a reclaimed surface mine in the Appalachian Plateau (Photo by J. Cox, University of Kentucky)

Within the Appalachian Plateau, elk inhabit a very different forest ecosystem than that which existed even a century ago (Yarnell 1998). In southwest Virginia, elk monitored with GPS-collars from 2014 to 2019 were found to use all forest types equitably with their proportion in the landscape; however, their use of reclaimed mine sites was greater than expected based on availability (Quinlan et al. 2020), a finding similar to elk in Kentucky (Cox 2003). The fragmentation of forests by surface mining and the extensive use of invasive exotic plant species to reclaim these areas has created an entirely new, non-analog plant-herbivore regional dynamic. Elk rapidly takes advantage of early plant germination following hydroseeding, frequently consuming the new-growth of both forbs and grasses preventing successful establishment or arrested growth. In a large mountaintop removal mine matrix landscape, Schneider et al. (2006) found that 51% of elk diet in Kentucky consisted of native and exotic grasses and forbs commonly planted on surface mines including Brome (*Bromus* spp.), Kentucky 31 fescue (*Lolium arundinaceum*), broom sedge bluestem, orchard grass (*Dactylis glomerata*), Chinese silver grass (*Miscanthus sinensis*), crown vetch (*Coronilla varia*), Chinese-bush clover (*Lespedeza cuneata*), and red clover (*Trifolium pretense*). Many of these grasses and forbs were used by elk in Tennessee (Lupardus et al. 2011) and Pennsylvania (Heffernan 2009) as well. In Tennessee, the summer diet was dominated by forbs with jewelweed (*Impatiens* spp.) being the most selected. Schneider et al. (2006) also observed that woody browse in elk diet was dominated by black locust and autumn olive (*Elaeagnus umbellate*), two species commonly planted or invasive on surface mines (Oliphant et al 2017). In Pennsylvania, Heffernan (2009) found elk heavily used autumn olive because of the high nutritional value of the fruit in the fall. Yearling females used autumn olive and other browse species more during summer compared to males or other age groups (Heffernan 2009). In Tennessee, within an area that only contained about 12% open habitat from pasture and legacy bench and strip mines, Lupardus et al. (2011) found a far greater percentage of native plant species in elk diets; however, non-native autumn olive, tall fescue (*Festuca arundinacea*), and exotic legumes still were important diet items.

Regionally, elk appear to be seed vectors of invasive species such as multi-flora rose (*Rosa multiflora*) that can become prolific in both mined and non-mined forest sties (Schneider et al. 2006; Lupardus et al. 2011). Hackworth et al. (2018) found that elk were responsible for much of the damage to planted and volunteer tree seedlings in an active area of mine reclamation. Black locust seems particularly vulnerable to elk browsing and girdling from elk antler rubbing. Accordingly, high elk densities may complicate FRA efforts through intense herbivory, trampling, and antler rubbing whereby succession patterns along forest-mine ecotones are arrested or altered. Throughout the Appalachian Plateau, mining and post-mining action create residual overburden and structures such as sediment ponds that contain high concentrations of sodium, sulfur, and in some cases heavy metals. These sites are attractive to elk and white-tailed deer as mineral licks encouraging geophagy and creating other localized areas of high-elk impact to vegetation and soils (Campbell et al. 2004). Whether these legacy environmental contamination effects are detrimental or beneficial to elk and white-tailed deer in the region is unknown.

Nonetheless, the return of elk to the Appalachian Plateau appears to be a restoration success story. Elk were and are once again an important component of the mixed-mesophytic ecosystem of central Appalachia. As large, gregarious herbivores, they perform vital ecological roles as a prey species (neonates) for black bear and coyote, food for scavengers such as ravens (*Corax corax*), plant seed vectors, and physical modifiers of soils and plant communities. In the absence of wolves, mountain lions, or harsh winters, however, elk have demonstrated a remarkable ability to irruptively grow from founding populations and expand their range in the Appalachian Plateau, in large part to the habitat provided by reclaimed mines. Managers are just now beginning to understand the ecological implications of returning elk to a region increasingly fragmented and denatured by surface mining for coal, mineral extraction, logging, and the post-extraction land practices that ensue. Hunting will likely remain the primary mechanism of elk population control in the immediate future, and we suggest long-term studies are needed to better characterize the role of elk in these novel landscapes.

5 Conclusion

Coal mining, particularly surface mining, has left an enduring legacy in Appalachia's landscapes; its impacts on wildlife communities are complex. Overall biodiversity has declined and some species have been negatively impacted, while other species have responded positively to the novel habitats on post-mining areas. How to manage these lands going forward remains a question not fully answered. Reaching consensus between reforestation to restore lost forest habitat or maintaining large, high-quality early-successional grasslands and shrub-scrub with both native and exotic plants will require an assessment of not only wildlife needs at both the local and landscape scales but also the logistical and financial constraints for managers and stake-holders. Nonetheless, opportunities for using mined lands of the Appalachian Plateau both as a demonstration for large-scale ecological restoration and to explore approaches for managing non-analog environments with novel plant communities that have conservation value for adaptive wildlife will abound in the coming decades.

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