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FINAL GRANT PERFORMANCE REPORT

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Final Report

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Establishing Long-term Ecological Monitoring to Enhance Habitat Restoration on Reclaimed Mined Land

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Project Title: Establishing Baseline Ecological Data to Enhance Habitat Restoration on Reclaimed Mined Land

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Abstract

Despite nearly a century of severe disturbance from surface mining activities, southeast Kansas remains the most biologically diverse part of the state. Yet, less is known about the status of conservation priority species in this region than elsewhere in Kansas. The cessation of mining activities in the 1970s left a diverse patchwork of disturbed habitats that may warrant prioritization. The objectives of this project were to establish a monitoring program on previously mined sites, and to collect baseline data on species occurrence and habitat relationships, specifically targeting species of greatest conservation need (SGCN) as designated in the Kansas State Wildlife Action Plan (SWAP). We accomplished the objectives via two comprehensive graduate student research projects, which targeted birds and herpetofauna. We established more than 150 study sites in Cherokee and Crawford counties, and used point counts, nest monitoring, drift fence arrays, cover boards, camera traps, aquatic traps, dipnetting, and acoustic surveys to document species over three seasons. We analyzed data using a variety of robust statistical models. We documented more than 87 bird species, 10 amphibian species, and 19 reptile species. Of the documented species, 18 were classified as SGCN, including a previously unknown population of Eastern Newts (*Notophthalmus viridescens*). We demonstrate that previously mined sites can support diverse species assemblages, including SGCN. We recommend that previously mined sites in southeast Kansas be protected and prioritized. Invasive vegetation control, removal of fish from ponds with amphibian SGCN, and water level management could benefit SGCN on sites that were previously disturbed by surface mining. Future research in the region should focus on the effects of these suggested management actions, further documentation of the eastern newt population, and collection of long-term biodiversity monitoring data.

Introduction

Surface mining has resulted in the destruction of over 2.4 million hectares of terrestrial habitat in the United States since the 1930s (Lemke et al., 2013). In contrast to other types of mining, surface mining changes the entirety of the ecosystem structure, starting at the soil level. Soil horizons and pH levels in mined soils can take decades or centuries to return to suitable conditions for the original plant community (Skousen et al., 1994). The long-term impacts of mining on vegetation and wildlife communities are influenced by the initial reclamation efforts on the mined site, which depend on when the mining occurred. Land mined before the passing of the Surface Mining and Control Act (SMCRA) in 1977 was more likely to be abandoned to natural succession (Skousen et al., 1994; SMRCA, 1977). Following the SMCRA, the key reclamation objectives were typically to restore soil horizons and vegetation structure to the pre-mining conditions.

Cherokee and Crawford counties in southeast Kansas were mined for coal, zinc, lead, and other metals from the 1850s until the 1980s, with most surface mining areas left unreclaimed to be naturally revegetated (Bailey & Hooey, 2017). The Kansas Department of Wildlife and Parks (KDWP) and the Kansas Department of Health and the Environment (KDHE) have worked to reclaim more than 14,500 acres of formerly strip-mined areas, which are collectively known as the Mined Land Wildlife Area (MLWA). The KDWP and KDHE have already reclaimed some of this land into grasslands and marshes to help improve habitat quality for wildlife, such as waterfowl and upland game birds. The remainder of the land cover on the MLWA is comprised of forest, shrub, and water, and is surrounded by agricultural and urban land uses.

Following preliminary surveys in 2018 – 2019, we initiated two separate projects to target Species of Greatest Conservation Need (SGCN) as listed in the State Wildlife Action Plan (SWAP). We targeted SGCN in 1) upland habitats, and 2) wetlands on Mined Land Wildlife Areas and private land in southeast Kansas from 2020 – 2022. Our primary objective was to implement and elaborate upon an ecological monitoring program in southeast Kansas, with the long-term goal of enhancing restoration efforts on reclaimed mined lands. The potential of the MLWA to support SGCN may have been previously overlooked due to the severity of disturbance from strip mining. Our project sought to document the impact of land use history on the occurrence of SGCN, and particularly relationships between these species and exotic plant invasions. We accomplished the project objectives through two graduate student thesis projects, which are summarized below. See published theses (linked below) for detailed methodology, study site descriptions, habitat models, etc.

Objectives and Accomplishments

1. <u>Implement upland habitat surveys and monitoring</u>. Generate spatial habitat models for herpetofauna and bird target species, incorporating species' presence, vegetation composition, structure, and soil quality;

   We used occurrence data from bird and amphibian surveys to develop predictive habitat models for birds and amphibians. Models accounted for detection probability via distance sampling or occupancy modeling, and related the densities or occurrences of species to habitat structure or landscape composition. Models are described in detail in the two linked theses and/or are under review for publication in academic journals.

2. Strip-pit wetland surveys and monitoring. Determine habitat associations of target species within mined land aquatic habitats, specifically anurans, central newts, and prothonotary warblers

   *We modeled occurrence of breeding and larval amphibians on unreclaimed mined sites, managed mined sites, and unmined sites. Models examined habitat features within individual wetlands and landscape composition at differing spatial scales. We discovered a previously unknown population of Eastern Newts, which we described in detail. While prothonotary warblers were not abundant enough to fit habitat models, we modeled occurrence and nesting habitats of other SCGN, such as Bell's Vireo (*Vireo belli*). See linked theses and associated publications.*

3. Provide recommendations for habitat improvements in mined land habitat. Contractor will provide habitat models as part of the final report.

   See models and management implications in attached theses and associated publications.

Summary of Results

All study sites and locations where SGCN were detected are included in Appendix I and II, respectively, and in the linked theses. Scientific names for all observed species are included in Appendix II. Sampling methods are included in Appendix III and elaborated upon in the linked theses.

Drift Fence and Coverboard Surveys

We observed 21 reptile and amphibian species across 5 of the 6 study sites that were previous used for drift fence arrays and coverboards in 2020 (Table 1). Mined Land Wildlife Area (MLWA) unit 4 was not surveyed in 2020 using these methods due to flooding. Of the species observed using these methods, one was a Tier 1 SGCN species (Broadhead Skink). The two captures of skink sp. were recorded for juvenile skinks that were not identified down to species.

In 2021, these surveys were conducted at all six study sites from May 18 to August 16. We observed 26 species through the drift fence arrays, coverboards, and opportunistic sightings (Table 1). Two SGCN species were observed: Broadhead Skink and Spring Peeper.

In 2022, we converted the drift fence surveys to camera traps instead of the traditional funnel and pit fall traps. A game camera was mounted to the bottom of a bucket facing the ground at the end of each arm to capture species going by. We surveyed the six locations from previous years from March 22 to October 22, 2022. However, some sites were moved slightly to account for flooding in wetter years. We observed 25 species, when grouping bird and small mammal species together through these modified drift fence arrays (Table 1).

Table 1. Species detected and individual counts observed in 2020, 2021, and 2022 field season via coverboards, drift fence arrays, and opportunistic encounters in the area. The three field seasons were used to determine the percent of sites at which species were observed. SGCN are bolded. The species listed as unknown resulted from camera trap observations.

Species		2020	2021	2022	Sites Observed (%)
American Bullfrog	*Lithobates catesbeianus*	2	10	1	83
American Toad	*Anaxyrus americanus*	9	28	20	100
Blanchard's Cricket Frog	*Acris blanchardi*	2	68	5	100
Boreal Chorus Frog	*Pseudacris maculata*	2	3	0	33
Broadhead Skink	***Plestiodon laticeps***	**9**	**4**	**3**	**17**
Common Gartersnake	*Thamnophis sirtalus*	2	3	40	67
Common Watersnake	*Nerodia sipedon*	0	0	1	17
Dekay's Brownsnake	*Storeria dekayi*	1	4	0	50
Common Box Turtle	*Terrapene Carolina*	2	8	82	83
Common Five-lined Skink	*Plestiodon fasciatus*	3	6	0	17
Little Brown Skink	*Scincella lateralis*	0	3	1	17
Nerodia sp.		0	0	1	17
Ornate Box Turtle	*Terrapene ornate*	1	0	4	17
Painted Turtle	*Chrysemys picta*	0	4	0	50
Plain-Bellied Watersnake	*Nerodia erthrogaster*	1	6	1	67
Prairie Kingsnake	*Lampropeltis calligaster*	1	2	13	50
Thamnophis sp.		0	0	4	33
Treefrog complex	*Hyla sp.*	1	4	1	33
Six-lined Racerunner	*Aspidoscelis sexlineatus*	0	1	0	17
Skink sp.		2	2	45	33
Small Mammal sp.		32	57	8465	100
Common Snapping Turtle	*Chelydra serpentina*	0	4	1	50
Southern Leopard Frog	*Lithobates sphenocephalus*	57	66	39	100
Spring Peeper	***Pseudacris crucifer***	**0**	**1**	**0**	**17**
Western Ratsnake	*Pantherophis obseletus*	2	11	26	100
Western Ribbonsnake	*Thamnophis Proximus*	1	3	10	83
North American Racer	*Coluber constrictor*	2	5	40	83
	Total Species	**21**	**26**	**25**	

Anuran Call Survey

In 2020, we surveyed 24 different wetlands of varying habitats and size using anuran call surveys. Blanchard's Cricket Frog was heard at 19 of the 24 wetlands and American Bullfrogs and Gray Treefrog sp. were heard at 11 of 24 sites (Table 2).

In 2021, we conducted call surveys six times from March 16 to June 12 at 65 sites throughout Crawford and Cherokee cos. Nine anuran species were recorded calling throughout the survey area, with naïve occupancy varying from 18% to 100%. Blanchard's Cricket Frog (*Acris blanchardi*) was the most abundant species recorded (Table 3).

In 2022, we conducted call surveys six times from March 14 to June 10 at the same 65 sites throughout Crawford and Cherokee cos. Nine anuran species were recorded calling (Table 3). We conducted single-species occupancy modeling for four species including American Bullfrog, Crawfish Frog, Gray Treefrog, and Spring Peepers, as the rest of the species were nearly ubiquitous.

Table 2. Call strength of each anuran species heard at 24 wetlands during the preliminary survey in May and June of 2020. Strength is based on North American Amphibian Monitoring Program: 1= one call at a time, can identify individuals, 2= calls overlap slightly, but individuals can be identified, 3= full chorus, calls overlap completely.

Site	**Blanchard's Cricket Frog** (*Acris blanchardi*)	**American Bullfrog** (*Lithobates catesbeianus*)	**Gray Treefrog complex** (*Hyla* sp.)	**Number of Species Heard**
MLWA 1.1	3	1		2
MLWA 1.2	3		2	2
MLWA 4.1			1	1
MLWA 4.2	3		3	2
MLWA 4.3	3	1		2
MLWA 6.1	3			1
MLWA 7.1	3		1	2
MLWA 8.1	3		1	2
MLWA 14.1	3			1
MLWA 21.1				0
MLWA 21.2	3		3	2
MLWA 23.1	3	2		2
MLWA 23.2	3	1	2	3
MLWA 24.1	3	1		2
MLWA 24.2	3			1
MLWA 25.1				0
MLWA 25.2				0
MLWA 29.1	3		2	2
MLWA 30.1	3		2	2
MLWA 35.1	3	2	1	3
MLWA 38.1	3	1		2
MLWA 44.1	3	1	2	3
Monahan	3	1		2
Total Sites	19	11	11	

Table 3. Detections of nine anuran species heard calling from 65 sites between 2021 – 2022. Detection at each site is indicated as the following: blank = not detected, 21 = only detected in 2021, 22 = only detected in 2022, and X = detected in 2021 and 2022. We recorded two Tier 2 SGCN species (*), the Spring Peeper and Crawfish Frog.

Survey Point	Lat	Long	American Bullfrog	American Toad	Blanchard's Cricket Frog	Boreal Chorus Frog	Cope's Gray Treefrog	*Crawfish Frog	Gray Treefrog	Southern Leopard Frog	*Spring Peeper
Buche Wildlife Area	37.317	-94.682	22	X	X	X	X	X	22	X	X
Ford N	37.361	-94.914	X	22	X	X	X		X	X	
Ford S	37.3574	-94.923	X	X	X	X	X			X	
MLWA 10	37.2667	-94.81	X	X	X	21	X		21	X	X
MLWA 11	37.2657	-94.838	X	X	X	X	X			X	X
MLWA 12N	37.2588	-94.816	22	X	X	X	X			X	X
MLWA 12W	37.2521	-94.824	X	X	X	X	X		X	X	X
MLWA 13	37.2517	-94.801	X	X	X	X	X			X	X
MLWA 14	37.2443	-94.814	X	X	X	X	X	22	21	X	X
MLWA 16	37.2369	-94.833		X	X	X	X			X	X
MLWA 17S	37.2873	-94.894	X	X	X	X	X			X	
MLWA 17W	37.294	-94.905	X	X	X	X	X			X	
MLWA 18E	37.2747	-94.909	X	X	X	X	X	22		X	
MLWA 18N	37.2788	-94.923	X	X	X	X	X			X	
MLWA 18S	37.267	-94.915	X	X	X	X	X	X	21	X	
MLWA 19	37.278	-94.896	X	X	X	X	X	22		X	
MLWA 1E	37.4771	-94.693	X	X	X	X	X			X	X
MLWA 1N	37.4821	-94.703	22	X	X	X	X	21	22	X	22
MLWA 1S	37.4705	-94.703		22	X	X	X		22	22	X
MLWA 21E	37.2468	-94.96	X	X	X	X	X			X	
MLWA 21S	37.2377	-94.961	X	X	X	X	X			X	
MLWA 21W	37.2455	-94.976	X	X	X	X	X		X	X	
MLWA 22E	37.231	-94.983	X	X	X	X	X			X	
MLWA 22S	37.2237	-94.991	22	22	X	X	X		21	X	
MLWA 23	37.2363	-94.973	X	22	X	X	X			X	
MLWA 24E	37.2088	-95.001	X	X	X	X	X		22	X	
MLWA 24W	37.213	-95.012	X	22	X	X	X			X	
MLWA 25	37.1937	-95.059	21	X	X	X	X			X	
MLWA 26	37.3329	-94.8	X	X	X	X	X			X	X
MLWA 27	37.202	-95.05	X	X	X	X	X	X	22	X	
MLWA 28	37.2029	-95.032	X	X	X	X	X	X	X	X	
MLWA 29	37.2019	-95.014	X	22	X	X	X	X	22	X	
MLWA 3	37.444	-94.617	X	X	X	X	X	21		X	X

MLWA 30	37.2083	-95.023	X	22	X	X	X		22	X	
MLWA 32	37.2087	-94.978	22	X	X	X	X			X	
MLWA 33	37.225	-95.032	X	X	X	X	X		21	X	
MLWA 35E	37.2237	-95.002	X	X	X	X	X	X		X	
MLWA 35W	37.2259	-95.013	X	X	X	X	X	21	22	X	
MLWA 36	37.2446	-95.038	X	X	X	X	X			X	
MLWA 38E	37.2518	-94.927	X	X	X	X	X	X		X	
MLWA 38W	37.2486	-94.94	X	X	X	X	X	22		X	
MLWA 39	37.2527	-94.985	X	X	X	X	X		X	X	
MLWA 40	37.264	-94.976	X	22	X	X	X	22	21	X	
MLWA 41	37.2615	-94.958	22		X	X	22			X	
MLWA 42E	37.2595	-94.924	X	X	X	X	X			X	
MLWA 42W	37.2573	-94.937	X	X	X	X	X			X	
MLWA 44	37.2671	-94.935	22	X	X	X	X	22		X	
MLWA 45	37.2834	-94.912	X	X	X	X	X			X	22
MLWA 4E	37.4331	-94.617	X	X	X	X	X			X	X
MLWA 4W	37.4381	-94.631	X	X	X	X	X	21		X	X
MLWA 5	37.412	-94.769	X	X	X	X	X			X	X
MLWA 6N	37.424	-94.755	X	X	X	X	X	X		X	X
MLWA 6S	37.416	-94.758	X	X	X	X	X			X	X
MLWA 7N	37.3963	-94.779	X	X	X	X	X	21		X	X
MLWA 7S	37.388	-94.784	X	X	X	X	X	21		X	22
MLWA 8	37.39	-94.773	X	X	X	X	X	X		X	X
MLWA 9	37.2876	-94.772	22	X	X	X	X	21		X	X
Monahan Outdoor Education Center	37.351	-94.801	22	22	X	X	X			X	X
Natural History Reserve	37.3743	-94.781	X	X	X	X	X	X		X	X
Pittsburg Bike Park	37.4288	-94.693		X	X	X	X		22	X	X
Pittsburg High School	37.4091	-94.67	X	X	X	X	X			X	X
Pittsburg Industrial Park	37.4332	-94.684		X	X	X	X	21	22	X	X
Pittsburg State University	37.3914	-94.698		22	X						
Private Residence	37.4061	-94.73		X	X	X	X	21	22	X	X
Wilderness Park	37.4548	-94.714		X	X	X	X	21	22	X	X

Larvae Sampling

In 2020, six different species were found using this methodology. Only MLWA 6.1 had no herpetofauna individuals from larvae sampling. MLWA 24.1 had the greatest number of species captured, with 4 species (Table 4).

In 2021, we conducted dipnet and trapping surveys at 30 wetlands sites throughout Crawford and Cherokee cos. Surveys were conducted primarily on mined lands and were completed six times from March 16 to June 30, 2021. Through these survey methods and opportunistic sightings within 20 m of the wetland, 23 species were observed (Table 5). Three SGCN were found including Crawfish Frogs, Spring Peepers and Eastern Newts.

In 2022, we conducted dipnet and trapping surveys at 30 wetlands sites throughout Crawford and Cherokee cos. in southeast Kansas. All but one wetland was the same wetland surveyed as in 2021; we surveyed a secondary wetland at Buche Wildlife Area in 2022 as the initial wetland was dry this year. Surveys were conducted primarily on mined lands and were completed six times from March 30 to June 29, 2021. Through these survey methods, 10 amphibian species were observed (Table 4). In addition, we captured four species of snakes and one species of turtle using dipnet and trapping methods including Diamondback Watersnake (*Nerodia rhombifer*), Graham's Crayfish Snake (*Regina grahamii*), Plain-bellied Watersnake (*Nerodia erythrogaster*), Western Ribbon Snake (*Thamnophis proximus*), and Common Snapping Turtle (*Chelydra serpentina*). Three SGCN were found: Crawfish Frogs, Spring Peepers and Eastern Newts.

Eastern Newts were discovered in a previously unknown population on the westernmost edge of their range and the first recorded for Crawford Co. (Buckardt et al. 2021). In total, 6 adults and 57 larvae were found throughout the 2021 field season and one additional sampling effort for newts at two wetlands just west of Pittsburg, KS (Table 6; Buckardt et al. 2022). In 2022, additional wetland surveys occurred between July 26 to August 3, 2022, on MLWA Unit 6. Twelve wetlands were surveyed twice to examine newt larvae presence and test the influence of habitat on the presence of newt breeding. We found newts at four wetlands, one of which was the wetland that the first individual was discovered at (Table 6).

Table 4. Number of individuals captured at each wetland site over two trap nights during June 2020.

Site	Bullfrog	Blanchard's Cricket Frog	Southern Leopard Frog	*Hyla* spp.	Plain-bellied Watersnake	Diamondback Watersnake	Total Individuals	Species Richness
MLWA 1.2	4	1				3	8	3
MLWA 4.1			99				99	1
MLWA 4.3					1	1	2	2
MLWA 6.1							0	0
MLWA 14.1		47		1			48	2
MLWA 23.1	5		1		1		7	3
MLWA 24.1	1	2	45	1			49	4
MLWA 39.1	5		1			1	7	3
Buche WA	371						371	1
Monahan		1		1		9	11	3
Total	**386**	**51**	**146**	**3**	**2**	**14**	**602**	

Table 5. Amphibian species captured by dipnet and trapping at 31 sites from 2021 and 2022 in southeast Kansas. Captures are indicated as the following: blank = not captured, 21 = only captured in 2021, 22 = only captured in 2022, and X = captured in 2021 and 2022. Buche was only surveyed in 2021 and Buche 2 was only surveyed in 2022. We recorded two Tier 2 SGCN, the Spring Peeper and Crawfish Frog (*), and one Tier 1 SGCN, Eastern Newt (**).

Site	American Bullfrog	American Toad	Blanchard's Cricket Frog	Boreal Chorus Frog	*Crawfish Frog	**Eastern Newt	*Hyla* spp.	Smallmouth Salamander	Southern Leopard Frog	*Spring Peeper
Buche Wildlife Area 1	21		21							
Buche Wildlife Area 2	22	22	22		22			22	22	22
Ford E		21	22	x	22		x	21	x	
Ford W	x	21	x	22	x		x		x	
HS			x							
ML1			x	22			x		x	22
ML10	x		x						22	

ML14	x	22	x				x		x	22
ML17	x		22							
ML18	22		22						22	
ML23 N	22		22	21					x	
ML23 S	x		x	21					x	
ML24			x	x					22	
ML25	x		x						x	
ML28			x						x	
ML30	x		x	22					x	
ML35									22	
ML36	x	22	x	x			x	x	x	
ML38	21		x							
ML39	22						22		x	
ML4 E	21		21						x	22
ML4 W			22						22	
ML40	22		x	21			x	x	x	
ML44			x	21			22		x	
ML6 N				22		x	22	x	x	22
ML6 S	22		x			x	21		x	
ML7	22		x				x		x	
Monahan			22						x	
O'Malley	21	21	22	x			22	21	x	
Reserve			22							
Private Residence		22	x	x			x	x	x	22

Table 6. Number of adult and larval Eastern Newt individuals found in the pond (37.41587, -94.75501) and the marsh (37.42269, -94.75624) during each month that we surveyed for the newts. The marsh was not surveyed in August 2021 or July 2022.

	Pond		**Marsh**	
	Adult	*Larvae*	*Adult*	*Larvae*
March 2021	1	0	0	0
May 2021	2	0	0	2
June 2021	3	1	0	14
August 2021	0	40	-	-
March 2022	1	0	0	0
May 2022	0	0	0	7
June 2022	0	0	1	7
July 2022	0	1	-	-
Total	**7**	**42**	**1**	**30**

Bird Survey Results

Point Count Surveys

In 2020, we observed 71 bird species across our point count sampling locations. Of the observed species, 13 were SGCN Tier 2 (Table 7). In 2021, we observed 70 bird species across our point count sampling locations, which included most of the 2020 locations and new sites. Of the observed species, 14 were SGCN Tier 2 (Table 7). In 2022, we observed 85 species during our point count sampling, of which 14 were SGCN Tier 2 (Table 7). No SGCN Tier 1 bird species (i.e. Golden-winged Warbler) have been observed over the 2020 – 2022 survey seasons.

Table 7. Bird species and individuals observed at point count sampling locations across both sampling years. Values indicate species counts. SGCN are bolded.

Species		**2020**	**2021**	**2022**
Acadian Flycatcher	*Empidonax virescens*	0	1	10
Alder Flycatcher	*Empidonax alnorum*	0	0	7
American Crow	*Corvus brachyrhynchos*	63	105	114
American Goldfinch	*Spinus tristis*	48	54	35
American Kestrel	*Falco sparverius*	1	0	0
Baltimore Oriole	***Icterus galbula***	**5**	**7**	**10**
Barred Owl	*Strix varia*	5	0	3
Barn Swallow	*Hirundo rustica*	6	8	13
Black-and-white Warbler	*Mniotilta varia*	0	2	0
Bell's Vireo	***Vireo belli***	**57**	**116**	**103**
Belted Kingfisher	*Megaceryle alcyon*	0	0	1
Blue-gray Gnatcatcher	*Polioptila caerulea*	29	54	49
Brown-headed Cowbird	*Molothrus ater*	82	102	143
Blue Grosbeak	*Passerina caerulea*	4	0	2
Blue Jay	*Cyanocitta cristata*	27	62	41
Brown Thrasher	*Toxostoma rufum*	7	4	16
Carolina Chickadee	*Poecile carolinesis*	41	54	74
Carolina Wren	*Thryothorus ludovicianus*	95	42	45
Canada Goose	*Branta canadensis*	6	50	8
Cedar Waxwing	*Bombycilla cedrorum*	0	0	6
Chipping Sparrow	*Spizella passerina*	0	2	3
Common Gallinule	*Gallinula galeata*	2	0	0
Common Grackle	*Quiscalus quiscula*	5	2	1
Common Nighthawk	***Chordeiles minor***	**2**	**3**	**2**
Common Yellowthroat	*Geothlypis trichas*	24	48	56
Chuck-will's-widow	***Antrostomus carolinensis***	**0**	**2**	**0**
Dickcissel	***Spiza americana***	**284**	**496**	**523**
Downy Woodpecker	*Picoides pubescens*	12	15	25
Eastern Bluebird	*Sialia sialis*	1	1	5
Eastern Kingbird	***Tyrannus tyrannus***	**6**	**4**	**6**
Eastern Meadowlark	***Sturnella magna***	**21**	**31**	**32**
Eastern Phoebe	*Sayornis phoebe*	1	1	2
Eastern Towhee	*Pipilo erythrophthalmus*	35	31	35
Eastern Wood Peewee	***Contopus virens***	**52**	**64**	**51**
Eurasian Collard-Dove	*Streptopelia decaoto*	0	0	1
European Starling	*Sturnus vulgaris*	1	0	1
Fish Crow	*Corvus ossifragus*	8	23	26
Field Sparrow	*Spizella pusilla*	146	137	106
Great Blue Heron	*Andrea herodias*	12	3	3
Great-crested flycatcher	*Myiarchus crinitus*	30	57	46
Gray Catbird	*Dumetella carolinensis*	5	21	10
Great Egret	*Andrea alba*	2	1	4

Hairy woodpecker	*Leuconotopicus villosus*	0	1	4
Henslow's sparrow	***Ammodramus henslowii***	**0**	**7**	**15**
House Sparrow	*Passer domesticus*	1	0	0
Indigo Bunting	*Passerina cyanea*	159	156	181
Kentucky Warbler	***Geothlypis formosa***	**2**	**1**	**7**
Killdeer	*Charadrius vociferus*	15	12	6
Lark Sparrow	***Chondestes grammacus***	**1**	**0**	**1**
Least Flycatcher	*Empidonax minimus*	0	0	4
Louisiana Waterthrush	*Parkesia motacilla*	0	0	1
Mourning Dove	*Zenaida macroura*	80	74	65
Mourning Warbler	*Geothlypis philadelphia*	0	0	1
Northern Bobwhite	***Colinus virginianus***	**52**	**80**	**41**
Northern Cardinal	*Cardinalis cardinalis*	215	281	261
Norther Flicker	*Colaptes auratus*	5	8	1
Northern Mockingbird	*Mimus polyglottos*	21	7	13
Northern Parula	*Setophaga americana*	20	33	28
Orchard Oriole	*Icterus spurius*	12	11	12
Painted Bunting	*Passerina ciris*	0	0	2
Pileated Woodpecker	*Dryocopus pileatus*	9	25	33
Prothonotary Warbler	***Protonotaria citrea***	**11**	**12**	**11**
Purple Martin	*Progne subis*	0	1093	5
Red-Bellied Woodpecker	*Melanerpes carolinus*	68	100	95
Red-eyed Vireo	*Vireo olivaceus*	30	32	34
Red-headed Woodpecker	***Melanerpes erythrocephalus***	**14**	**5**	**6**
Red-shouldered Hawk	*Buteo lineatus*	5	5	8
Red-tailed Hawk	*Buteo jamaicensis*	11	5	5
Ruby-throated Hummingbird	*Archilochus colubris*	4	7	6
Red-winged Blackbird	*Agelaius phoeniceus*	47	87	70
Song Sparrow	*Melospiza melodia*	2	0	0
Scissor-tailed Flycatcher	***Tyrannus forficaatus***	**3**	**11**	**3**
Summer Tanager	*Piranga rubra*	13	16	17
Tree Swallow	*Tachycineta bicolor*	3	4	1
Tufted Titmouse	*Baeolophus bicolor*	73	121	100
Turkey Vulture	*Cathartes aura*	13	7	4
Warbling Vireo	*Vireo gilvus*	12	10	13
White-breasted Nuthatch	*Sitta carolinesis*	3	1	11
White-eyed Vireo	*Vireo griseus*	5	20	23
Willow Flycatcher	*Empidonax traillii*	7	3	1
Wild Turkey	*Meleagris gallopavo*	2	0	1
Wood Duck	*Aix sponsa*	0	1	1
Wood Thrush	*Hylocichla mustelina*	4	1	3
Yellow-breasted Chat	*Icteria virens*	66	115	120
Yellow-billed Cuckoo	*Coccyzus americanus*	107	116	77
Yellow Warbler	*Setophaga petechia*	1	0	3
Yellow-throated Vireo	*Vireo flavifrons*	1	3	0

Nest Monitoring

In 2020, we found 48 nests belonging to eight species across our point count study locations. In 2021, we found 78 nests belonging to six species across our point count study locations. In 2022, we found 178 nests belonging to eight species. In both 2021 and 2022, we focused our search efforts on shrub nesting species and SGCN. The majority of nests found were Bell's Vireo nests (n = 122), with a failure rate of (83%; Table 8).

Table 8. Number of nests per species found within survey area, including their success rates by year. SGCN are bolded.

Species	2020 (%Fledged)	2021 (%Fledged)	2022 (%Fledged)
Bell's Vireo	16 (23%)	49 (18%)	122 (17%)
Carolina Wren	1 (0%)	0	0
Common Nighthawk	0	2 (0%)	3 (33%)
Dickcissel	5 (20%)	2 (0%)	12 (25%)
Field Sparrow	5 (20%)	0	0
Indigo Bunting	2 (0%)	6 (17%)	8(0%)
Kentucky Warbler	0	0	1 (0%)
Lark Sparrow	0	0	1 (0%)
Northern Cardinal	8 (50%)	18 (6%)	29 (21%)
Scissor-tailed Flycatcher	0	1 (100%)	2 (50%)
Vireo sp.	2 (0%)	0	0
Yellow-billed Cuckoo	2 (50%)	0	0
White-eyed Vireo	1 (100%)	0	0
Undetermined	6 (0%)	0	0
Total Nests	**48**	**78**	**178**

Discussion

Reclamation efforts following intense human disturbances can supply habitat for a wide variety of wildlife. Even with minimal restoration efforts, the strip-mined land in our study region hosts considerable habitat variation and associated species diversity (i.e., 87 bird species, 19 reptile species, and 10 amphibian species) in both upland and wetland habitats. Formerly mined lands provide an excellent opportunity to manage a diverse habitat matrix that may benefit a wide range of species throughout the region.

We found that occupancy of anuran species depended on the specific habitat needs of each species, rather than one prevailing habitat disturbance or land cover feature. Even though the survey area has been highly impacted by anthropogenic changes like surface mining, agriculture, and urbanization, the landscape provided aquatic and terrestrial habitats that are necessary to support populations of Crawfish Frogs and Spring Peepers, both SGCN in Kansas, as well as American Bullfrogs and Gray Treefrogs. We also detected a previously unknown population of Eastern Newts in Crawford County, a Kansas SGCN, through our wetland surveys.

Many anuran species were nearly ubiquitous in this area, suggesting that this altered landscape provides the appropriate habitats to support common anuran species.

Similar to anurans, we found that the density of bird species was in response to multiple habitat features. While managing for shrubs in restored mined lands may not be suitable for all species, focusing efforts to improve habitats for shrub-dependent species of conservation concern could benefit bird diversity overall. Thirteen bird SCGN were detected throughout our surveys, of which six were found to be nesting in formerly mined lands of southeast Kansas. Daily nest survival rates of our target shrub nesting species, particularly Bell's Vireo, were not high enough to maintain their population. We did not observe a relationship between invasive shrub species and nest success; however, we did record failures for a number of nests due to livestock interactions.

Continued research is needed on anthropogenically altered landscapes to understand to a fuller extent how the landscape composition is influencing wildlife populations, as some of the species' results had high levels of uncertainty. Our surveys were based on the MLWA to study the impacts of remnant strip mined areas, but most of this region has been affected by mining. Therefore, all land cover types are impacted. However, the addition of sites not directly related to the MLWA would provide a clearer picture of how historic mining in the region influenced anuran and bird species occupancy, even for the species that were considered ubiquitous in this area. Additionally, modeling various landscape metrics like mean patch size may provide a deeper understanding how the landscape mosaic is influencing wildlife populations.

Anthropogenetic changes to a landscape impact wildlife in a variety of ways. Even so, the variation and diversity in habitat types resulting from these changes may provide sufficient habitats to support amphibians, reptiles, and birds. Due to the unique land use and mining history of this region, the availability of habitats such as forests, grasslands, open water, and wetlands, supports a variety species, including several SGCN species. The management of aquatic and terrestrial habitats across all anthropogenetic landcover types will support current and future conservation efforts.

Publications and Presentations

Master's Theses

Buckardt, E. M. (2022). *Amphibian occupancy and diversity on a post-mined landscape* (Publication No. 399) [Master's thesis, Pittsburg State University]. https://digitalcommons.pittstate.edu/etd/399

Headings, L. (2023). *Density and nest success of shrub-dependent birds on formerly strip-mined lands* (Publication No. TBD) [Master's thesis, Pittsburg State University]. https://digitalcommons.pittstate.edu/etd/

Publications

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. *In Prep*. Patterns of anuran occupancy on a post-mined landscape.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. *In Prep*. To glow or not to glow: Effectiveness of glow sticks and survey method on the capture rates of larval amphibians.

Headings, L., George, A. D., & Rega-Brodsky, C. C. *In Prep*. Densities and nest success of three shrub-dependent bird species in a post-mined landscape.

Scholes, S. A., Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. *In Prep* Getting to the root of the newt: Larval occupancy patterns of eastern news in southeast Kansas.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. *In Review*. Post-mined wetlands provide breeding habitat for amphibians. *Wetlands*.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. 2022. New newts: First Crawford County, Kansas records. *Collinsorum, 11*(1), 5-6.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. 2021. *Notophthalmus viridescens* (Eastern Newt). *Herpetological Review 53*, 571.

Presentations

2023

Headings, L., George, A., & Rega-Brodsky, C. C. (Feb 2023) Evaluating the avian and vegetative communities on strip mined land. Kansas Natural Resources Conference, Manhattan, KS.

Loomis, J., Rega-Brodsky, C. C., & George, A. D. (Apr 2023) Evaluating effectiveness of AHDriFT system for surveying small mammal and herpetofauna communities on reclaimed mined lands. Pittsburg State University Research Colloquium, Pittsburg, KS.

Loomis, J., Rega-Brodsky, C. C., & George, A. D. (*Planned* - Nov 2023) Effectiveness of AHDriFT system for surveying herpetofauna communities on reclaimed mined lands. The Wildlife Society Annual Meeting, Louisville, KY. Poster.

2022

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. (Apr 2022) Landscape composition affects anuran occupancy patterns on mined lands. Pittsburg State University Research Colloquium, Pittsburg, KS. **Tied 1st Place Graduate Oral Presentation Award**

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. (Feb 2022) Landscape composition affects anuran occupancy patterns on mined lands. Kansas Natural Resources Conference, Manhattan, KS.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. (Mar 2022) Landscape composition affects anuran occupancy patterns on mined lands. Capitol Graduate Research Summit, Topeka, KS. Poster. **1st Place Poster Award for Pittsburg State**

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. (Nov 2022) To glow or not to glow: effectiveness of glow sticks and trap method on the capture rates of larval amphibians. Kansas Herpetological Society Annual Meeting, Joplin, MO.

Headings, L., George, A., & Rega-Brodsky, C. C. (Apr 2022) Evaluating the avian and vegetative communities on strip-mined land. Pittsburg State University Research Colloquium, Pittsburg, KS. **Tied 1st Place Graduate Oral Presentation Award**

Headings, L., George, A., & Rega-Brodsky, C. C. (Feb 2022) Avian community composition and nest success on strip-mined land. Kansas Natural Resources Conference, Manhattan, KS.

Headings, L., George, A., & Rega-Brodsky, C. C. (September 2022) Evaluating the avian and vegetative communities on strip-mined land. Kansas Ornithological Society Annual Meeting, Atchison, KS.

Loomis, J., Rega-Brodsky, C. C., & George, A. D. (Nov 2022) Evaluation of AHDriFT camera trap system for long-term herpetofauna monitoring on reclaimed mined lands in southeast Kansas. Kansas Herpetological Society Annual Meeting, Joplin, MO. Poster.

Scholes, S. A., Buckardt, E. M., Rega-Brodsky, C. C., & George, A. D. (Nov 2022) To infi-newt-y and beyond: An expanding eastern newt population in Southeast Kansas. Kansas Herpetological Society Annual Meeting, Joplin, MO. Poster.

Sisson, G. L., Buckardt, E. M., Rega-Brodsky, C. C., & George, A. (Feb 2022) A newly discovered eastern newt population on abandoned mined land in Southeast Kansas. Kansas Natural Resources Conference, Manhattan, KS. Poster.

2021

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. 2021. Anuran occupancy patterns on mined lands. Kansas Herpetological Society, Pittsburg, KS. **George Toland Award for Ecological Research on North American Herpetofauna**.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. 2021. Herpetofauna communities on mined lands in Southeast Kansas. Pittsburg State University Research Colloquium, Virtual.

Buckardt, E. M., Rega-Brodsky, C. C., & George, A. 2021. Searching for amphibians in need of conservation on mined lands. Midwest Partners in Amphibian and Reptile Conservation, Virtual.

Buckardt, E.M., Rega-Brodsky, C., & George, A. 2021. Herpetofauna communities on mined lands in Southeast Kansas. Kansas Natural Resources Conference. Virtual Poster.

Headings, L., George, A., & Brodsky, C. 2021. Evaluating the avian and vegetative communities of Mined Land Wildlife Areas in Cherokee and Crawford counties. Kansas Natural Resources Conference. Virtual Poster.

Headings, L., George, A., Rega-Brodsky, C. C. 2021. Evaluating the avian and vegetative communities of Mined Land Wildlife Areas in Cherokee and Crawford counties. Pittsburg State University Research Colloquium, Virtual. **2nd Place Graduate Presentation**

Headings, L., George, A., Rega-Brodsky, C. C. 2021. Evaluating the avian and vegetative communities on strip mined land: Year one update. Kansas Ornithological Society Annual Meeting, Virtual.

Sisson, G. L., Buckardt, E. M., Rega-Brodsky, C. C., George, A. 2021. A newly discovered Eastern Newt population on abandoned mined land in southeast Kansas. Kansas Herpetological Society Annual Meeting. Poster.

2020

Buckardt, E.M., Rega-Brodsky, C., & George, A. 2020. Herpetofauna communities on mined lands in Southeast Kansas. Kansas Herpetological Society Annual Meeting, Virtual.

Headings, L., George, A., & Brodsky, C. 2020. Evaluating the avian and vegetative communities of Mined Land Wildlife Areas in Cherokee and Crawford counties. Kansas Ornithological Society Annual Meeting, Virtual.

People Reached with Project

Graduate Research Assistants

Emma Buckardt (2020 – 2022)
Luke Headings (2020 – 2023)

Field Assistants / Volunteers

Annika Anzjon (2021)
Ashlyn Henderson (2020, 2021)
Austin Abram (2021, 2022)
Gizelle Sisson (2021)
Haley Price (2021)
James Loomis (2022, 2023)
Jeremiah Casner (2022)
John Jameson (2021)
Jossie Shumaker (2021)
Kit Garvin (2021)
Kyle Findley (2020)
Maddie Gay (2021)
Mary Marine (2021)
Ryan McGinty (2020)
Sara Scholes (2022)
Taylor Michael (2021)

Presented findings at the following conferences via poster and oral presentations:

Kansas Natural Resource Conference: 2021, 2022, 2023
Kansas Ornithological Society Annual Meeting: 2020, 2021, 2022
Kansas Herpetological Society Annual Meeting: 2020, 2021, 2022
Midwest Partners in Amphibian and Reptile Conservation: 2021
Pittsburg State University Research Colloquium: 2021, 2022, 2023
Capitol Graduate Research Summit: 2022
The Wildlife Society Annual Meeting: *Planned* 2023

Appendix I. Region and individual site maps with sampling locations for bird and herpetofauna communities.

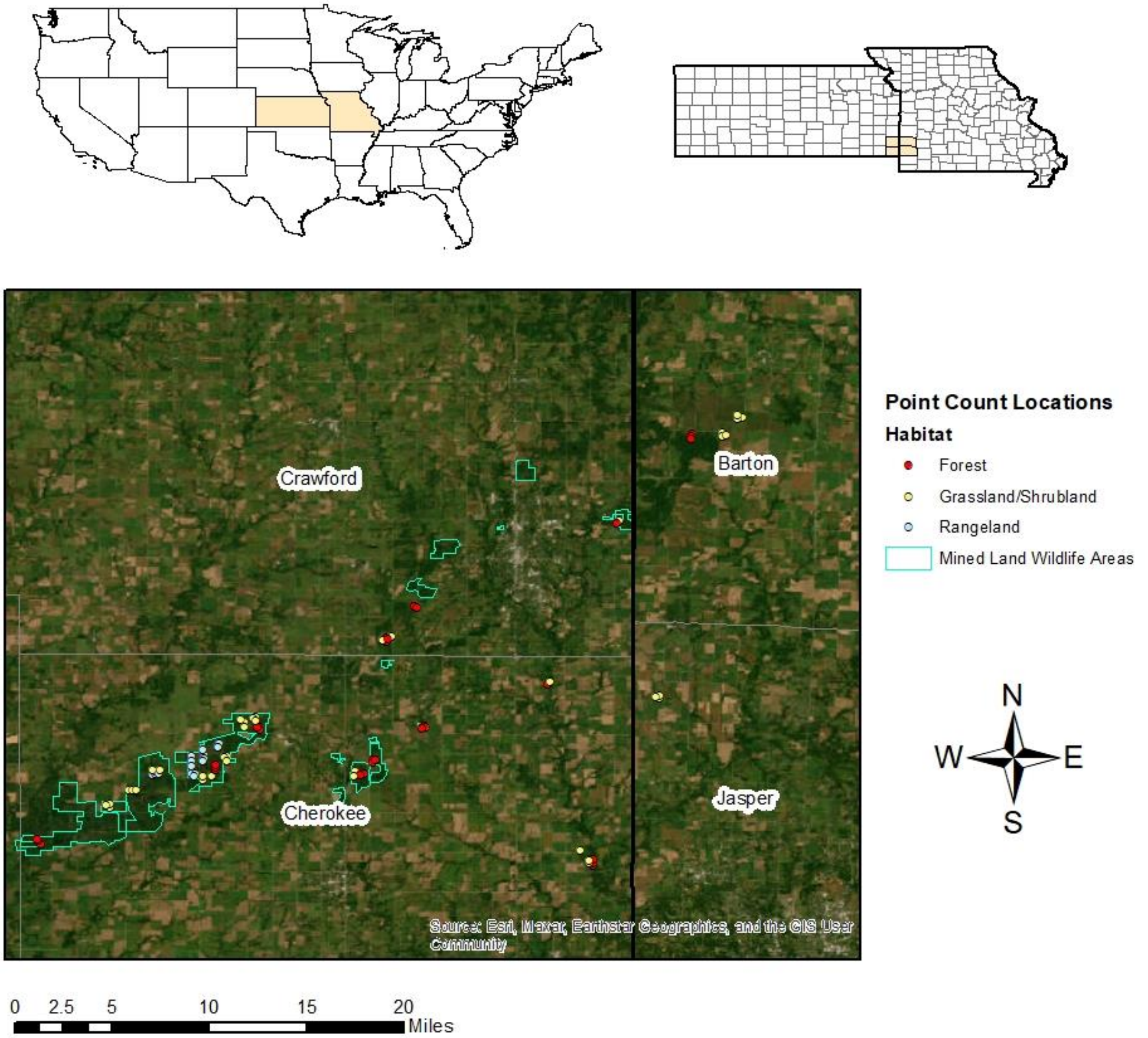

Figure A1. Regional map of the bird sampling locations across three habitat types between 2020 – 2022.

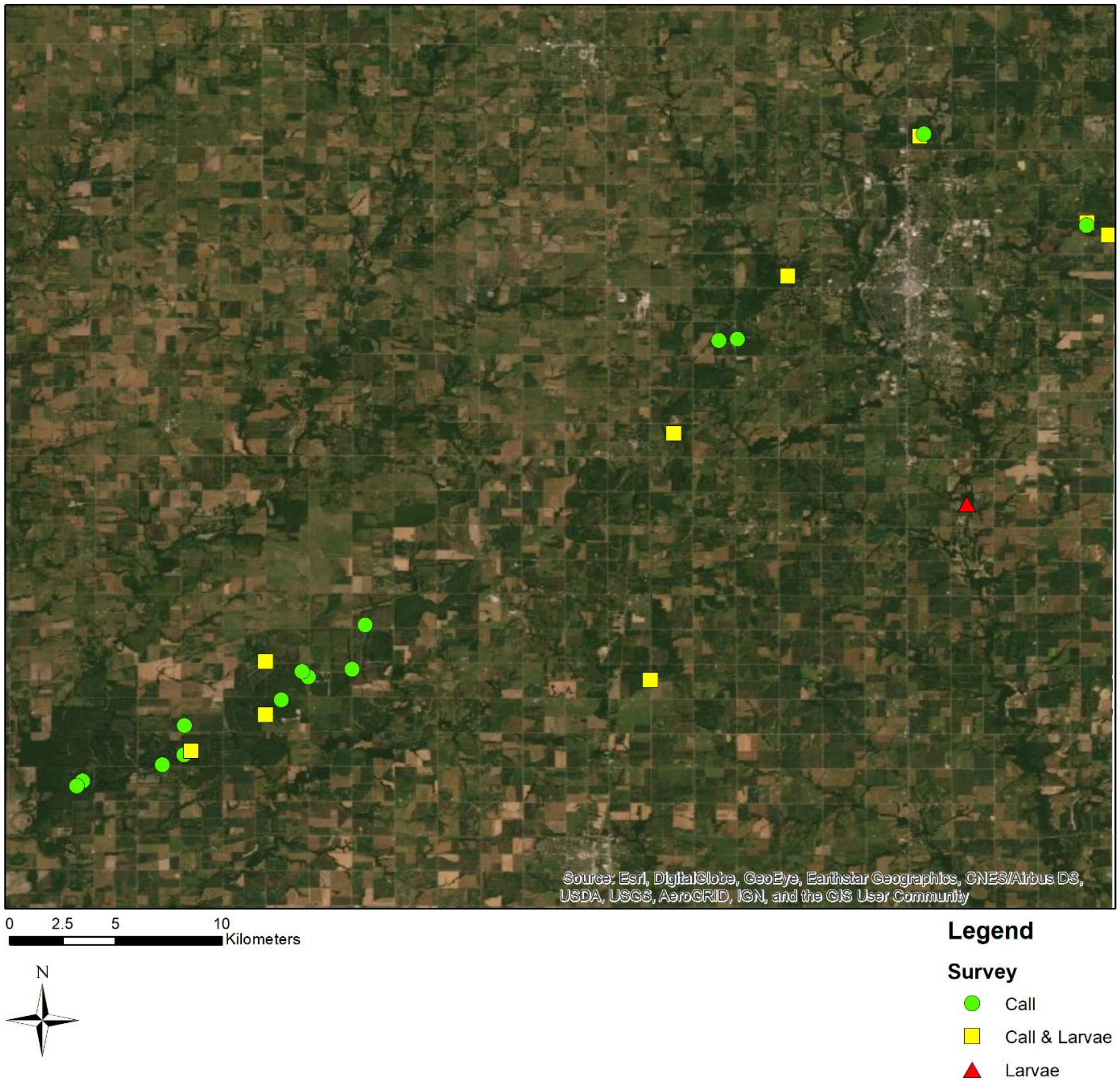

Figure A2. Regional map of herpetofauna preliminary sampling locations during 2020 within upland and bottomland habitats. Sites are indicated by their surveys conducted on site.

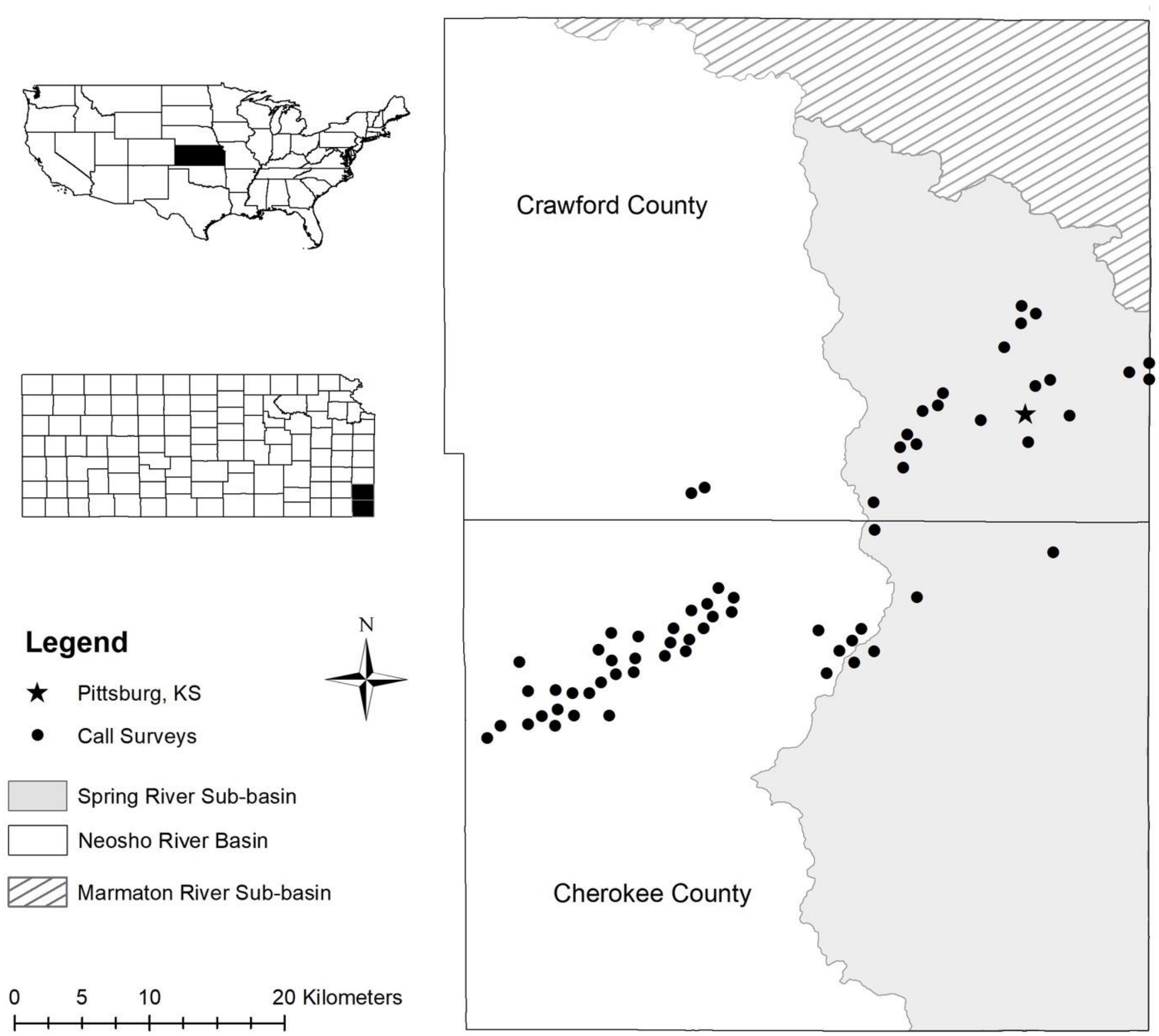

Figure A3. Anuran call survey locations with watersheds depicted. Dots represents call survey locations for 2021 and 2022.

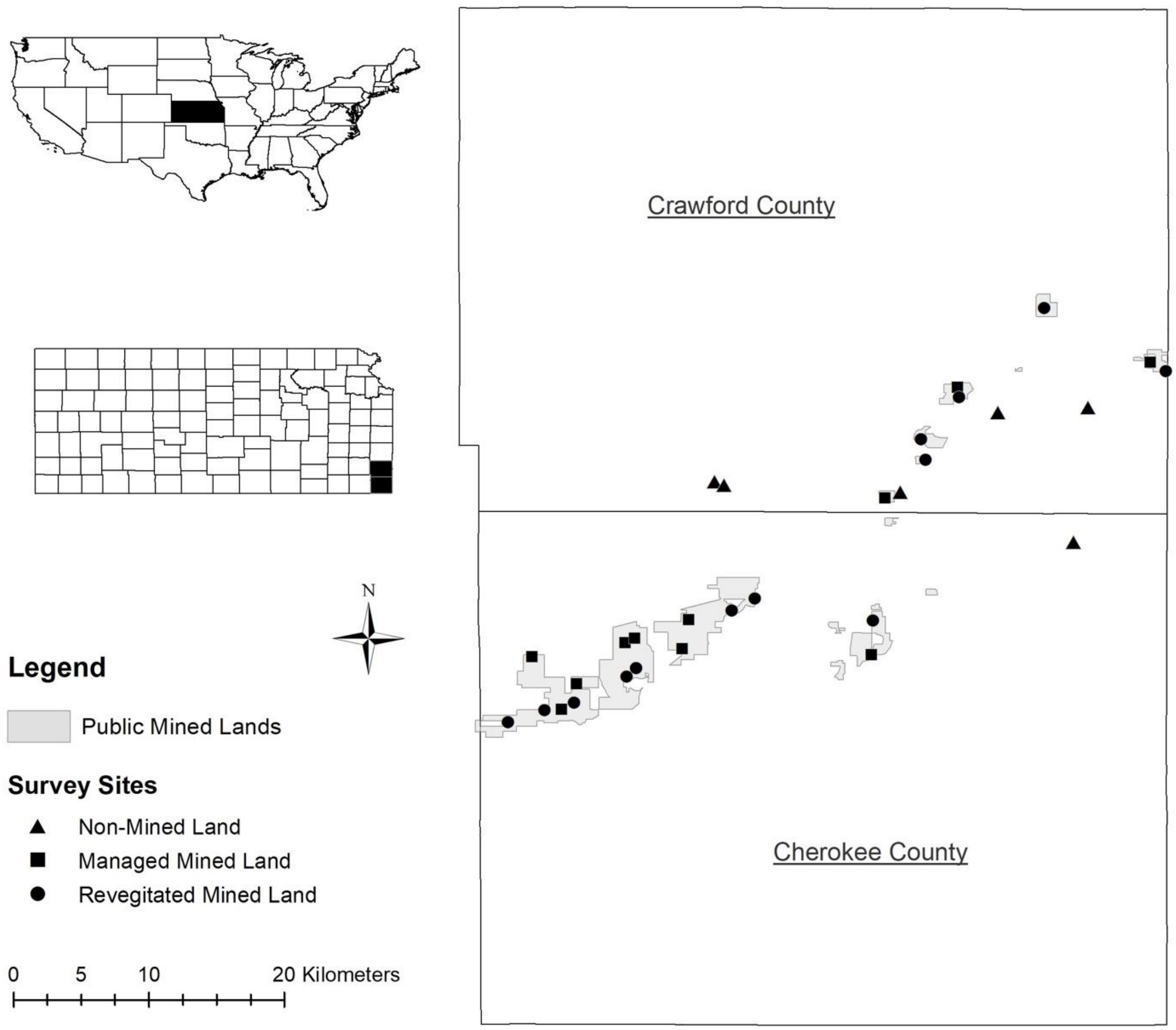

Figure A4. Map of the wetland survey area with sites indicated by the mining and reclamation status of each wetland in southeast Kansas during 2021 and 2022. Public mined lands in the area are shaded, including Mined Land Wildlife Area and Southeast Kansas Biological Station.

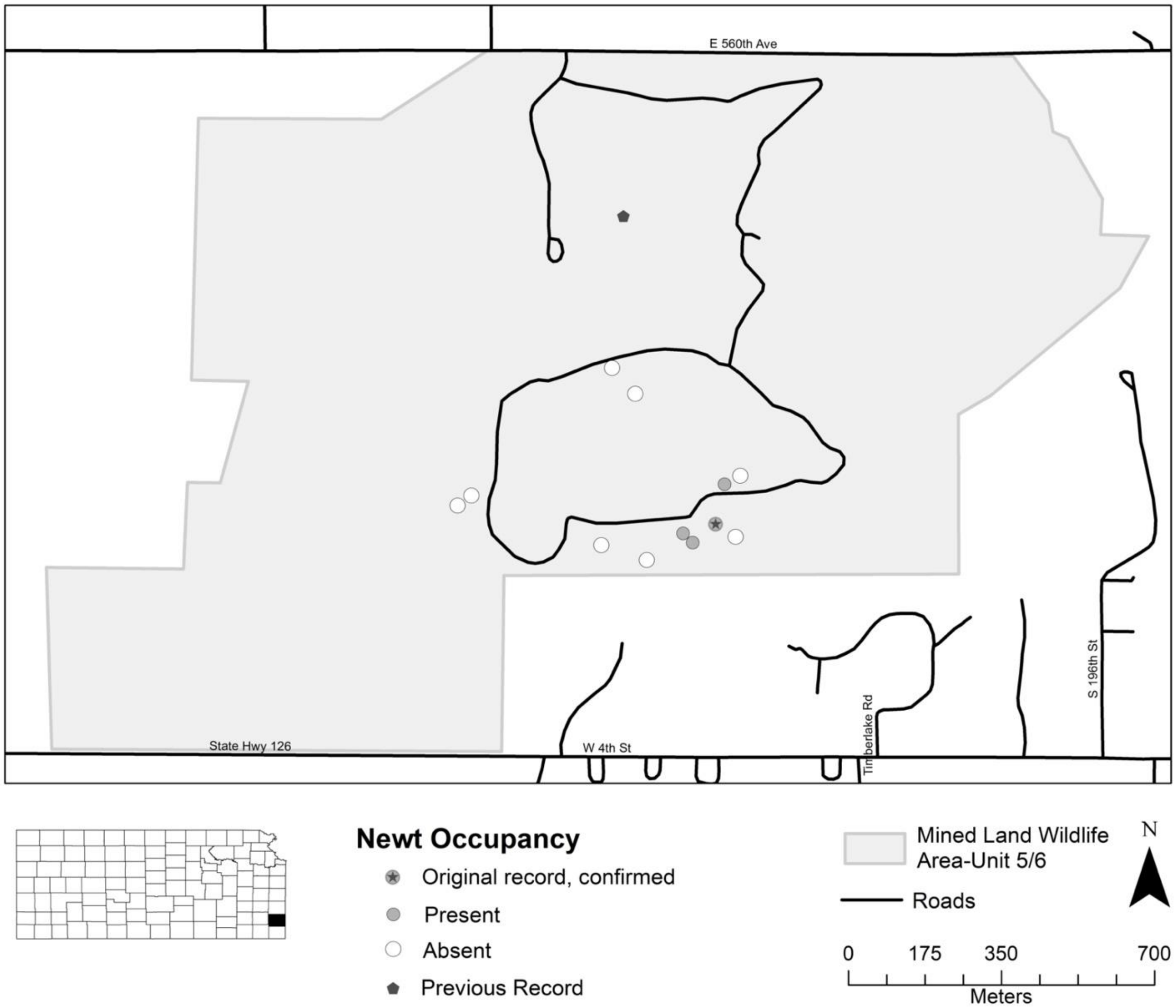

Figure A5. Survey locations of Eastern Newt surveys in July and August 2022. Locations are marked by the presences or absence of eastern newts.

Appendix III. SGCN detection locations.

Table A1. Amphibian and reptile SGCN detection locations across the project sampling period (2020 – 2022).

Location	Lat	Long	Broadhead Skink	Crawfish Frog	Eastern Newt	Spring Peeper
Buche Wildlife Area 2	37.316974	-94.682287	X	X		X
Ford E	37.35854	-94.91227		X		
Ford W	37.36025	-94.91761		X		
MLWA 10	37.266704	-94.80956				X
MLWA 11	37.2657	-94.837879				X
MLWA 12N	37.25883	-94.815712				X
MLWA 12W	37.25212	-94.823983				X
MLWA 13	37.251744	-94.800832				X
MLWA 14	37.244293	-94.814228		X		X
MLWA 16	37.236934	-94.832718				X
MLWA 18E	37.27468	-94.908684		X		
MLWA 18S	37.266982	-94.914834		X		
MLWA 19	37.278018	-94.895768		X		
MLWA 1E	37.477094	-94.692814				X
MLWA 1N	37.482111	-94.702619		X		X
MLWA 1S	37.470528	-94.702748				X
MLWA 26	37.332893	-94.800483				X
MLWA 27	37.202004	-95.050163		X		
MLWA 28	37.202911	-95.031941		X		
MLWA 29	37.201895	-95.013651		X		
MLWA 3	37.443976	-94.6174		X		X
MLWA 35E	37.223696	-95.002268		X		
MLWA 35W	37.22587	-95.013272		X		
MLWA 38E	37.251762	-94.926703		X		
MLWA 38W	37.248576	-94.940461		X		
MLWA 40	37.264013	-94.976427		X		
MLWA 44	37.267074	-94.934636		X		
MLWA 45	37.283367	-94.912269				X
MLWA 4E	37.433128	-94.617333				X
MLWA 4W	37.43806	-94.630769		X		X
MLWA 5	37.411957	-94.7687				X
MLWA 6 Newt Marsh	37.42269	-94.75624			X	
MLWA 6 Newt Pond	37.41587	94.75501			X	
MLWA 6N	37.423991	-94.754964		X	X	X
MLWA 6S	37.415987	-94.758231			X	X
MLWA 7N	37.396332	-94.778641		X		X

Location	**Lat**	**Long**	**Broadhead Skink**	**Crawfish Frog**	**Eastern Newt**	**Spring Peeper**
MLWA 7S	37.38804	-94.783519		X		X
MLWA 8	37.389996	-94.77259		X		X
MLWA 9	37.287609	-94.772275		X		X
Monahan Outdoor Education Center	37.350972	-94.801386				X
Natural History Reserve	37.374343	-94.781406		X		X
Newt 2022 Location A	37.41679	-94.7552			X	
Newt 2022 Location B	37.41557	-94.75562			X	
Newt 2022 Location C	37.41586	-94.75603			X	
Pittsburg Bike Park	37.428762	-94.69338				X
Pittsburg High School	37.409146	-94.670453				X
Pittsburg Industrial Park	37.433169	-94.683672		X		X
Private Residence	37.406102	-94.729889		X		X
Wilderness Park	37.454764	-94.713891		X		X

Table A2. Bird SGCN detection locations across the project sampling period (2020 – 2022). The following SGCN were detected: Baltimore Orioles (BAOR), Bell's Vireo (BEVI), Common Nighthawk (CONI), Chuck-will's-widow (CWWI), Dickcissel (DICK), Eastern Kingbird (EAKI), and Eastern Meadowlark (EAME). Additional bird SGCN are listed in Table A2.

Location	Lat	Long	BAOR	BEVI	CONI	CWWI	DICK	EAKI	EAME
HUNKAH_1	37.50479	-94.5517		X			X		
HUNKAH_2	37.50209	-94.5516		X			X	X	X
HUNKAH_3	37.50302	-94.5482		X			X	X	X
ML_12G	37.253	-94.8247		X			X	X	
ML_13_2	37.2618	-94.8118	X						X
ML_13_3	37.26007	-94.8118	X						X
ML_14_1F	37.25096	-94.8179			X				
ML_14G	37.24955	-94.826		X			X		
ML_17_1G	37.293	-94.8986		X			X		X
ML_17_2F	37.28442	-94.8957					X		
ML_17_2G	37.29214	-94.9011		X			X		X
ML_17_3G	37.29069	-94.8992		X			X		
ML_17_4F	37.28614	-94.898	X						
ML_20_1	37.23908	-94.9933		X					
ML_20_3	37.23901	-94.9875	X					X	
ML_21_1	37.25033	-94.9757		X			X	X	
ML_21_2	37.25123	-94.9736	X	X			X		
ML_21_3	37.25108	-94.9708		X			X		
ML_25_1	37.1993	-95.0591	X						
ML_35_1	37.22588	-95.0072		X	X		X		
ML_35_2	37.22835	-95.0077					X	X	
ML_35_3	37.22798	-95.0104					X		X
ML_37_1	37.2514	-94.9468		X			X		X
ML_37_2	37.25142	-94.9435	X	X			X		X
ML_37_3	37.24955	-94.9451		X			X		
ML_38_2	37.24954	-94.9317					X	X	
ML_38_3	37.24636	-94.9381		X			X		
ML_38_4	37.24903	-94.9382		X			X		
ML_40_1	37.25325	-94.9734		X	X		X		
ML_40_2	37.25445	-94.9754	X	X			X	X	
ML_40_3	37.25367	-94.9703	X	X			X		
ML_41_1	37.26399	-94.9462	X	X			X		X
ML_41_3	37.26046	-94.9464	X		X		X		X
ML_41_4	37.2569	-94.9464		X			X	X	X
ML_42_1F	37.25395	-94.9285					X		
ML_42_1G	37.26539	-94.9374		X			X		X

Location	Lat	Long	BAOR	BEVI	CONI	CWWI	DICK	EAKI	EAME
ML_42_2G	37.26271	-94.9375	X	X			X	X	
ML_42_3G	37.26033	-94.938	X	X			X		
ML_43_1	37.26473	-94.9203	X	X			X		
ML_43_2	37.26309	-94.9219		X			X		
ML_43_3	37.26108	-94.9203	X	X			X	X	
ML_44_1	37.26912	-94.9378					X		X
ML_44_3	37.27381	-94.9265		X			X		
ML_44_4	37.27101	-94.9265		X			X		
ML_45_1	37.29174	-94.9104		X			X		
ML_45_2	37.29003	-94.907	X	X			X		
ML_45_4	37.28623	-94.9071		X			X		
MO_1F	37.35004	-94.8016					X		
MO_1G	37.35275	-94.8019		X			X		
MO_2F	37.35116	-94.8002		X			X		
MO_2G	37.35035	-94.8039		X			X		
O'MALLEY	37.35323	-94.7969					X		
PSP_1	37.51661	-94.5364		X			X		X
PSP_2	37.51569	-94.5398		X			X		X
PSP_3	37.51839	-94.5398		X			X		X
RESERVE_2	37.37474	-94.7792	X						X
SR_1G	37.18439	-94.6511					X		X
SR_2G	37.18657	-94.6511				X	X		
SR_3G	37.19451	-94.6577	X	X			X		X
Wah-Sha-She_1	37.30702	-94.5981					X		X
Wah-Sha-She_2	37.30979	-94.598					X		X
Wah-Sha-She_1	37.308	-94.6015					X		X

Table A2. Continued. Bird SGCN detection locations across the project sampling period (2020 – 2022). The following SGCN were detected: Eastern Wood-Pewee (EAWP), Henslow's Sparrow (HESP), Kentucky Warbler (KEWA), Lark Sparrow (LASP), Northern Bobwhite (NOBO), Prothonotary Warbler (PROW), Red-headed Woodpecker (RHWO), and Scissor-tailed Flycatcher (STFL).

Location	Lat	Long	EAWP	HESP	KEWA	LASP	NOBO	PROW	RHWO	STFL
BUCHE_F	37.31742	-94.6813	X						X	
BUCHE_G	37.31946	-94.6801	X							
HUNKAH_1	37.50479	-94.5517		X			X			
HUNKAH_2	37.50209	-94.5516		X			X			
HUNKAH_3	37.50302	-94.5482		X			X			X
LD_1	37.50431	-94.5737	X							
LD_2	37.5025	-94.5748	X		X					
LD_3	37.50064	-94.5748	X							
ML_04F	37.43737	-94.6297	X					X		
ML_04G	37.43937	-94.6284	X					X		
ML_09_1	37.28525	-94.7757	X				X	X	X	
ML_09_2	37.28696	-94.7738	X							
ML_09_3	37.28556	-94.7722	X					X	X	
ML_12G	37.253	-94.8247	X						X	
ML_13_2	37.2618	-94.8118	X		X					
ML_13_3	37.26007	-94.8118	X		X					
ML_13_4	37.26188	-94.8095	X							
ML_14_1F	37.25096	-94.8179	X		X					
ML_14_2F	37.25041	-94.8215	X					X		
ML_17_1F	37.28627	-94.8957	X		X			X		
ML_17_1G	37.293	-94.8986	X				X		X	
ML_17_2F	37.28442	-94.8957	X							
ML_17_2G	37.29214	-94.9011					X			
ML_17_3G	37.29069	-94.8992					X			
ML_17_4F	37.28614	-94.898	X				X			

Location	Lat	Long	EAWP	HESP	KEWA	LASP	NOBO	PROW	RHWO	STFL
ML_20_1	37.23908	-94.9933	X				X	X		
ML_20_2	37.23901	-94.9903	X				X	X		
ML_20_3	37.23901	-94.9875	X							
ML_21_1	37.25033	-94.9757	X			X	X		X	
ML_21_2	37.25123	-94.9736					X			
ML_21_3	37.25108	-94.9708					X			
ML_25_1	37.1993	-95.0591	X					X	X	
ML_25_2	37.20121	-95.0613	X							
ML_25_3	37.20305	-95.0622	X		X			X	X	
ML_35_1	37.22588	-95.0072					X			
ML_35_2	37.22835	-95.0077	X				X			
ML_35_3	37.22798	-95.0104					X			
ML_37_1	37.2514	-94.9468					X		X	X
ML_37_2	37.25142	-94.9435					X			
ML_37_3	37.24955	-94.9451				X	X			X
ML_38_2	37.24954	-94.9317	X				X			
ML_38_4	37.24903	-94.9382					X			
ML_40_3	37.25367	-94.9703							X	
ML_41_1	37.26399	-94.9462					X			X
ML_41_3	37.26046	-94.9464					X			X
ML_41_4	37.2569	-94.9464					X			
ML_42_1F	37.25395	-94.9285	X					X	X	
ML_42_1G	37.26539	-94.9374					X			X
ML_42_2F	37.25577	-94.9288	X					X		
ML_42_2G	37.26271	-94.9375					X			
ML_42_3F	37.25771	-94.9287	X					X		
ML_42_3G	37.26033	-94.938	X				X			X
ML_43_1	37.26473	-94.9203					X			
ML_43_2	37.26309	-94.9219					X			

Location	Lat	Long	EAWP	HESP	KEWA	LASP	NOBO	PROW	RHWO	STFL
ML_43_3	37.26108	-94.9203					X			
ML_44_1	37.26912	-94.9378					X			X
ML_44_3	37.27381	-94.9265					X		X	X
ML_44_4	37.27101	-94.9265					X			
ML_45_1	37.29174	-94.9104					X			
ML_45_2	37.29003	-94.907					X			
ML_45_4	37.28623	-94.9071					X			
MO_1F	37.35004	-94.8016	X							
MO_1G	37.35275	-94.8019							X	X
MO_2F	37.35116	-94.8002	X							
MO_2G	37.35035	-94.8039	X							
O'MALLEY	37.35323	-94.7969	X						X	
PSP_1	37.51661	-94.5364					X			
PSP_2	37.51569	-94.5398					X			
PSP_3	37.51839	-94.5398		X			X			
RESERVE_1	37.3759	-94.7809	X							
RESERVE_2	37.37474	-94.7792						X		
SR_1F	37.18297	-94.6483	X							
SR_1G	37.18439	-94.6511					X	X		
SR_2F	37.18477	-94.6482	X		X					
SR_2G	37.18657	-94.6511	X				X			
SR_3F	37.18747	-94.6482	X							
SR_3G	37.19451	-94.6577					X			
Wah-Sha-She_1	37.30702	-94.5981		X			X			
Wah-Sha-She_2	37.30979	-94.598		X			X			X
Wah-Sha-She_1	37.308	-94.6015					X			

Appendix III. Objectives, study sites, and methods for each objective. Further details are provided in the corresponding theses.

Study Area Selection

In 2018, monitoring stations were established on six sites in Crawford and Cherokee counties, including the Monahan Outdoor Education Center, O'Malley Prairie, the Natural History Reserve, the Buche Wildlife Area, and Mined Land Areas 4 and 14. Beginning in 2020, we sampled additional locations in most MWLAs in those two counties (see maps and tables within Appendix I and II). These areas are representative of the region's diverse habitats and may contain our target species. Sites were located on both private and public lands that may amenable to habitat improvement to promote biodiversity. All sampling locations were recorded with a handheld GPS unit.

Project Objectives

1. Upland Habitat Surveys and Monitoring

    a. *Study objectives*: 1) Generate spatial habitat models for herpetofauna and bird target species, incorporating species' presence, vegetation composition, structure, and soil quality; 2) Provide recommendations for terrestrial habitat management in mined lands

    b. *Habitat assessment:* Standard 0.04 ha vegetation plots (11.3 m radius; James and Shugart, 1970) were centered at each sampling location. Within each plot, we assessed the following: tree species, abundance, and diameter-at-breast-height, and canopy cover with a spherical densiometer. Shrub species and their percent cover were assessed within the plot. Percent ground cover were assessed in 5 randomly located quadrats in the vegetation sampling plot with a Daubenmire frame. The following ground cover classes were used to characterize the vegetation: artificial surface, bare soil, forbs, grass, leaf litter, rock, shrubs, trees, woody litter, and water. Vertical density was assessed with a Nudds board for five height classes, each 0.5 m. Particular emphasis was placed on assessing the location and density of any exotic shrub or ground cover within each vegetation plot. We sampled soil horizon A depth at each sampling location.

    c. *Bird surveys*

        i. **Point Counts:** Fixed-radius point counts were centered at the sampling location during the breeding season (May – July; Bibby et al. 2000). All species identified via sight and/or sound were recorded, along with their distance from the observer, within a five-minute period. Surveys occurred during peak bird activity, between sunrise and four hours post-sunrise. We recorded wind speed, temperature, time of observation, observer, and date to account for these variables' impacts on detection.

        ii. **Nest Monitoring**: We searched for and monitored nests of Species of Greatest Conservation Need following standard nest monitoring procedures. We also targeted shrub-nesting species, specifically Northern Cardinal (*Cardinalis cardinalis*) and Indigo Bunting (*Passerina cyanea*), and monitored any nest found.

d. *Herpetofauna surveys*

i. **Cover Boards**: We monitored each of the study sites for herpetofauna from May through July using standard techniques (Willson and Gibbons 2009, Graeter et al. 2013). We placed approximately eight 0.6 x 1.2 m cover boards along transects on six of our study areas. Coverboards were checked once per week.

ii. **Drift Fence and Funnel Traps**: A drift fence array with pitfall and funnel traps was constructed at each of the six study sites in 2018 (Fig. A1). We continued using these sites in 2020, with the exception of the one located at MWLA 4 due to flooding. All six sites were used in 2021. Each of the arrays includes three 15-m sections of silt-fencing placed at 120^{o} angles. Each array includes four pitfall traps placed in the center and terminal ends, and three funnel traps in the center of each arm.

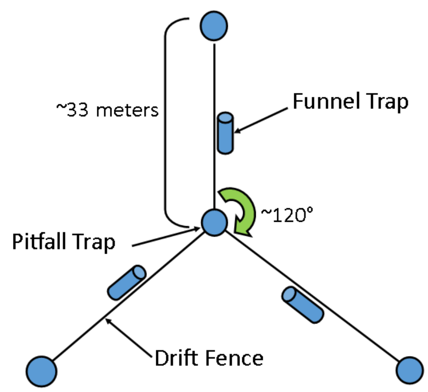

Figure A6. Drift fence array schematics.

iii. **AHDriFT Array**: A modified drift fence array survey with camera traps at the terminal nodes was constructed at each of the six study sites in 2021 (Fig. A6). Each of the arrays included three game cameras suspended in an overturned bucket at the ends of three 15-m sections of corrugated plastic fencing placed at 120^{o} angles. Camera photographs were downloaded biweekly – monthly and processed via Wildlife Insights (https://www.wildlifeinsights.org/).

Figure A7. AHDriFT array terminal buckets with suspended game cameras.

e. *Data analysis:* See corresponding theses (Buckardt, 2022; Headings, 2023) for data analysis methods.

2. Strip-pit Wetland Surveys and Monitoring

a. Study objectives: 1) Determine habitat associations of target species within mined land aquatic habitats, specifically anurans, central newts, and prothonotary warblers; 2) Provide recommendations for habitat alterations to promote the colonization or persistence of target species.

b. Habitat assessment: Standard measurements included water depth, temperature, substrate characteristics, vegetation density, and basic water quality metrics (pH, dissolved oxygen, etc.). We also assessed habitat surrounding wetlands and landscape characteristics. See Buckardt (2022) thesis for data analysis methods.

c. Herpetofaunal Surveys:

i. **Anuran call surveys:** During the 2020 field season, auditory Anuran surveys were conducted at 23 of the 24 sites above between June 1st and June 6th. The Buche Wildlife Area was not included in the auditory surveys, because this site was not added until after the proposed survey window had passed. At each site, a surveyor stood still approximately 1 m away from the wetland edge and roughly in the same location during each survey window. There was a 1-minute acclimation period before calls are recorded (Stevens et al., 2002). After the acclimation period, surveyor listened for anuran calls during a 5-minute listening window and record the strength of the chorus for every species heard (Crouch & Paton, 2002; Pierce & Gutzwiller, 2004). The strength of the chorus was determined on the following index based on North American Amphibian Monitoring Program (NAAMP): 1-Individuals can be counted; space between calls, 2-calls of individuals can be distinguished; some overlapping of calls, 3-full chorus, calls are constant, continuous, and overlapping (Weir & Mossman, 2005). During the 2021 field season, these surveys were conducted at 65 roadside sites between March 15 and June 12, 2021. Call surveys were conducted in the same manner as 2020, except survey sites were in parking lots and along roadways.

ii. **Dip-net surveys:** We conducted dip net surveys at 10 wetlands to document the presence of central newts or larval anurans between June 29 – July 9, 2020. Following the methods of Anderson and Arruda (2006), we estimated abundance of larval amphibians and document physical abnormalities. Larvae were collected using dip nets and minnow traps. Minnow traps with glow sticks were used to increase capture success of more elusive species. In 2021, 30 wetlands were sampled ranging from naturally revegetated mined land, reclaimed mined lands, and non-mined land sites. Six trap nights were conducted at each wetland between March 24 – June 30, 2021.

d. *Bird surveys:* We utilized point count data collected at the sampling areas to determine the presence of species associated with surveyed habitats.

e. *Data analysis:* Where sufficient data were available, we used a model-selection approach to identify the most important habitat or landscape characteristics affecting whether larval amphibians and focal bird species were present. The best supported models permit us to identify potential locations for habitat restoration to promote conservation of target species. See corresponding theses (Buckartdt, 2022; Headings, 2023) for data analysis methods.

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AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE

A Thesis Submitted to the Graduate School in Partial Fulfillment of the Requirements for the Degree of Master of Science

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AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE

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AMPHIBIAN OCCUPANCY AND DIVERSITY ON A POST-MINED LANDSCAPE

An Abstract of the Thesis by
Emma M. Buckardt

Amphibian populations are declining globally, with habitat loss and fragmentation being a leading cause for their decline. Anthropogenic changes to a landscape, such as urbanization, agriculture, and surface mining, leave few native habitats intact, which can influence amphibian populations and communities to varying degrees. Amphibians can provide insight into the health of ecosystems because they are sensitive to changes in their environment. Thus, they can be considered indicator species in anthropogenically altered wetlands. The goal of this study was to characterize amphibian communities that are using surface mined lands that have undergone vegetative succession. For Chapter I, we used call surveys to model occupancy of four anuran species, two of which are species in need of conservation (SINC; crawfish frog [*Lithobates areolatus*] and spring peeper [*Pseudacris crucifer*]). We found that anthropogenic landscape features, such as the percent of open water and cropland land cover, provided the necessary habitat to support the anuran community. In Chapter II, we evaluated the wetland characteristics that influenced the occupancy of five focal larval anuran species and the species richness and diversity of the amphibian community. We captured ten species of amphibians, including the first county record of eastern newt (*Notophthalmus viridescens*), a SINC species. Although our findings varied for each species, the change in wetland area, presence of predatory fish, water conductivity level, and percent of emergent vegetative cover explained the variation in occupancy patterns for most species and for the amphibian community within a wetland. We also found that larval amphibian

communities did not differ between management or land use history of the site. Lastly in Chapter III, we assessed the efficacy of survey methodology on the capture rates of larval amphibians. We found that baiting minnow traps with green glowsticks increased capture rates overall, but this effect was species-specific and varied by the time of year. The findings from all three studies provide important insights regarding amphibian use of formerly mined landscapes. The factors that determine species occupancy and community structure are related to both landscape composition and local habitat features, regardless of land-use history. Even sites that have been heavily disturbed by surface mining can potentially provide habitat for multiple amphibian species, including at-risk species. The conservation value of these recovering wetlands warrants their management and protection.

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CHAPTER I

PATTERNS OF ANURAN OCCUPANCY ON A POST-MINED LANDSCAPE

ABSTRACT

Anuran populations are declining globally, with habitat loss and fragmentation being a leading cause for their decline. Anthropogenic changes to a landscape, such as urbanization, agriculture, and surface mining, leave few native habitats intact, which can influence anuran populations and communities to varying degrees. Our study aimed to assess the connection between anuran occupancy and anthropogenic and native habitats across a landscape that was heavily disturbed by surface mining and row crop agriculture. We conducted call surveys six times from mid-March to mid-June in 2021 and 2022 at 65 sites throughout Crawford and Cherokee cos. in southeast Kansas. We conducted single-species single-season occupancy modeling for four out of nine detected anuran species, as the other species were nearly ubiquitous on the landscape. We used land cover types to model occupancy for American bullfrog (*Lithobates catesbeianus*), crawfish frog (*Lithobates areolatus*), gray treefrog (*Hyla versicolor*), and spring peeper (*Pseudacris crucifer*). We recorded nine anuran species calling, with naïve occupancy varying from 38% to 100%. American bullfrogs were positively associated with open water and built cover, while gray treefrogs had a weak association with grasslands. Crawfish frogs were positively associated with croplands and had a slightly higher occupancy in the Spring

River sub-basin in 2022. Spring peepers' occupancy was nearly exclusively within the Spring River sub-basin, and negatively associated with cropland and urbanization. The anthropogenic landscape provided the necessary habitats to support species such as the crawfish frog and spring peeper, which are species in need of conservation, as well as more ubiquitous species like the boreal chorus frog (*Pseudacris maculata*). Management of habitats within an anthropogenic landscape can support current and future anuran communities, including imperiled species.

INTRODUCTION

Amphibian populations are declining globally, including species that are locally common. For example, approximately 33% of anurans are currently considered threatened by the International Union for Conservation of Nature and Natural Resources (Adams et al., 2013; Bishop et al., 2012; IUCN, 2022). While the severity and specific mechanisms affecting amphibian populations vary across species and regions (Campbell Grant et al., 2020; Cushman, 2006; Gallant et al., 2007), anuran populations are greatly impacted by the loss of both the aquatic and terrestrial habitats used throughout their life cycle (Knutson et al., 1999). The loss and fragmentation of wetlands across the landscape has altered species composition, especially limiting species with low dispersal capabilities (Brodman, 2008; Gibbs, 2000). Human influence on the landscape has been another important cause of amphibian declines in wetland habitats to the extent that 22% wetland-dependent amphibians in North America are considered threatened by the IUCN (Ramsar Convention on Wetlands, 2018). Herein, we focus on three anthropogenic disturbances upon landscapes facing amphibian populations: urbanization, row-crop agriculture, and surface mining.

Urbanization alters wetland and upland habitats through changes at both local and landscape scales (Johnson et al., 2013). For example, housing developments can have a prolonged, detrimental effect on amphibian populations due to increased pollutant exposure and increased habitat fragmentation (Gagné & Fahrig, 2010; Johnson et al., 2013; Pillsbury & Miller, 2008). Habitat fragmentation resulting from roads and buildings further isolates aquatic habitats and upland habitats that are needed to support anurans (Eigenbrod et al., 2008; Gibbs, 2000). The increase in urban sprawl increases the abundance and density of impervious surfaces, which not only can increase road mortality, but also alter can wetlands through stormwater and pollution run-off (Beebee, 2013; Johnson et al., 2013; Smallbone et al., 2011). These local and landscape changes are particularly important as global human populations become more concentrated and urbanized landscapes expand (Seto et al., 2012), yet amphibian populations continue to be understudied in urban ecosystems (Rega-Brodsky et al., 2022).

Agricultural practices also may negatively affect amphibian populations and communities. The degradation of native habitat, such as the removal of forest for row crops, can reduce anuran diversity and populations, especially as the increased use of agricultural pesticides can influence species survival (Cayuela et al., 2015; Smith et al., 2006). The strengths and directionality of these effects can vary by the intensity of the agricultural operation and the species studied, in some cases positively affecting anuran populations (Koumaris & Fahrig, 2016). In one study, the amount of cropland around wetlands positively influenced American toad (*Anaxyrus americanus*) and northern leopard frog (*Lithobat*es *pipens*) occupancy (Swanson et al., 2019). Other agricultural practices, like farm pond management, may provide more wetland habitat that is otherwise limited on the landscape, increasing anuran species richness and diversity (Swartz & Miller, 2021).

Current anthropogenic habitat disturbance like urbanization and agriculture greatly impacts anuran occupancy. However, the historic land use and cover of the landscape can be just as influential to current populations, creating a need to understand how past and present land cover is driving anuran occupancy (Piha et al., 2007). For example, the reclamation of past surface mining operations influences habitat quality as the reclamation process often changes the hydrology and vegetation of wetlands (Stiles et al., 2017). Through this reclamation process, additional breeding habitats may be created to support an amphibian community that is at least as diverse as natural wetlands (Fetting, 2014; Lannoo et al., 2009; Lannoo et al., 2014; Timm & Meretsky, 2004). Thus, the land use history can dictate future vegetative succession and how the anuran populations respond to the change in land cover.

Urban, agricultural, and post-mining landscapes are each impacting anuran communities through habitat loss and fragmentation. This study sought to provide a connection between anuran occupancy and landscape composition in a highly altered landscape and provide insight for anuran management. We used anuran call surveys and landscape metrics for five land cover types (i.e., water, grassland, cropland, forest, and built environment) to associate species occupancy with the landscape. We predicted that the native habitat types such as forest, water, and grasslands were the most important land cover type for anuran species in the area, as many of the species studied require these features for reproduction. In contrast. urbanization and agriculture should negatively impact anuran occupancy due to the resulting habitat changes and overall loss of wetland habitats. Understanding the association between anuran distributions and landscape composition could help inform conservation actions to support anurans in anthropogenic landscapes.

METHODS

Study Area

We sampled a study area spanning southern Crawford Co. and northern Cherokee Co. in southeast Kansas. These counties belong to the Cherokee lowland physiographic region, which is characterized by rolling plains with patches of riparian forests and revegetated former surface mining areas (Fig. 1.1; Kansas Geological Survey 1999). The eastern portion of study area is a part of the Spring River sub-basin, and the western portion is a part of the Neosho River basin (Fig. 1.1). This region was mined for coal and other metals from the 1850s to the 1980s, with most surface mining areas left unreclaimed to be naturally revegetated (Bailey & Hooey, 2017; Kansas Historical Society, 2013). The Kansas Department of Wildlife and Parks (KDWP) and the Kansas Department of Health and the Environment (KDHE) have been working to reclaim 14,500 acres of historic strip-mined areas, which are collectively known as the Mined Land Wildlife Area (MLWA; KDWP, 2018). The KDWP and KDHE have already reclaimed some of this land into grasslands and marshes to help improve habitat quality for wildlife, such as waterfowl and upland game birds. The remainder of the land cover on the MLWA is comprised of forest, shrub, and water, and is surrounded by agricultural and urban land uses.

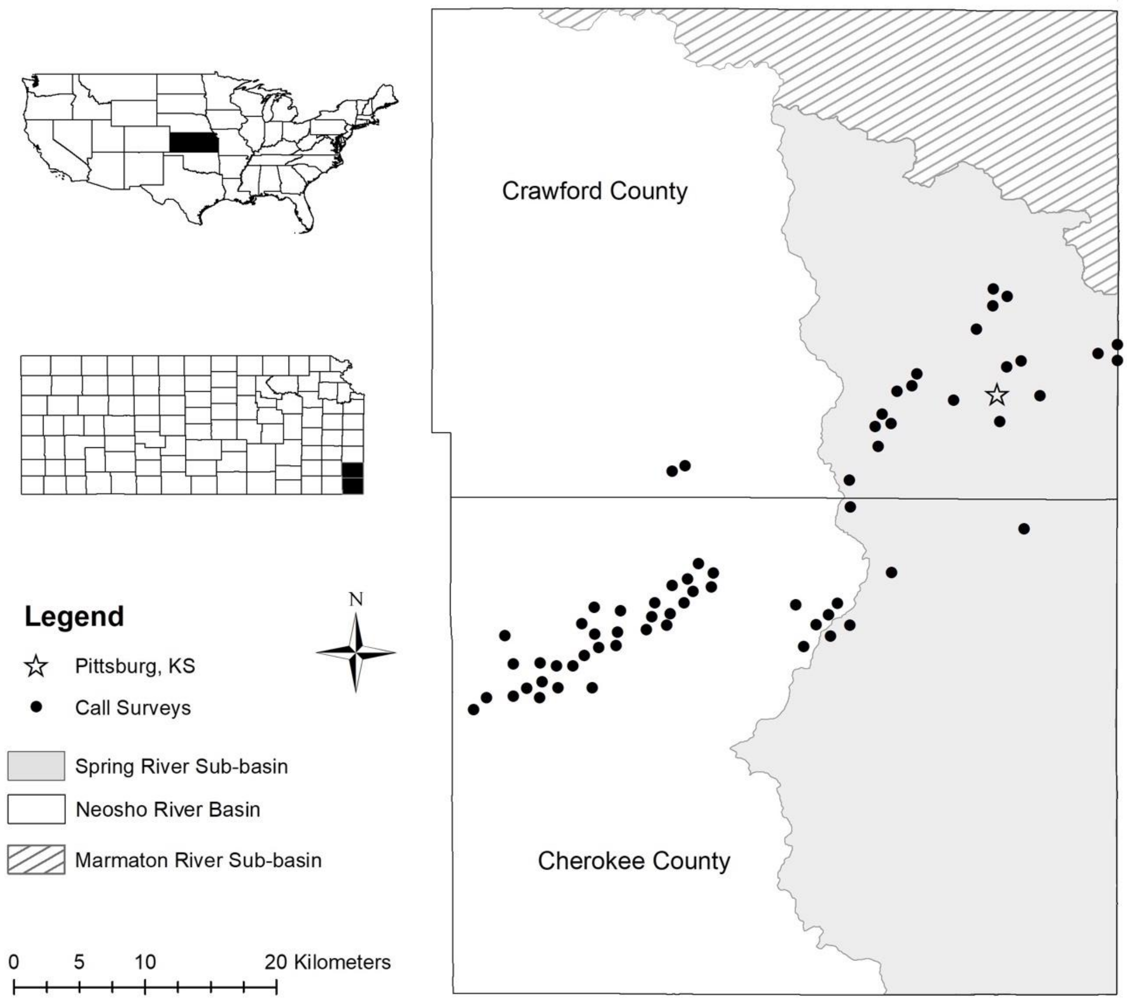

Figure 1.1. Map of the survey area with watersheds depicted. Dots represents call survey locations.

Auditory Call Surveys

We conducted anuran call surveys at 65 sites across the region on mined and non-mined lands during the 2021 and 2022 breeding seasons (Fig. 1.; Appendix I). To get the broadest coverage and ensure spatial independence of samples, we chose sites that were greater than 500 m apart and accessible from roadways or parking lots throughout the survey area, following the North American Amphibian Monitoring Program (NAAMP) protocol (Weir & Mossman, 2005). Surveys occurred twice during three different survey windows defined as mid-March to mid-April (early spring), May (spring), and June (early summer) to account for the variability of breeding times of anuran species in Kansas. Within each survey window, we sampled groups of sites in a random order and all sites were surveyed within 10 days of each other. We conducted surveys between 30 min after sunset and 0100 hrs (Weir & Mossman, 2005).

After arriving at each site, we had a 1-minute acclimation period before calls were recorded to reduce disturbance impacts on area anurans (Stevens et al., 2002). During this period, the surveyor measured detection variables including air temperature, average wind speed (Kestrel Weather Meter 2000), and percent cloud cover. Surveys were not conducted when the wind was greater than 16 kph or when there was heavy rain (Weir & Mossman, 2005). After the acclimation period, the surveyor listened for anuran calls for five minutes and recorded the strength of the chorus for every species heard (Crouch & Paton, 2002; Pierce & Gutzwiller, 2004). The surveyor determined the strength of the chorus by following index based on NAAMP: 1= individuals can be counted with a space between calls, 2= calls of individuals can be distinguished with some overlapping of calls, 3= full chorus with constant, continuous, and overlapping calls (Weir & Mossman, 2005). Surveyors also recorded the presence of other noise events (e.g. trains, passing cars, braking dogs, and people talking) that may have inhibited detection based on the NAAMP noise scale (Weir & Mossman, 2005). All surveyors were trained prior to data collection to ensure consistent and accurate aural data collection.

Land Cover Data Collection

We used the most recent National Wetland Inventory (NWI) and National Land Cover Data (NLCD) to collect proportions of land cover types within 500-m buffers around each survey location. We chose a 500-m buffer because it is considered within in the range of core habitat and average known dispersal distance of anuran species (Eigenbrod et al., 2008; Semlitsch & Bodie, 2003). To simplify land cover types, NWI was reclassified into two water body types and NLCD was reclassified into five land cover types within Program R, version 1.3.1073 (Table 1.1;R Core Team, 2020). Much of the MLWA was classified as woody wetlands, but the MLWA is primary a terrestrial habitat with distinct waterbodies, instead of trees in standing water. Therefore, we reclassified woody wetlands as forest to represent the true forest cover more accurately in the area using the NLCD data and used NWI data to assess all distinct aquatic habitats that may have been lost with reclassification. We included bare ground within urban land cover since this land cover type in this region was a result of human manipulations. Grassland land cover included pastures because they likely function as grasslands in the study area for amphibian populations. We used package “raster” to obtain the class percent from the landcover data created from the reclassified NWI and NLCD data with a 500-m circular buffer (Hijmans, 2022). These percentages were z-transformed prior to use in statistical analysis.

Table 1.1. Reclassification of the National Wetland Inventory (NWI) into two water body types and the National Landcover Database (NLCD) into five landcover types (Dewitz, 2021; U.S. Fish & Wildlife Service, 2022).

Landcover Category	Cover Types included in NWI or NLCD
NWI	
Open Water	Lake
	Freshwater Pond
	Riverine
Wetland	Freshwater Emergent Wetland
	Freshwater Forested/Shrub Wetland
NLCD	
Forest	Deciduous Forest
	Evergreen Forest
	Mixed Forest
	Shrub/Scrub
	Woody Wetlands
Water	Open water
	Emergent Herbaceous Wetlands
Built	Developed, Open Space
	Developed, Medium Intensity
	Developed, High Intensity
	Barren Land
Grass	Grassland/Herbaceous
	Paster/Hay
Crop	Cultivated Crops

Data analysis

We used single-season occupancy models to determine how landscape composition and wetland types around the survey point affected occupancy for each species using the package “unmarked” (Fiske & Chandler, 2011; Weir et al., 2005, 2014; Weir & Mossman, 2005). Species detected on >90% of sites were excluded from analyses because they lacked sufficient variability for modeling occupancy. Before fitting models, we tested covariates for multicollinearity and only included variables with $r < 0.7$

within the same models. The sampling window for each species reflected their average call phenology window in Kansas (Taggart, 2022).

We fitted models using presence-absence data in a stepwise process starting with detection probability using detection covariates. We used an additive approach to determine one or two variables that influence detection probability. We used Akaike's information criterion corrected for small sample size (AIC_c) to determine which models were supported by the data ($\Delta AIC_c<2$; Hurvich & Tsai, 1989). Models for detection that were supported were used in modeling for occupancy. We then created models that estimated the probability of occupancy using the occupancy covariates (Table 1.2). We used an additive approach to determine one to three influential variables based on AIC_c and model weight. We tested for overdispersion and examined goodness of fit to assess the overall fit of the best model (MacKenzie & Bailey, 2004). We concluded the modeling procedure for gray treefrog after the addition of one occupancy variable due to the lack of convergence.

Table 1.2. Variables used as detection and occupancy covariates in occupancy models of anuran species surveyed in southeast Kansas. All occupancy variables represent measurements from a 500-m buffer around the survey point except watershed, which was based on the specific survey point.

Model Parameter	Description
Detection Covariates	
day	The ordinal date of survey
year	Survey year: 2021, 2022
time	Minutes past sunset, calculated as the difference in sunset time and survey start time
cloud	Estimated percent of cloud cover at time and site of survey
noise	Ambient noise level based on NAAMP index
obs	Observer conducting the survey
Occupancy Covariates	
year	Survey year: 2021,2022
watershed	Watershed survey occurred in (Spring River sub-basin or Neosho River basin)
water	Proportion of open water, based on National Wetland Inventory (1985)
wetland	Proportion of wetlands, based on National Wetland Inventory (1985)
forest	Proportion of forest, based on National Land Cover Database (2019)
crop	Proportion of cropland, based on National Land Cover Database (2019)
grass	Proportion of grassland, based on National Land Cover Database (2019)
built	Proportion of built environment, based on National Land Cover Database (2019)

RESULTS

Our surveys resulted in the detection of nine anuran species: Blanchard's cricket frog (*Acris blanchardi*, detected at 100% of surveyed sites), American toad (98%), boreal chorus frog (*Pseudacris maculata*, 98%), Cope's gray treefrog (*Hyla chrysoscelis*, 98%), southern leopard frog (*Lithobates sphenocephalus*, 98%), American bullfrog (*Lithobates catesbeianus*, 89%), spring peeper (46%), crawfish frog (40%), gray treefrog (*Hyla versicolor*, 38%; Appendix II). Detection probability for all four modeled anuran species included ordinal day along with an addition variable (Table 1.3). Detection probability of American bullfrog, crawfish frog, and gray treefrog was also influenced by the observer, while detection probability of spring peepers was also explained by the ambient noise levels (Table 1.3).

The best supported model for American bullfrog occupancy included the amount of open water and built environment; open water and the built environment increased with the likelihood of occupancy (Table 1.3; Fig. 1.2; Appendix III). The best supported model for crawfish frog occupancy included the proportion of cropland, watershed, and year (Table 1.3; Appendix IV). Crawfish frogs were more likely to occupy a site with greater cropland coverage within 500 m, if the site was within the Spring River sub-basin, and in the second year (Table 1.4; Fig. 1.3). The best supported model for gray treefrog occupancy included the amount of grassland cover (Table 1.3; Appendix V). However, the estimated coefficient for grassland cover included zero, limiting the strength of the inferred relationship (Table 1.4; Fig. 1.4). The best supported model for spring peeper occupancy included watershed and the proportion of built environment and cropland (Table 1.3; Appendix VI). Spring peepers were more likely to occupy sites within the

Spring River sub-basin and with a small proportion of built and cropland cover types within 500 m (Table 1.4; Fig. 1.5).

The map derived from top model for crawfish frogs predicted occurrence throughout a large percentage of the survey area (Fig. 1.6B). The areas with limited occurrence were primarily tied to the MLWA and urban centers like Pittsburg, KS. The map derived from the top model for spring peepers predicted occurrence was more limited across the survey area (Fig. 1.6C). The areas with the highest likelihood of occurrence were primarily within historical surface mined areas through the center of the survey area.

Table 1.3. Top-ranked (ΔAIC_c <2) occupancy models estimating the probability that American bullfrogs (*Lithobates catesbeianus*), crawfish frogs (*Lithobates areolatus*), gray treefrogs (*Hyla versicolor*), and spring peepers (*Pseudacris crucifer*) occupied a call site during 2021 and 2022 in southeast Kansas. Null models have also been included, along with each models' parameters (K) and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
American bullfrog			
p(day + obs) ψ(water + built)	8	0	0.92
p(day + obs) ψ(.)	6	32.35	0
Crawfish frog			
p(day + obs) ψ(crop + watershed + year)	9	0	0.65
p(day + obs) ψ(.)	6	9.23	0.01
Gray treefrog			
p(day + obs) ψ(grass)	7	0	0.30
p(day + obs) ψ(built)	7	0.42	0.24
p(day + obs) ψ(wetland)	7	1.65	0.13
p(day + obs) ψ(.)	6	2.76	0.08
Spring peepers			
p(day + noise) ψ(watershed + built + crop)	10	0	0.83
p(day + noise) ψ(.)	7	97.95	0

Table 1.4. Estimates of each occupancy parameter with the top occupancy model for American bullfrogs (*Lithobates catesbeianus*), crawfish frogs (*Lithobates areolatus*), gray treefrogs (*Hyla versicolor*), and spring peepers (*Pseudacris crucifer*), based on call surveys conducted in 2021 and 2021 in southeast Kansas. The beta estimates, standard errors (SE), and the lower and upper 95% confident interval (CI) for each parameter were included.

Species	Parameters	Estimate	SE	Lower 95% CI	Upper 95% CI
American bullfrog	intercept	0.39	0.64	-0.67	1.45
	water	1.58	0.74	0.37	2.79
	built	-0.52	0.31	-1.02	-0.01
Crawfish frog	intercept	-3.29	3.22	-8.59	2.02
	crop	2.44	1.63	-0.24	5.11
	watershed	6.33	62.01	-95.67	108.33
	year	4.00	2.92	-0.81	8.81
Gray treefrog	intercept	12.90	15.00	-11.81	37.57
	grass	10.70	11.50	-8.18	29.64
Spring peeper	intercept	-5.37	1.58	-7.97	-2.78
	watershed	7.36	1.50	4.88	9.83
	built	-1.08	0.33	-1.62	-0.53
	crop	-3.24	1.29	-5.36	-1.13

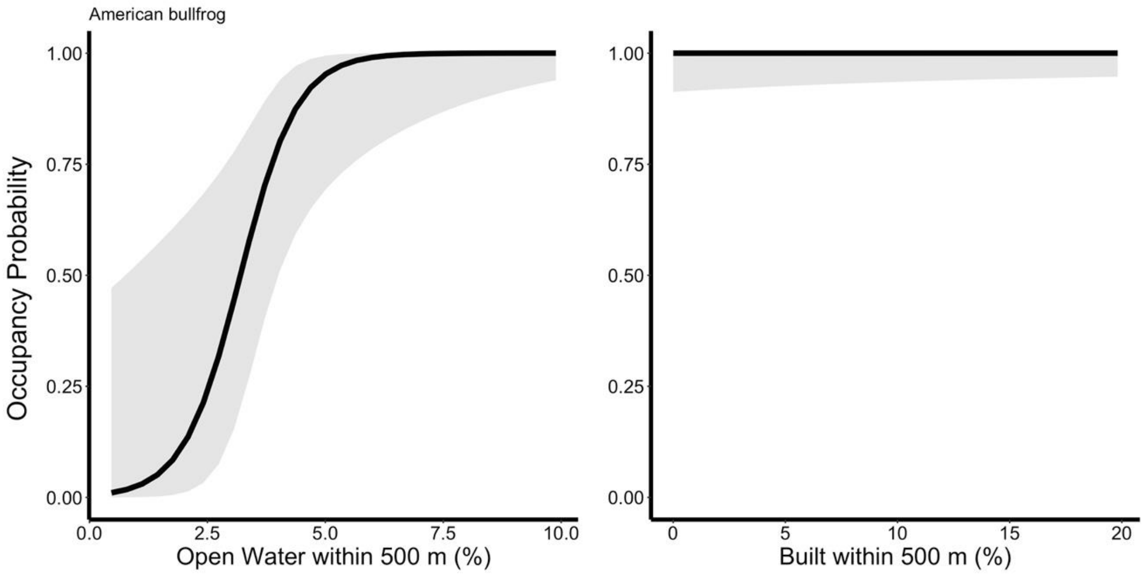

Figure 1.2. Predictive plots based on the top occupancy model for American bullfrogs (*Lithobates catesbeianus*) during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

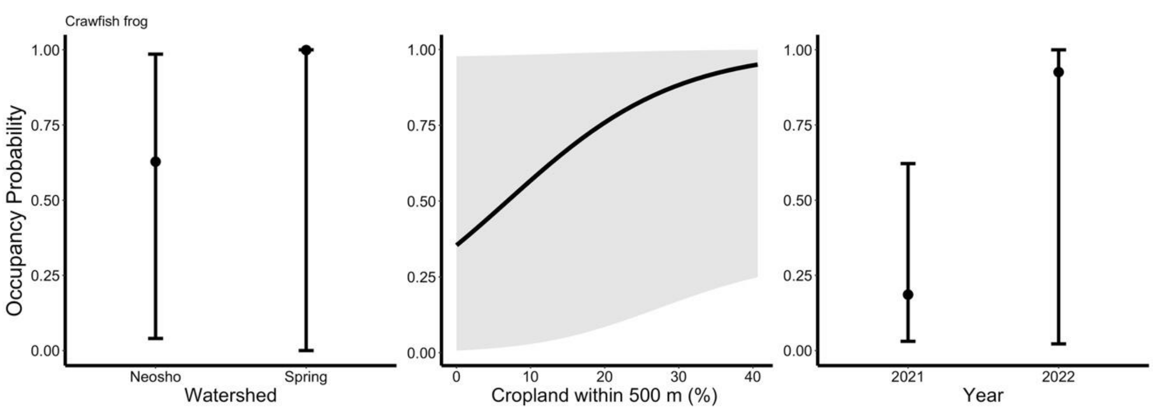

Figure 1.3. Predictive plots based on the top occupancy model for crawfish frog (*Lithobates areolatus*) during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals. Watershed was held at its intercept when making predictive plots for other variables within the top model.

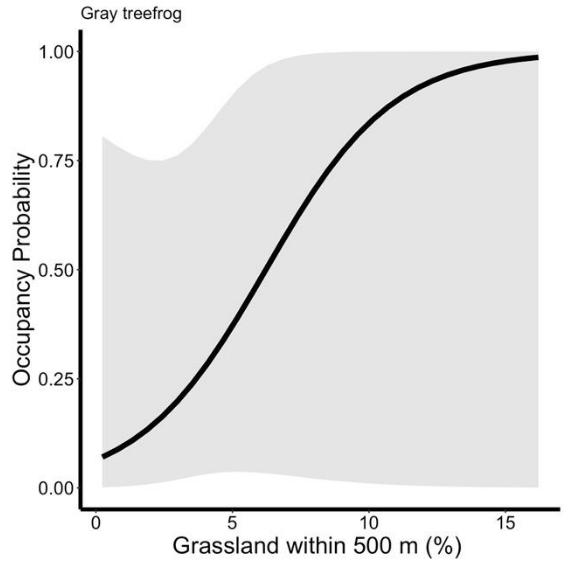

Figure 1.4. Predictive plots based on the top occupancy model for gray treefrogs (*Hyla versicolor*) during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

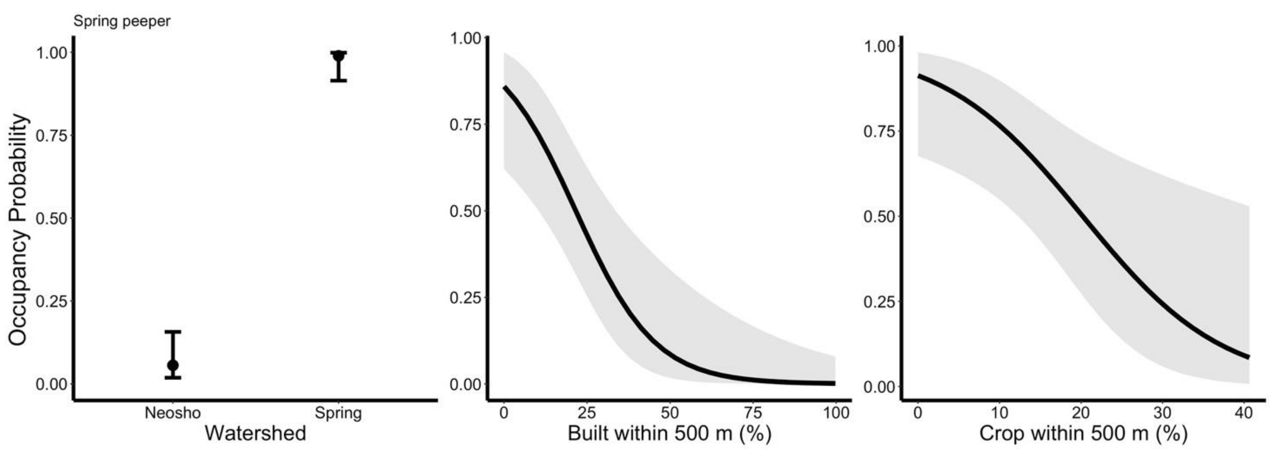

Figure 1.5. Predictive plots based on the top occupancy model for spring peeper (*Pseudacris crucifer*) during breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

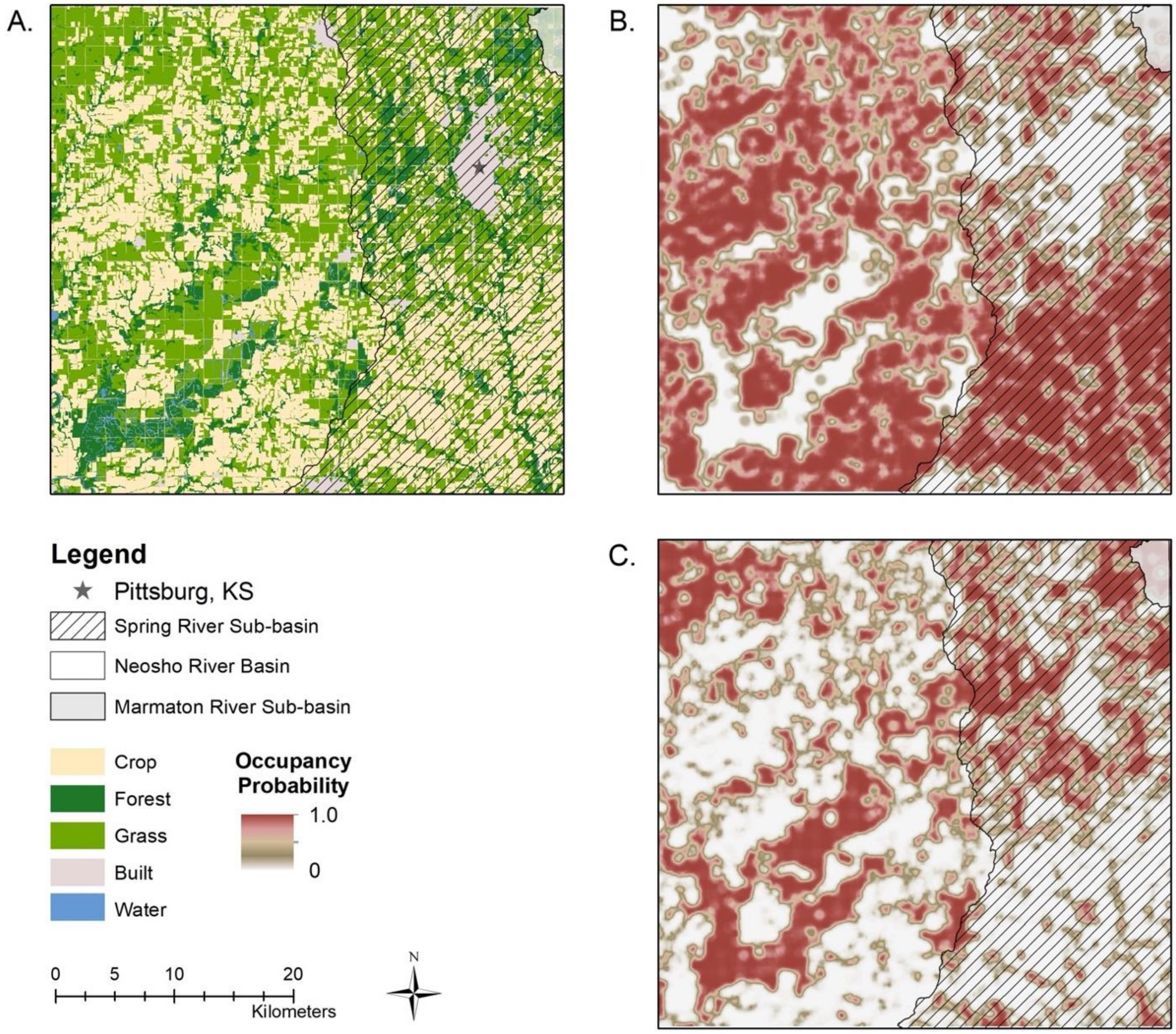

Figure 1.6. Land cover categories reclassified from the NLCD and associated watersheds in southeast Kansas A), and the resulting probability of SINC species occupancy, B) crawfish frog (*Lithobates areolatus*) and C) spring peeper (*Pseudacris crucifer*), during the breeding season in 2021 and 2022. Occupancy plots were created based on the best model for single-species occupancy models. Categorical variables were held at their respective intercepts.

DISCUSSION

In a highly fragmented and disturbed landscape, we found that occupancy of anuran species depended on the specific habitat needs of each species rather than one prevailing habitat disturbance or land cover feature. Even though the survey area has been highly impacted by anthropogenic changes like surface mining, agriculture, and urbanization, the landscape provided aquatic and terrestrial habitats that are necessary to support populations of crawfish frogs and spring peepers, both SINC species in Kansas, as well as American bullfrogs and gray treefrogs (Rohweder, 2015). Many anuran species were nearly ubiquitous in this area, suggesting that this altered landscape provides the appropriate habitats to support common anuran species.

Survey locations were split between the Spring River sub-basin and the Neosho River basin. Although there were no apparent defining landscape features that separated these watersheds, both SINC species were more likely to occupy the Spring River sub-basin. Spring peepers and crawfish frogs may have been found throughout this watershed because of the proximity to source populations to the east in Missouri, or because the region provided sufficient habitats allowing these species to be supported at the western edge of their distributional range.

Anthropogenic changes to this landscape have created a mosaic of interspersed cropland, forest, water, grassland, and impervious surfaces. In much of this area, cropland was one of the few places that still had deep topsoils, which crawfish frogs have been known to rely upon (Busby & Brecheisen, 1997). The mined lands were heavily disturbed with rocky tillage and minimal to no topsoil, which may have limited crawfish frogs' use or association with other habitats on the mined lands areas. Spring peepers had a strong negative association with cropland, likely because of their breeding habitat preferences of tree cover and temporary pools, which crop fields often cannot provide (Collins & Fahrig, 2017; Swanson et al., 2019).

The dramatic changes in soil structure associated with strip mining, and subsequent successional stages, have created waterbodies with varying hydroperiods and habitats. These habitats may not have existed in the absence of surface mining operations (Lannoo et al., 2009; Lannoo et al., 2014). The large open bodies of water, like the more permanent strip pits on mined lands or agricultural ponds, provided more American bullfrog breeding habitats, likely resulting in their strong association with the amount of open water on the landscape (Koumaris & Fahrig, 2016).

Spring peepers and gray treefrogs have been found to have variable breeding habitats, potentially making the amount of open water or wetlands less important at landscape scale relative to local scales, but allowed for these species to breed in the study area (Babbitt et al., 2003). Even though gray treefrog had percent wetland in the top model, this variable was not meaningful and did not explain their occupancy in this area. Cope’s gray and gray treefrogs are notoriously difficult to distinguish by ear, especially as they often occupy similar habitats (Gerhardt, 2005). The inclusion of observer as a detection covariate suggests that there may be uncertainty of detection due to observer and competing calls, resulting in the uncertainty between specific landscape variables like percent of wetlands and gray treefrog occupancy.

Urbanization has been shown to negatively affect amphibian populations because of reduced aquatic and terrestrial habitat quality and availability (Rubbo & Kiesecker, 2005). In this study, the built environment included all impervious surfaces, primarily roads and urban development in the area. Spring peepers had a strong negative association with the built environment. Other studies have found that urbanization most often impacts amphibian species that require shallower, fishless ponds for breeding habitat, such as spring peepers (Hamer & Parris, 2011; Rubbo & Kiesecker, 2005). Species like American bullfrogs may, however, be more suited for urbanization, as urban ponds often provide the habitat needed for aquatic species that prefer permanent hydroperiods (Sauer et al., 2022). The negative association between spring peepers and the built environment was also likely driven by the limited dispersal capabilities within an urban environment, due to the increased amount of roads (Eigenbrod et al., 2008; Pillsbury & Miller, 2008). Although urbanization increases the noise and light at night which might decrease the detection of anuran species, urbanization has not been shown to decrease the occupancy of the area. Therefore association of spring peepers with the built environment is likely tied to availability of breeding habitats within urban areas (Cronin et al., 2022). Although this region's urban areas had relatively low population density (i.e., < 39,110 POP), it was likely impacting spring peepers in a similar way to larger population centers resulting in the strong negative association with the built landscape cover.

The native terrestrial habitats, like forest and grassland cover, did not have strong relationships to anuran occupancy in this study. Although gray treefrog had grassland in the top model, grassland cover showed a weak relationship to gray treefrog occupancy. Even so, the lack of support in our models for forest and grassland land cover does not indicate the lack of importance of these habitats on the landscape, as many studies with fewer anthropogenic changes to the landscape show relationships between species occupancy and forest and grassland cover. Forest is largely considered important for spring peepers and gray treefrogs, as this land cover type was often the preferred breeding habitat for these species (Collins & Fahrig, 2017; Eigenbrod et al., 2008; Knutson et al., 1999; Simpson et al., 2021). This pattern can be seen in the predictive plot for spring peepers; the predicted occupancy is highest on forested areas (Fig. 1.6A-B). As for grasslands, we classified hay fields and pastures as grassland land cover, which historically would have been tallgrass prairie in southeast Kansas. Anuran populations have been positively associated or have a neutral association with livestock and pasture land cover likely due to the lower intensity agricultural practices like pasture rotations or no-till row crops, which could be the driving force behind the predicted occupancy of crawfish frogs being in the cropland and grassland patches (Fig. 1.6; Howell et al., 2019; Koumaris & Fahrig, 2016).

Continued research is needed on anthropogenically altered landscapes to understand to a fuller extent how the landscape composition is influencing anuran populations, as some of the species' results had high levels of uncertainty. Our surveys were based on the MLWA to study the impacts of remnant strip mined areas, but most of this region has been affected by mining. Therefore, all land cover types are impacted. However, the addition of call sites not directly related to the MLWA would provide a clearer picture of how historic mining in the region influenced anuran occupancy, even for the species that were considered ubiquitous in this area. Additionally, modeling various landscape metrics like mean patch size, may provide a deeper understanding how the landscape mosaic is influencing anuran occupancy. The use of acoustic detectors could allow for a more accurate representation of gray treefrog occupancy on the landscape and provide information about differences in habitat preference between Cope's gray and gray treefrogs.

In addition, focused research should target the SINC species in the area to further understand their relationship to landscape composition and land use. Spring peepers appeared on this landscape after most of the mining activity had concluded, with the first report of individuals in Cherokee Co. in 1951 (Rundquist, 1977) and in Crawford Co. in 2000 (Collins, 2001), suggesting that the land cover changes since mining have provided appropriate habitat for them to colonize the area. The mined lands continue to the west of spring peeper's current range limit; thus, research could address the potential for spring peepers to extend their range. Examining the underlining causes for the association of crawfish frogs and croplands such as the amount of topsoil or connectivity to breeding ponds, would allow for a better understanding of habitat use in this anthropogenic landscape to provide support for conservation efforts.

CONCLUSION

Anthropogenetic changes to a landscape impact anuran occupancy in a variety of different ways. Even so, the variation and diversity in habitat types resulting from these changes may provide sufficient habitats to support anuran populations and communities. Due to the unique land use and mining history of this region, the availability of habitats such as forests, grasslands, open water, and wetlands, supports a variety of anuran populations, including SINC species. The management of aquatic and terrestrial habitats across all anthropogenetic landcover types will support current and future anuran population.

CHAPTER II

POST-MINED WETLANDS PROVIDE BREEDING HABITAT FOR AMPHIBIANS

ABSTRACT

Wetlands are complex, threatened ecosystems that have frequently become degraded over time. Post-mined landscapes can provide an increased number of wetlands, but little is known about the health of these wetlands on mined sites that have only been altered by vegetative succession, i.e., they have never been deliberately reclaimed. Amphibian persistence in wetlands in heavily disturbed ecosystems can help to determine the quality of habitat for amphibians and other wetland dependent species. This study aimed to describe the wetland characteristics that influence amphibian community composition and occupancy of individual species. Single species occupancy models were used to determine the wetland characteristics that influenced larval presence of five common species, including American bullfrog, (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), gray treefrog species complex (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*). The response of the amphibian community (i.e., richness, diversity, composition) to wetland features was examined through linear models and non-metric multidimensional scaling (NMDS). Occupancy for each species varied, but the presence of predatory fish, hydroperiod, and emergent vegetation cover were the most influential

predictors of occupancy. Amphibian richness and diversity were influenced by the water conductivity level, the presence of predatory fish, hydroperiod, and emergent vegetation cover within the wetland. The NMDS showed that amphibian community composition was similar among wetlands regardless of the mining history or management. While species' occupancy patterns varied, the wetlands across the post-mined landscape provided sufficient habitat to support a diverse amphibian community. Increasing the variation in wetlands through protection, reclamation, and management could allow these amphibians and other wetland-dependent species to persist on the landscape.

INTRODUCTION

Wetlands are considered a threatened ecosystem globally, with 35% of wetlands lost since 1970 and even more degraded due to human disturbances, including changes in agriculture, urbanization, and surface mining (Dahl, 1990; Ramsar Convention on Wetlands, 2018). The degradation of wetlands across North America impacts the majority of amphibians that are dependent on wetland habitats (Church et al., 2008). Additionally, amphibians provide critical functions to ecosystems, such as the efficient transfer of biomass and nutrients between habitats (Burton & Likens, 1975; Hopkins, 2007; Semlitsch et al., 2014). Amphibians can also be used as water quality indicators due to their semipermeable skin; thus, they can provide habitat quality assessments for entire vertebrate communities (Boyer & Grue, 1995; Pollet & Bendell-Young, 2000). All of these features make amphibians useful in examining wetland health after disturbances such as surface mining.

Surface mining alters landscapes, destroys habitat, and disrupts ecosystem function by removing the top layer of earth to access mineral seams. In the decades following the Surface Mining Control and Reclamation Act (1997), mining companies have been required to reclaim disturbed mined areas with native habitats, with the goal of creating wetland habitats in and around the strip pits or deep rectangular pits left after mining activity. Multiple studies have assessed amphibian communities that use reclaimed wetlands on surface mined lands and found that natural and reclaimed wetlands had similar amphibian communities (Lannoo et al., 2014; Pollet & Bendell-Young, 2000; Sasaki et al., 2015; Stiles et al., 2017). This suggests that reclamation has created habitats that function similarly to natural wetland systems. However, most of previous studies focused on mined lands that were reclaimed in the years immediately following mining operations. Yet a large portion of surface mined lands were never reclaimed or were not reclaimed for decades after mining ended, resulting in vegetative succession.

Wetlands are complex ecosystems that have many biotic and abiotic factors that may influence the quality of breeding habitat for amphibians, like water quality, hydroperiod, vegetation, and predators. Water quality variables, such as water temperature, conductivity, dissolved oxygen (DO), and pH, have affected amphibian communities (Chambers, 2011; Karraker et al., 2008). For example, species richness responded positively to DO in an urban area, while pH can have either positive or negative effects, depending on the context (Brodman et al., 2003; Calderon et al., 2019; Camacho-Rozo & Urbina-Cardona, 2021). Conductivity is an important factor in heavily mined areas, as mining may introduce heavy metals and salts that increases conductivity, which can decrease the survival of amphibians (Chambers, 2011). Hydrologic conditions may be the most influential factor when creating wetlands, particularly the variety of hydroperiods that can support the large breeding populations of some amphibian species

(Brodman et al., 2003; Collinge et al., 2013; Nagel et al., 2021). Hydroperiod often drives other factors that influence breeding success, like the vegetation density and presence of predatory fish (Amburgey et al., 2014; Babbitt et al., 2003; Brodman, 2008). Aquatic vegetation can positively influence amphibian communities through the addition of microhabitats (Burne & Griffin, 2005; Hamer & Parris, 2011). Amphibian communities may also be negatively associated with the size class and density of predatory fish due to the increased predation rate of amphibian egg masses and larvae (Hartel et al., 2007; Kloskowski, 2009).

The present study aimed to identify wetland characteristics that affect the occupancy of five common amphibian species, which can be used as wetland indicators for the monitoring and management of the amphibian community. In addition, this study aimed to understand the biotic and abiotic characteristics of wetlands that influence amphibian communities (i.e., diversity and structure) at wetlands with a variety of mining histories and management activities. Information about the amphibian use of wetlands on previously mined lands can be used to determine conservation value and guide management practices of these disturbed systems to promote an ecosystem that supports a wide variety of biota.

METHODS

Study Area

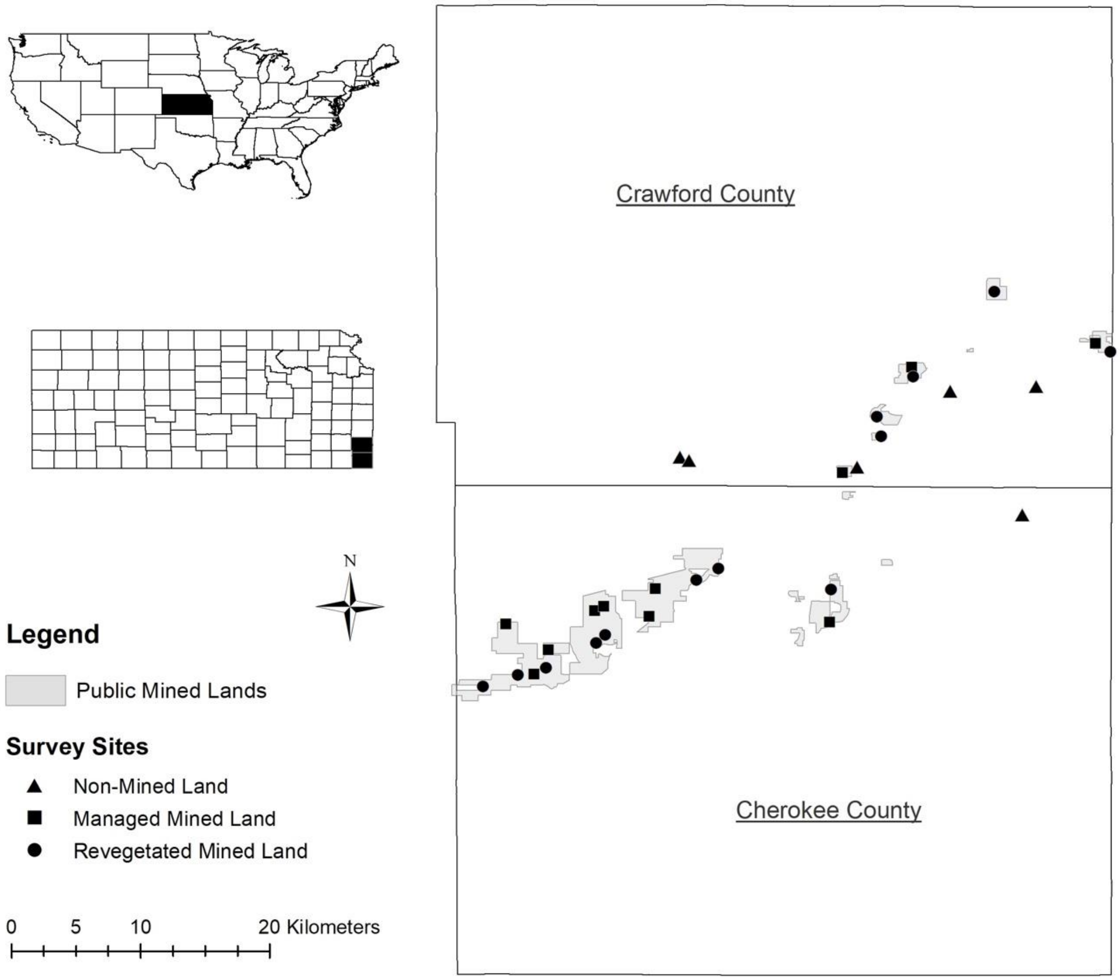

Figure 2.1. Map of the survey area with sites indicated by the mining and reclamation status of each wetland in southeast Kansas. Public mined lands in the area are shaded, including Mined Land Wildlife Area and Southeast Kansas Biological Station.

We sampled an area spanning southern Crawford Co. and northern Cherokee Co. in southeast Kansas. These counties are part of the Cherokee Lowlands physiographic region, which is characterized by rolling plains that have patches of forest along streams and on the abandoned mining areas (Fig. 2.1; Kansas Geological Survey 1999). This region was mined for coal and other metals from the 1850s to the 1980s, with most areas left unreclaimed, to be naturally revegetated (Bailey & Hooey, 2017; Kansas Historical Society, 2013). The Kansas Department of Wildlife and Parks (KDWP) has been working to reclaim 14,500 acres of historic strip-mined areas, which are collectively known as the Mined Land Wildlife Area (MLWA; KDPW, 2018). The KDWP has already converted some of this land into grasslands and marshes to help improve habitat quality for wildlife, such as waterfowl and Northern Bobwhite (*Colinus virginianus*). We surveyed wetlands within the MLWAs, the Southeast Kansas Biological Station, Buche Wildlife Area, and private properties, which included revegetated, managed, and non-mined sites.

Survey Methods

We chose 31 wetlands to survey across the study area that had varied mining histories and management activities (Fig. 2.1, Appendix VII). For the purpose of this paper, wetlands were categorized as revegetated (i.e., shallower [< 5m deep] wetlands on mined lands that have not been manually altered since mining), managed (i.e., wetlands on mined lands that were created or maintained by vegetation control and water-level manipulation), or non-mined (i.e., wetlands with no mining history whether actively managed or not). Of the selected wetlands, 13 were revegetated, 11 managed, and seven non-mined. Revegetated wetlands were chosen based on the accessibility of the waterline (i.e., low vegetation density and shallow slope). Most of the managed wetlands had been modified to support waterfowl hunting and were considered marshes by KDWP. Sites were located at least 300 m from each other and were scattered across the extent of the survey area (Fig. 2.1, Appendix VII). Two different wetlands at the Buche Wildlife Area were surveyed even though they were within 300 m of each other, but each was only surveyed in a single year.

To sample the amphibian community across varying breeding periods, we used minnow trap and dipnetting surveys for two consecutive days during three different survey windows defined as mid-March to mid-April (Early Spring), May (Spring), and June (Summer). We evenly spaced sampling locations along the shore from a random center point in either direction. We scaled distance between sampling locations by wetland size, ranging from 5 m between samples (wetlands < 0.05 ha), 10 m (wetlands between 0.05 ha to 0.35 ha), and 20 m (wetlands > 0.35 ha). Sampling locations alternated between minnow trap (four locations) and dipnetting (four locations) for a total of eight sampling locations per wetland. For each survey window, we adjusted the sampling locations within sites to account for seasonal fluctuations in the water line. Within each survey window, traps and dipnet locations were the same for both consecutive days.

We modified minnow traps (Gee's Galvanized Wire Minnow Trap) with window screening and baited each trap for a random trap night per survey window with a green glow sticks to increase catch rate and potentially attract eastern newts (*Notophthalmus viridescens*), a species in need of conservation (SINC) in Kansas (Bennett et al., 2012; Grayson & Roe, 2007; Swartz & Miller, 2018). We placed traps at varying distances from shore to have the funnel entrance at least half covered with water while also having a portion of the trap exposed to the air for air-breathing species. At each dipnetting location, we extended the dipnet approximately 1 m from the waterline into the wetland and quickly pulled toward the shore along the bottom following a zig-zag motion (Babbitt et al., 2003). We conducted both surveys within a 24-hr period. We identified all invertebrates and vertebrates in samples based on field markings and recorded the total number captured for each trap and dipnet location before organisms were returned to the wetland. When species of fish were captured, we identified them to species level. Amphibians were identified based on field marks. Therefore, we couldn't distinguish among Cope's gray treefrog (*Hyla chrysoscelis*) and gray treefrog (*Hyla versicolor*) larvae and will hereafter refer to the gray treefrog complex as *Hyla* spp.

Wetland Characteristics

We recorded the area of the wetland with a Garmin eTrex 10 GPS unit. The surveyor walked the perimeter of the wetland and we calculated the wetland area based on the standing water line in Google Earth Pro (*Google Earth Pro*, 2022). We recorded the area with this method during each survey period unless the water level was not observably different. We considered the change in area across the three sample periods to be a proxy for hydroperiod. We sampled water quality once during each survey window. Water quality sampling included pH (HI 9812-5 Portable Meter), conductivity (µs/cm; HI 9812-5 Portable Meter; Babbitt et al., 2003), water temperature (ºC; YSI ProODO), and dissolved oxygen (DO; mg/L; YSI ProODO). During 2022 we replaced the water temperature and dissolved oxygen meter with an ExStik DO600 (Extech Instruments). Conductivity was later grouped into three categories (low < 500 µs/cm, medium = 500–1499 µs/cm, and high ≥1500 µs/cm), as some wetlands exceeded the range of the meter.

During each water quality sample, we recorded measurements from three random dipnet or trap locations chosen prior to sampling day, and all samples were taken approximately 1 m from the waterline (Babbitt et al., 2003). We visually estimated the percent cover of emergent vegetation within the wetland (Burne & Griffin, 2005). We also recorded the presence of predatory fish species, such as bass (*Micropterus* spp.), sunfish (*Lepomis* spp.), and gar (*Lepisosteus* sp.). The presence of these fish species was categorized at never, sometimes, or always based on captures through dipnetting and traps, and opportunistic sightings. The category of "sometimes" refers to sites that had a change in the presence of predatory fish between survey seasons, primarily due to flooding events connecting the wetland with a fish source, such as a nearby streams or other strip pit wetlands.

Data Analysis

We used an information theoretic approach to analyze the effects of wetland characteristics on individual species occupancy and wetland communities based on larval captures within each year (Burnham & Anderson, 2004). Before fitting models, we tested covariates for multicollinearity and excluded variables with $r > 0.7$ from the same models. We used Akaike's information criterion corrected for small sample size (AIC_c) to determine what models were supported by the data ($\Delta AIC_c < 2$).

We used the package "unmarked" in Program R (version 1.3.1073) to fit single-season occupancy models for five of the most common species found: American bullfrog, (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), *Hyla* spp., and southern leopard frog (*Lithobates sphenocephalus*). We fitted models in a stepwise process starting with detection probability using detection covariates. We used an additive approach to determine which covariates influence detection probability (Table 2.1). Detection covariates from the top models for detection were then included in all models for occupancy. We created models that estimated the probability of occupancy using the occupancy covariates (Table 2.1). We used an additive approach to determine occupancy covariates that were most influential based on AIC_c and model weights. We examined the model with the greatest number of parameters for the goodness of fit (MacKenzie & Bailey, 2004). If the model was overdispersed, we used $QAIC_c$ to compare the candidate model set.

We used the package “vegan” to analyze the larval amphibian community (Oksanen et al., 2022). We calculated Choa1 richness and Shannon diversity for each year at each site to describe the larval community diversity. We used linear models to examine the influence of wetland characteristics on the amphibian community (Table 2.1). We used an additive approach to determine the wetland characteristic variables that influenced richness and diversity. To compare the similarity of amphibian community structure between the three wetland types, we performed a non-metric multidimensional scaling (NMDS) ordination. Amphibian captures were included in a site × species matrix. We calculated the NMDS on Bray-Curtis distance matrices derived from a Wisconsin square root transformed capture numbers. We evaluated the stress to decide the number of ordination dimensions.

Table 2.1. Wetland characteristics included as covariates in liner models of amphibian species richness and diversity in southeast Kansas during 2021 and 2022. Superscripts indicate the use of the parameter for single species occupancy models for American bullfrog (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), Hyla spp. (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*). Temperature and day were only included in single species occupancy models. Parameters were averaged for each site within a single year of surveys.

Model Parameter	Description
day†	Ordinal date of survey
year$^{\dagger *}$	Year the survey took place (2021, 2022)
type$^{\dagger *}$	Classification of the wetland based on the history of the site (managed, non-mined, revegetated)
temp†	Average water temperature (^{o}C)
pH*	Average pH of the wetland
DO*	Average dissolved oxygen of the wetland (mg/L)
cond*	Conductivity level based on average readings: low <500 μs/cm, medium = 500 – 1499 μs/cm, and high ≥1500 μs/cm
area*	Average area of the site (ha)
hydro*	Percent change in wetland area over the year as a relative proxy for hydroperiod
emveg*	Average percent cover of emergent vegetation in the wetland (%)
fish*	Presence of predatory fish (bass and sunfish; 0 = no presence, 1 = sometimes present, 2 = always present)

† Parameter used as a detection covariate for single species occupancy models
* Parameter used as an occupancy covariate for single species occupancy models

Results

We detected 10 amphibian species across the 31 sites in 2021 and 2022 (Appendix VIII). Most notably, three SINC species, crawfish frog (*Lithobates areolatus*), eastern newt, and spring peeper (*Pseudacris crucifer*), were captured at 9.7%, 6.4%, and 19.4% of the sites, respectively. Other species captured include American bullfrog (61.3% of sites), American toad (*Anaxyrus americanus*, 22.6%), Blanchard's cricket frog (90.3%), boreal chorus frog (41.9%), *Hyla* spp. (41.9%), smallmouth salamander (*Ambystoma texanum*, 22.6%), and southern leopard frog (83.9%). Richness estimation for sites in each year ranged from zero to nine, with a mean species richness of 3.37. Shannon Diversity for sites based on each year ranged from zero to 1.67 with a mean of 0.56.

Wetland occupancy varied by species. The best supported model for American bullfrog occupancy was the intercept-only for occupancy with site type affecting detection probability (Table 2.2; Appendix IX). The hydroperiod best explained occupancy for Blanchard's cricket frog (Table 2.2, Appendix X); the greater the change in wetland area over a year, the less likely cricket frogs were to occupy the wetland (Table 2.3, Fig. 2.2A). Boreal chorus frog occupancy was best explained by the average percent cover of emergent vegetation (Table 2.2, Appendix XI). Boreal chorus frog occupancy increased with more emergent vegetation within the wetland (Table 2.3, Fig. 2.2B). Although the presence of predatory fish was the best supported model for *Hyla* spp. (Table 2.4, Appendix XII), there was not a clear pattern for occupancy because of the large confidence intervals for predatory fish (Table 2.3, Fig. 2.2C). Lastly, the best supported model for southern leopard frog occupancy was the change in area of wetland over the survey period (Table 2.2, Appendix XIII). Southern leopard frog occupancy increased with a greater change in wetland area over the year (Table 2.3, Fig. 2.2D).

The best supported model for amphibian species richness included the presence of predatory fish, conductivity level, and percent cover of emergent vegetation within the wetland (Table 2.4, Appendix XIV). Richness was greatest in wetlands that sometimes had predatory fish, had low conductivity, and those with more emergent vegetation (Table 2.5, Fig. 2.3A–C). The best supported model for amphibian diversity included conductivity, presence of predatory fish, and the change in wetland area over the year (Table 2.4, Appendix XV). Amphibian diversity was the highest in wetlands with low conductivity, and that sometimes had predatory fish, and showed a negative relationship with percent change in wetland area within a year (Table 2.5, Fig. 2.3D – F).

The first two dimensions of the NMDS ordination had a goodness of fit of 0.17, suggesting there was a fair representation of dissimilarity between wetland in reduced dimensions, but some distances may be misleading. We did not find strong evidence that amphibian community structure differed among non-mined, managed, and revegetated sites; however, the non-mined and revegetated wetlands showed some differentiation in composition from each other (Fig. 2.4A). The wetland characteristics of predatory fish presence, DO, conductivity level, wetland type, percent change in wetland area, and percent emergent vegetation within the wetland were associated with the ordination scores between the sites (Fig 2.4B).

Table 2.2. Top-ranked ($\Delta QAIC_c$ <2) occupancy models estimating the probability that American bullfrog (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), *Hyla* spp. (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*) occupied wetland sites during 2021 and 2022 in southeast Kansas. Due to overdispersion, $QAIC_c$ was used for all species except boreal chorus frog, in which case AIC_c was used. Null models have also been included, along with each model's parameters (K) and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
American bullfrog			
p(type) ψ(.)	5	0	0.17
p(.) ψ(.)	3	0.58	0.13
p(type) ψ(pH)	6	1.09	0.10
p(type) ψ(hydro)	6	1.60	0.08
p(type) ψ(emveg)	6	1.64	0.08
p(type) ψ(area)	6	1.82	0.07
Blanchard's cricket frog			
p(day) ψ(hydro)	5	0	0.36
p(day) ψ(hydro + area)	6	1.29	0.19
p(day) ψ(.)	4	2.45	0.11
p(.) ψ(.)	3	21.61	0
Boreal chorus frog			
p(day) ψ(emveg)	4	0	0.46
p(day) ψ(emveg + DO)	5	1.40	0.23
p(day) ψ(.)	3	7.33	0.01
p(.) ψ(.)	2	24.47	0
Hyla spp.			
p(day) ψ(fish)	6	0	0.23
p(day) ψ(fish + cond)	8	1.02	0.14
p(day) ψ(fish + area)	7	1.69	0.10
p(day) ψ(fish + emveg)	7	1.95	0.09
p(day) ψ(.)	4	6.04	0.01
p(.) ψ(.)	3	11.95	0

Table 2.2. Continued.

Model	K	Δ $QAIC_c$	Model Weight
Southern leopard frog			
p(type + day) ψ(hydro)	7	0	0.22
p(type + day) ψ(.)	6	0.85	0.14
p(.) ψ(.)	3	12.48	0

Table 2.3. Estimates of each occupancy parameter with the top occupancy model for Blanchard's cricket frog (*Acris blanchardi*), boreal chorus frog (*Pseudacris maculata*), *Hyla* spp. (*Hyla chrysoscelis/versicolor*), and southern leopard frog (*Lithobates sphenocephalus*), based on wetland surveys conducted in 2021 and 2021 in southeast Kansas. Beta estimates, standard errors (SE), and the lower and upper 95% confidence interval (CI) are included for each parameter.

	Parameter	Estimate	SE	Lower 95% CI	Upper 95% CI
Blanchard's cricket frog	intercept	2.99	1.23	0.96	5.01
	hydro	-3.39	1.67	-6.14	-0.64
Boreal chorus frog	intercept	-1.83	0.52	-2.69	-0.97
	emveg	0.05	0.02	0.02	0.07
Hyla spp.	intercept	0.21	0.38	-0.42	0.84
	sometimes fish	-0.61	0.70	-1.76	0.54
	always fish	-9.53	26.13	-52.51	33.44
Southern leopard frog	intercept	0.26	0.42	-0.43	0.96
	hydro	4.05	1.80	1.09	7.01

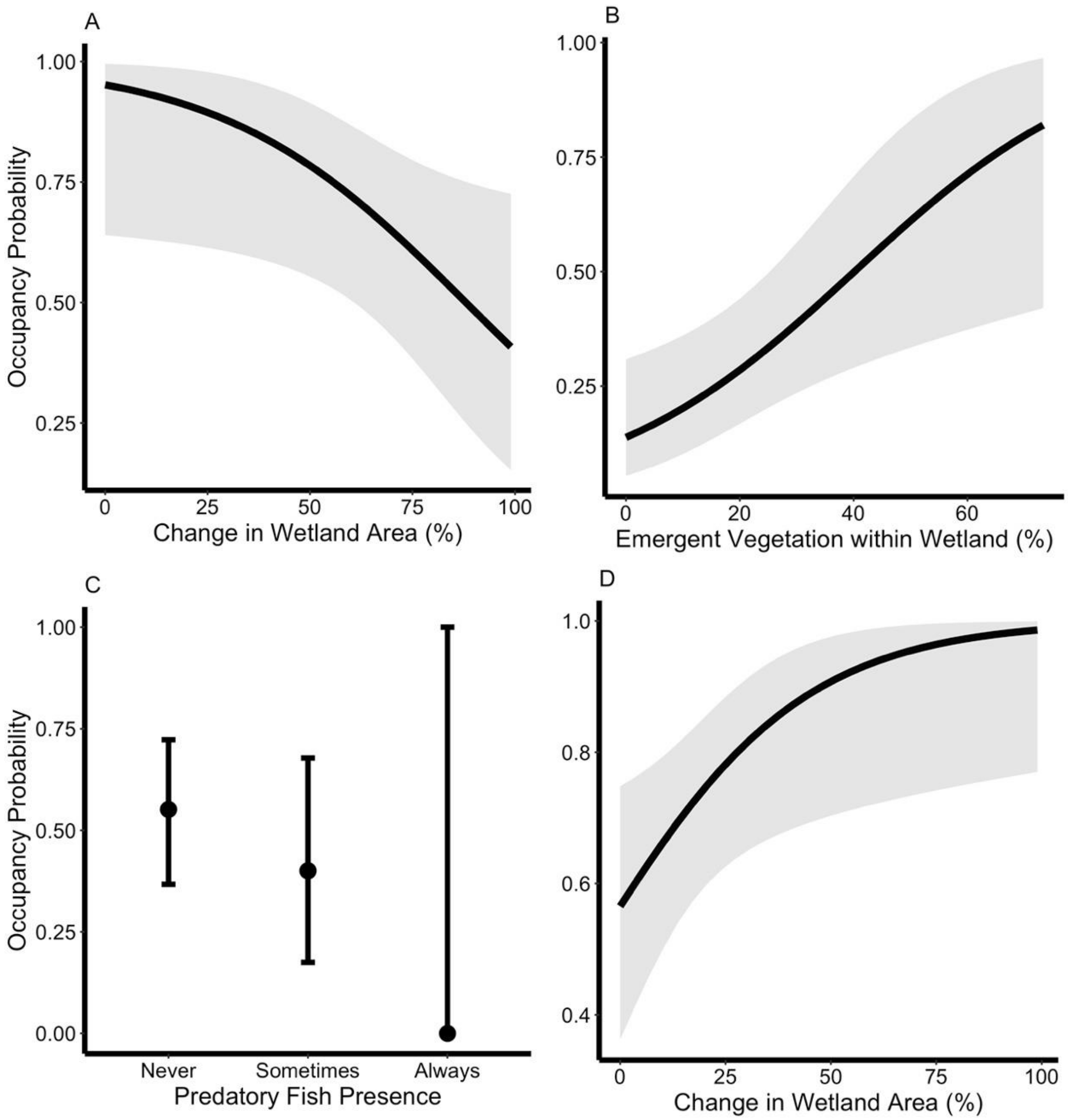

Figure 2.2. Predictive plots based on the top occupancy model for A) Blanchard's cricket frog (*Acris blanchardi*), B) boreal chorus frog (*Pseudacris maculata*), C) *Hyla* spp. (*Hyla chrysoscelis/versicolor*), and D) southern leopard frog (*Lithobates sphenocephalus*) in wetlands during the breeding seasons of 2021 and 2022 in southeast Kansas. Error bars represent 95% confidence intervals.

Table 2.4. Top models (ΔAICc <2) of the effects of wetland characteristics on the amphibian species richness and Shannon diversity in wetlands across southeast Kansas during 2021 and 2022. Null models have also been included, along with each model's parameters (K) and weights. See Table 2.1 for variable definitions.

Response Variable	Model	K	ΔAICc	Weight
Richness	fish + cond + emveg	7	0	0.41
	fish + cond	6	0.69	0.29
	fish + cond + do	7	0.88	0.26
	null	2	26.85	0.00
Diversity	cond + fish + hydro	7	0	0.75
	null	2	13.23	0

Table 2.5. Estimated coefficients for the top model of the effects of wetland characteristics on the species richness and Shannon diversity index of amphibian species in wetlands across southeast Kansas during 2021 and 2022.

Response Variable	Parameter	Estimate	Standard Error	Lower 95% CI	Upper 95% CI
Richness	intercept	4.14	0.45	3.38	4.90
	sometimes fish	0.81	0.55	-0.11	1.74
	always fish	-1.82	0.52	-2.69	-0.94
	medium cond	-1.36	0.48	-2.16	-0.55
	high cond	-2.27	0.60	-3.28	-1.26
	emveg	0.02	0.01	0.00	0.04
Diversity	intercept	0.95	0.11	0.76	1.13
	medium cond	-0.27	0.12	-0.47	-0.08
	high cond	-0.43	0.15	-0.69	-0.17
	sometimes fish	0.14	0.13	-0.09	0.36
	always fish	-0.38	0.13	-0.59	-0.16
	hydro	-0.42	0.18	-0.72	-0.13

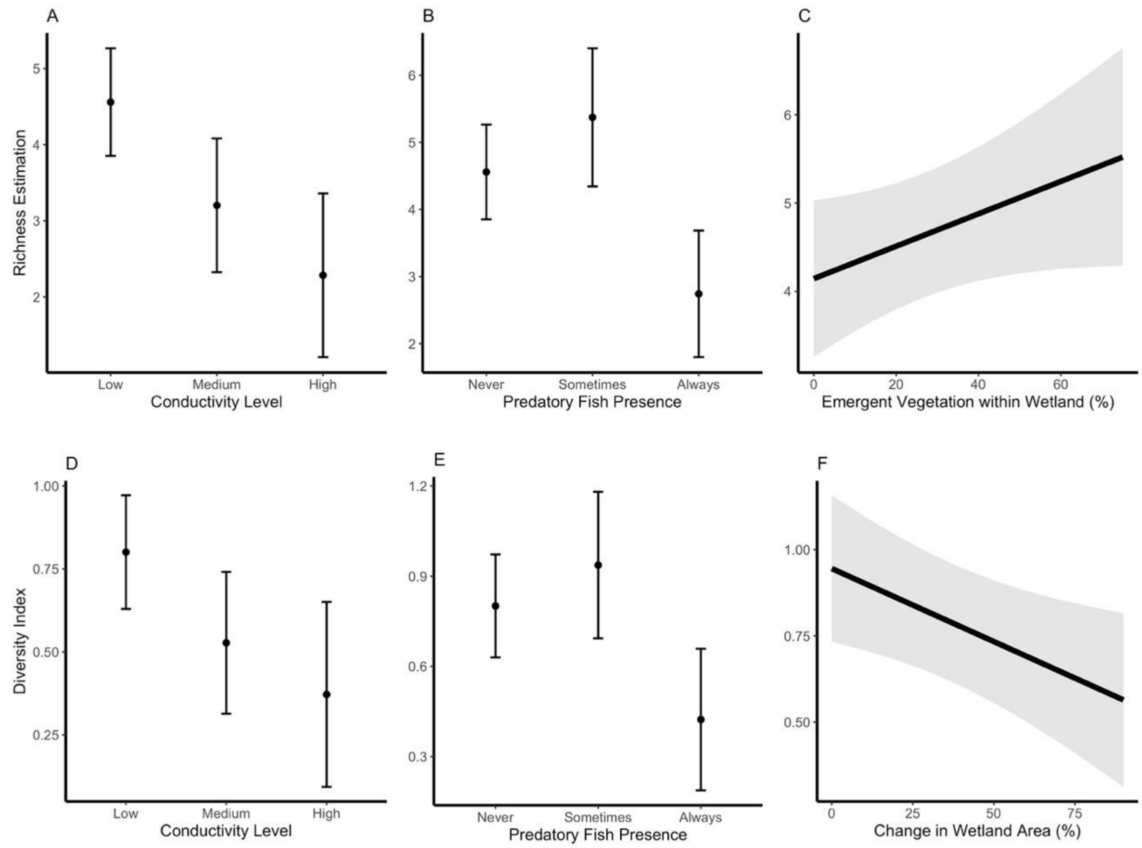

Figure 2.3. Wetland characteristics that influence the species richness and Shannon diversity of amphibian communities in southeast Kansas in 2021 and 2022. Species richness was affected by A) conductivity level, B) the presence of predatory fish, and C) percent of emergent vegetation within wetlands. Diversity was affected by D) the conductivity level, E) the presence of predatory fish, and F) the percent change in wetland area within a year. Fish presence was measured as the level of predator fish species (e.g., bass, sunfish, and gar) presence, where "sometimes" refers to the change in fish presence within a year. Conductivity level was measured as low (< 500 µs/cm), medium (500 – 1499 µs/cm), and high (≥ 1500 µs/cm).

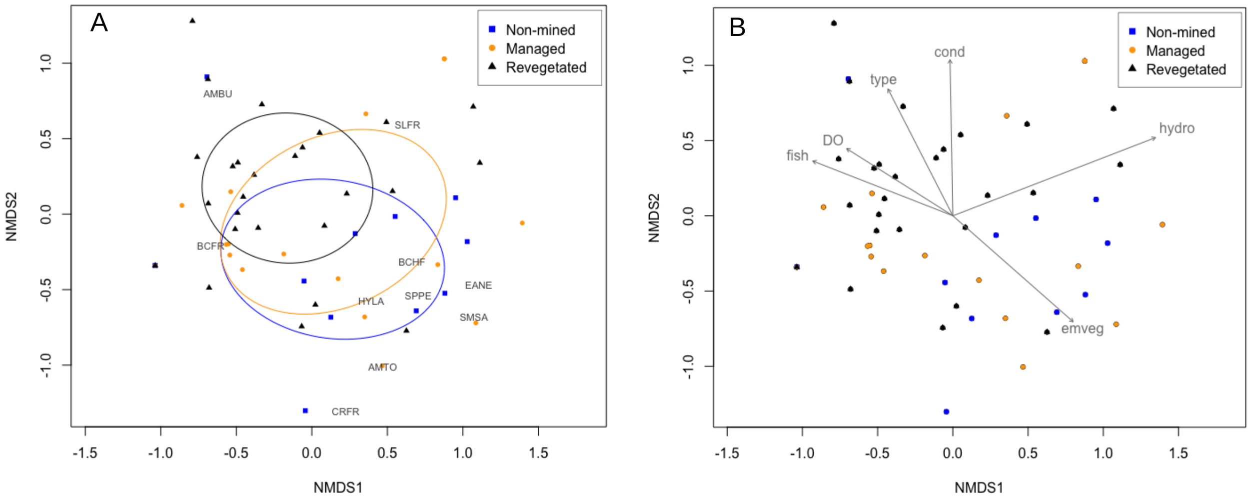

Figure 2.4. NMDS ordination plot showing amphibian community structure in southeast Kansas during 2021 and 2022. A) Wetland type was used to depict differences between communities, with the ellipses representing the standard deviations of site scores. Letters represent species codes: AMBU = American bullfrog, AMTO = American toad, BCFR = Blanchard's cricket frog, BCHF = boreal chorus frog, EANE = eastern newt, *Hyla* = *Hyla* spp., SMSA = smallmouth salamander, SLFR = southern leopard frog, and SPPE = spring peeper. B) Amphibian community structure associated with wetland characteristics (variables with $p < 0.05$). Arrows represent the direction and magnitude of the wetland characteristics in relation to the wetland communities. Points represent sites with the color and shape indicating the wetland type of each site.

Discussion

The post-mined landscape provided larval habitats for individual species and for the entire amphibian community. Occupancy of individual species varied, but generally was associated with the absence of predatory fish, hydroperiod and percent cover of emergent vegetation. Richness and diversity of the larval amphibian community were associated with conductivity levels, predatory fish presence, percent change in wetland area, and percent of emergent vegetation within the wetland.

Conductivity and other water quality metrics are generally considered important for the survival of amphibians due to their semi-permeable skin that can absorb pollutants and other impurities that can alter their behavior and development (Chambers, 2011; Karraker et al., 2008; Pollet & Bendell-Young, 2000). We found that lower conductivity levels were associated with higher species richness and diversity. High conductivity levels can disrupt larval behavior and decrease survival rates (Chambers, 2011). Conductivity measurements in waterways are often associated with the increased runoff of road salt, which can decrease the number of egg masses and larvae (Karraker et al., 2008). However, in our survey area the primary cause for higher conductivity was likely not from road salt, but instead from heavy metals from mining activities, as mining can release various heavy metals into the water (Evans et al., 2021). Although the amphibian communities did not vary between sites with different mining histories and management, sites with high conductivity were more often revegetated wetlands.

Risk of predation can also influence amphibian communities. For example, limiting fish, both predatory and non-predatory, or the absence of predatory fish, can increase amphibian species richness and occupancy (Boone et al., 2007; Hartel et al., 2007; Hecnar & M'Closkey, 1997). Particularly, the age and size structure of predatory fish were meaningful in describing the diversity of amphibian communities, with wetlands containing larger, older fish resulting in smaller amphibian populations and communities (Kloskowski, 2009). By breaking up the presence of predatory fish into three categories, we were able to examine the change in fish presence over the season and its influence on amphibian communities and *Hyla* spp. occupancy. The complete and partial absence of predatory fish increased the likelihood of *Hyla* spp. occupancy and increased richness and diversity. Often wetlands that sometimes had predatory fish were smaller and more likely to dry out completely each year. Thus, these wetlands only had the addition of predatory fish due to flooding events that connected smaller wetlands with larger, deeper wetlands. The temporary influx of predatory fish decreased the number of amphibians in the short term, but likely allowed for a quick return to high quality habitat for a large number of amphibian species due to the limited time that larger fish persisted within that wetland system (Kloskowski, 2009).

The hydroperiod of wetlands often indicates which species will breed in a wetland because other characteristics, like fish and vegetation, are often a result of the hydroperiod (Brodman, 2008). Wetlands with shorter hydroperiods are often associated with emergent vegetation and fewer fish because there are seasonally dry periods with little or no standing water, while longer hydroperiods are more likely to contain predatory fish and limit emergent vegetation. Although we did not directly measure the hydroperiod of each wetland, the percent change in wetland area over the survey period can be used as a proxy because the wetlands that dry early will likely stay dry and the wetlands with no size differences likely are wet year-round. We found that percent change in wetland area and emergent vegetation were informative for a number of our studied species and the overall amphibian community.

Blanchard’s cricket frog occupancy was negatively associated with the percent change in wetland area between March and June. The relatively quick rate at which some wetlands may dry has been shown to decrease the survival rate of cricket frog larvae, which can ultimately decrease the occupancy in the wetlands that regularly dry quickly and early in the year (Gordon et al., 2016). On the other hand, southern leopard frog occupancy was positively associated with the percent change in wetland area. Southern leopard frogs breed early in the year, allowing for increased time to reach metamorphosis. The positive association with a shorter hydroperiod is likely the driving force for other wetland characteristics that supported the occupancy of other species, such as the absence of predatory fish and increased emergent vegetation.

Emergent vegetation influenced boreal chorus frog occupancy and amphibian species richness. Increased emergent vegetation within a wetland provides expanded microhabitats that can support an increased number of species (Burne & Griffin, 2005). Boreal chorus frogs breed in early spring when there is limited emergent vegetation, so emergent vegetation likely represents other wetland characteristics that chorus frogs prefer in their breeding habitat. More emergent vegetation within a wetland often indicates that the wetland is shallow, which can lead to periodic drying events that limit the number of predatory fish species.

Wetland characteristics can often easily be seen as correlated with one another, making it a challenge to determine which characteristic is driving others that may be influencing occupancy of individual species and the entire amphibian community. The manipulation of wetlands has various impacts on the amphibian community, which are often driven by how the characteristics of natural wetlands are represented in reclaimed or managed wetlands (Brown et al., 2012; McPherson et al., 2020; Shulse et al., 2010, 2012). The amphibian community was not delimited by the type of mining and management history that a wetland had undergone, but some individual species, like American bullfrog, Blanchard's cricket frog, and southern leopard frog, tended to occur specific sites that were not occupied by other species in any large numbers. This similarity in amphibian communities shows that the wetlands across the landscape provided the variation and habitat conditions needed to support the full community. Although this study primarily focused on the mined land wetlands, the sampling region is surrounded by farm ponds, which also can support breeding populations of amphibians (Swartz & Miller, 2021). The combination of mined land wetlands and pond wetlands on the landscape may provide the diversity and connectivity of wetland habitats necessary to promote highly diverse communities or sustained populations (Brodman, 2008; Gibbs, 2000).

Reclaimed mined lands are often used for recreational opportunities, like hunting and fishing. Common practices for the management of deep wetlands include stocking game fish like trout and bass, or seasonal draining to increase waterfowl habitat for the winter hunting season. These practices may be beneficial to amphibians by providing a variety of wetland conditions, including the absence of predatory fish, hydroperiod, and amount of emergent vegetation. However, these practices may also be detrimental to some individual species that have other requirements that are not being met by management for waterfowl or fish. For example, the eastern newt (Kansas SINC species) was found to be breeding in only two ponds, one of which was actively managed for waterfowl by drying and tilling the area by July (Buckardt et al., 2022). This early drying likely limits survival rates of larvae prior to and after metamorphosis, due to the decreased larval sizes (Werner, 1986).

Our study demonstrates that variability among wetlands across the landscape provided common and SINC amphibian species with diverse habitats, although there was some uncertainty in our results. Variation in water quality, fish presence, emergent vegetation, and hydroperiod at various wetlands allowed the larval amphibian community to persist in a post-mined landscape. The continued protection and reclamation of wetlands could mitigate amphibian population declines and support other vertebrate communities that use wetlands (Ramsar Convention on Wetlands, 2018). Variation in the timing of management practice for managed wetlands can help to mimic the natural variations that occur from year to year and therefore help promote a more diverse biotic community.

CHAPTER III

TO GLOW OR NOT TO GLOW: EFFECTIVENESS OF GLOW STICKS AND TRAP METHOD ON THE CAPTURE RATES OF LARVAL AMPHIBIANS

ABSTRACT

Monitoring larval amphibians can be an important part of assessing populations and wetland health. The preferred methods to capture larval amphibians often vary based on the research question and logistics. But the relative efficacy of differing survey methods for a given species is often unknown. We aimed to examine how the capture rates for all amphibian larvae and five focal larval species were affected by season and survey method (i.e., dipnetting, un-baited minnow traps, and baited minnow traps). We surveyed 28 wetlands for amphibian larvae from mid-March to the end of June during 2021 and 2022 in southeast Kansas. We surveyed each wetland three times each year with 4 dipnet and 4 minnow trap locations for a 48-hr period, resulting in a total of 681 dipnet and 664 trap sampling locations. Green glow sticks were randomly placed in traps for a 24-hr period during each survey event, resulting in 1327 trap nights. We used generalized linear mixed-effects models to determine the effects of time of year and survey method on capture rates of individual larval species and of all larval species. Capture rates for total amphibians, American bullfrog (*Lithobates catesbeianus*), *Hyla*

spp., and southern leopard frog (*Lithobates sphenocephalus*) changed over the seasons, depending on capture method. Capture rates of American bullfrogs and Blanchard's cricket frogs (*Acris blanchardi*) changed throughout the year based on the presence of bait. Minnow traps baited with glowsticks increased the total number of amphibian larvae captured, but these effects varied for individual species. The choice of dipnets, baited traps and un-baited traps for sampling larval amphibians needs to be carefully considered but using both methods may provide a more complete understanding of the larval wetland community.

INTRODUCTION

Amphibians are considered indicators of wetland health because their physiology and life history are closely tied to the wetland conditions(Taylor et al., 2020). Thus, reliable sampling methodology is important for assessing both amphibian populations and for monitoring wetland dynamics over time. Methods of capturing aquatic larvae range from dipnetting and seining to various types of traps; each technique has its own benefits and drawbacks (Skelly & Richardson, 2009). Research question and logistics are often the deciding factors between these methods. The choice can become more difficult when the relative merits of each method are unknown. Thus, to ensure studies can fit within a project's limited time and resources, survey methods should be compared to maximize larval amphibian capture rates.

Common survey methods for aquatic amphibians are dipnetting and trapping. Dipnetting has the advantage when a study has limited resources, as there is minimal equipment costs and surveys at a single location can be completed in one day (Skelly & Richardson, 2009). On the other hand, trapping requires more time, equipment, and often additional labor because a single project may require multiple traps to be checked within 24 hrs (Skelly & Richardson, 2009). Dipnet surveys have the potential to disturb habitat by scrapping the bottom of the wetland, which can change the microhabitats that the egg masses and larvae need. Trapping minimizes these disturbances because the traps are placed on top of the substrate or within the water column (Richter, 1995). Additionally, the decision to trap over dipnet, or vice versa, may depend on the project's focal species. Elusive or nocturnal species such as the greater siren (*Siren lacertina*) and the two-toed amphiuma (*Amphiuma means*) may be more easily captured with traps (Denton & Richter, 2012; Johnson & Barichivich, 2004; Willson et al., 2011).

The use of bait within a trap may attract target species, increasing their capture rates. Recent studies have demonstrated increased trap capture rates with the use of glow sticks, particularly for salamanders like the eastern newt (*Notophthalmus viridescens*; Grayson & Roe, 2007) and tiger salamander (*Ambystoma tigrinum*; Liebgold & Carleton, 2020). These baited traps captured other amphibian species in addition to the target species; however, the efficacy of glowstick-baited traps for other amphibian larvae and adults has not been examined.

Our goal was to assess the effectiveness of dipnetting, minnow traps, and the use of glow sticks as bait for capturing larval amphibians. We compared catch per unit effort (CPUE) between a standardized dipnet method and a modified metal minnow trap, as well as the CPUE for glow stick baited and un-baited traps. We predicted that there would be an increased total CPUE for baited traps, but that CPUE would differ by species due to varying attraction to the green light. Documenting capture rates for alternate methods will allow researchers to more accurately assess larval amphibian communities.

METHODS

We sampled larval amphibians at 28 wetlands with known breeding populations across southeast Kansas. We surveyed for two consecutive days during three different survey windows defined as mid-March to mid-April (Early Spring), May (Spring), and June (Summer) during 2021 and 2022 to account for the variability of breeding times of anuran species in Kansas. We evenly spaced sampling locations along the shoreline starting from a random center point and placed sampling locations in both directions. We scaled distance between sampling locations by wetland size, ranging from 5 m between samples (wetlands < 0.05 ha), 10 m (wetlands between 0.05 ha to 0.35 ha), and 20 m (wetlands > 0.35 ha). We alternated between placing a minnow trap (four locations) and dipnetting (four locations) along the transects, for a total of eight sampling locations per wetland. For each survey window, the sampling locations within a site changed based on the fluctuating water line throughout the spring and summer. We also attempted to sample as much of the wetland edge as possible. This study design resulted in a total of 681 dipnet and 664 trap sampling locations in 2021 and 2022. The difference between the number of dipnet and trapping locations is due to lower water levels at some wetlands due to summer drying. There were1327 trap nights (24hr period) in 2021 and 2022.

We used modified minnow traps (Gee's Galvanized Wire Minnow Trap) with window screening to ensure that smaller larvae could be captured (Skelly & Richardson, 2009; Swartz & Miller, 2018). We baited each trap with a green glow stick for one night per survey window with a green glow stick (Glow with Us 6" light sticks), resulting in most nights having only two traps baited at a single time. Traps were placed at varying distances from the shore to ensure that the funnel entrance was at least half covered with water, while also having a portion of the trap exposed to the air for air-breathing species.

We checked traps after ca 24 hrs, removed spent glow sticks, and baited the previously un-baited traps.

We used a D-frame dipnet to conduct dipnet surveys. At each dipnetting location, we extended the dipnet approximately 1 m from the waterline into the wetland and quickly pulled it toward the shore along the bottom following a zig-zag motion (Babbitt et al., 2003). We dipnetted at the same locations on the two consecutive days of the sampling period. We identified and counted all amphibian larvae in the traps and dipnets in the field prior to releasing them at their capture sites.

We used generalized linear mixed models with a Poisson error distribution using the lme4 package in R to compare the effects of survey method on CPUE. (Bates et al., 2022; R Core Team, 2020). We compared the total number of larvae captured (CPUE) in dipnets and traps for a 48-hr period during each season because sampling locations remained in the same during this time. Models included site and year as random effects to account for non-independence of samples from the same time periods and locations. Fixed effects included survey method, survey season, and their interaction. To examine the effect glowstick baited and un-baited traps on capture rates, we used the CPUE of a singular trap night (ca. 24-hr period) to account for non-independence of samples from the same wetland and year. Random effects included year and site and fixed effects included day or year, baited or un-baited, and their interaction.

We examined models for the total larval amphibian CPUE and for five focal species that were captured in high enough quantities for analysis, including American bullfrog (*Lithobates catesbeianus*), Blanchard's cricket frog (*Acris blanchardi*), gray treefrog complex (*Hyla chrysoscelis/versicolor*; hereafter referred to as *Hyla* spp.), smallmouth salamander (*Ambystoma texanum*), and southern leopard frog (*Lithobates sphenocephalus*). Models for individual species were confined to only survey windows that represented when larvae could be captured based on local breeding phenology (Taggart, 2022). The number of sites included for each species was also limited to sites with breeding presence to prevent overinflating zero captures.

RESULTS

We captured 10 species of larval amphibians, including American bullfrog, American toad (*Anaxyrus americanus*), Blanchard's cricket frog, boreal chorus frog (*Pseudacris maculata*), crawfish frog (*Lithobates areolatus*), eastern newt, *Hyla* spp., smallmouth salamander, southern leopard frog, and spring peeper (*Pseudacris crucifer*). All species were captured using the various survey methods except the eastern newt larvae, which were captured by dipnet only.

Dipnet and trap CPUE for all amphibian larvae (n = 1345) was related to the interaction effect between season and survey method (Fig. 3.1). Traps captured more individuals than dipnets in the spring and summer while dipnet captures stayed consistent over the seasons (Fig. 3.1). American bullfrog larvae (n = 552) capture rates were higher in traps in the early spring, but this species had a similar CPUE between survey methods during the spring and summer (Fig. 3.1). CPUE for Blanchard's cricket frog (n = 720) was best explained by season and capture method, with more captures in the summer and with a dipnet (Fig. 3.1). *Hyla* spp.'s (n = 384) CPUE also increased in the summer and with a dipnet (Fig. 3.1). The CPUE for southern leopard frogs (n = 1177) was related to the interaction between survey method and season, with more captures in traps later in the year, but similar CPUE for dipnet throughout the year (Fig. 3.1). Smallmouth salamander (n = 208) capture rates increased using traps and during the spring (Fig. 3.1).

The total number of amphibians captured by traps (n= 1327) increased with day of year and increased with the use of glow stick bait (Fig. 3.2). *Hyla* spp. (n= 367) capture rates increased with the day of the year, but did not change with the bait presence (Fig. 3.2). Smallmouth salamander (n = 159) capture rates decreased with the day of year and there was no meaningful difference in capture rates with the use of bait (Fig. 3.2). Southern leopard frog (n = 1159) capture rates increased with the day of year, and with baited traps (Fig. 3.2). American bullfrog (n = 648) capture rates were best explained by an interaction between baiting and day of year. Capture rates for American bullfrog decreased in baited traps over the year, while capture rates remained constant for un-baited traps (Fig. 3.2). Blanchard's cricket frog (n = 743) captures also had an interaction effect; as the year progressed, capture rate slightly increased for traps that were baited (Fig. 3.2).

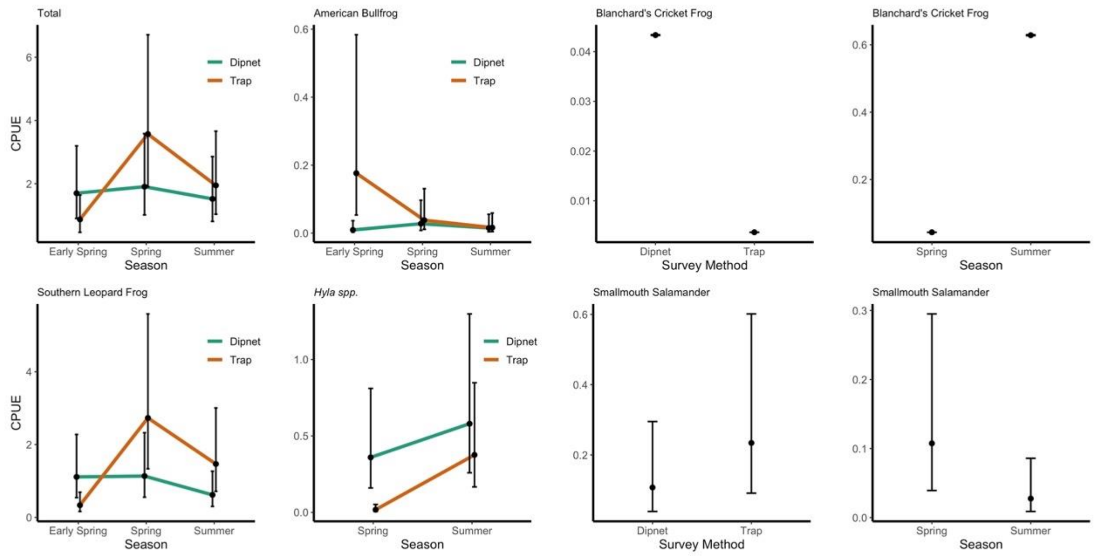

Figure 3.1. The effects of survey method and season (early spring, spring, and summer) on the capture per unit effort (CPUE) of larval amphibians in wetlands in southeast Kansas during 2021 and 2022. Capture rates for total amphibians, American bullfrog (*Lithobates catesbeianus*), *Hyla* spp., and southern leopard frog (*Lithobates sphenocephalus*) differed by survey method and season. Season and survey method affected CPUE for Blanchard's cricket frog (*Acris blanchardi*) and smallmouth salamander (*Ambystoma texanum*). Blanchard's cricket frog, *Hyla* spp., and smallmouth salamanders only were examined for spring and summer, as larvae were only found during those seasons. Error bars and shading indicate 95% confident intervals.

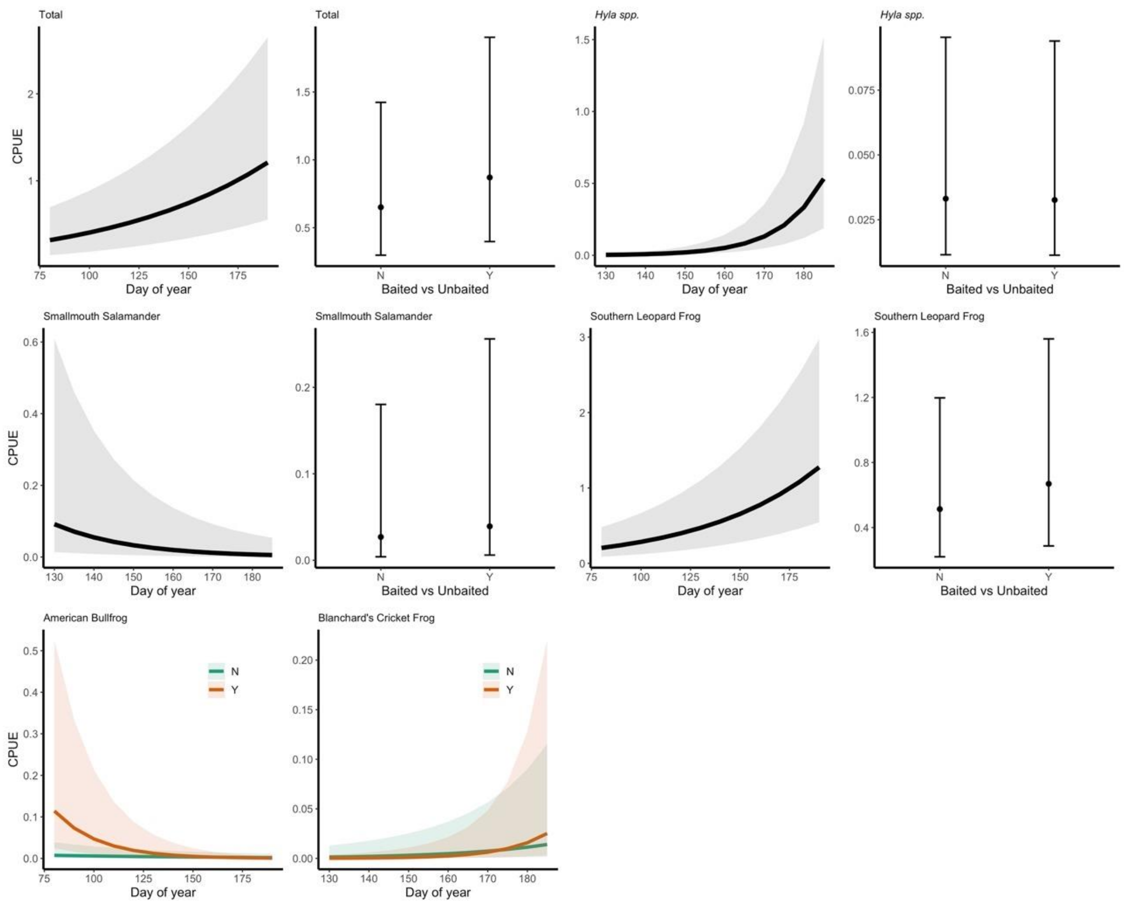

Figure 3.2. The effects of baiting minnow traps with glow sticks and season (early spring, spring, and summer) on the capture per unit effort of larval amphibians in wetland in southeast Kansas during 2021 and 2022. Capture rates for total amphibians, *Hyla* spp., smallmouth salamander (*Ambystoma texanum*) and southern leopard frog (*Lithobates sphenocephalus*) were affected by day of year and bait presences. American bullfrog (*Lithobates catesbeianus*) and Blanchard’s cricket frog (*Acris blanchardi*) capture rates changed throughout the year based on bait presence. Error bars and shading indicate 95% confident intervals.

DISCUSSION

While the effects of capture method and baiting varied by species, minnow traps baited with glowsticks increased the total number of amphibian larvae captured. The optimal method to use will depend on the season and target species. To maximize the capture rates for larvae, the use of traps during the time of year that a wetland has the highest number of larvae present.

Although traps produced higher capture rates for most species because of their passive approach to capturing individuals, *Hyla* spp. and Blanchard's cricket frogs were primarily captured by dipnet. These species have similar breeding times, restricting their potential sampling to only the spring and summer, and thus limiting the size of larvae available when wetlands were surveyed. While we modified our minnow traps to decrease mesh size, smaller bodied larvae were not captured as often in traps compared to dipnets. Thus, our findings may have been skewed towards species with larger larvae such as American bullfrog and southern leopard frog, causing the differences in CPUE between survey method. Varying capture rates for *Hyla* spp. between survey methods have been reported elsewhere, suggesting that other factors are at play such as time of year and wetland characteristics (Denton & Richter, 2012). The change in the effectiveness of either survey method through the year is likely linked to breeding cycle, where the highest capture rates for larvae are the season directly after the primary calling period of the adults.

The presence of glow sticks in the minnow traps reflected the difference between dipnetting and trapping but showed an overall smaller effect. The use of glowsticks increased the capture rates of eastern newts and American bullfrogs, suggesting that light as a bait source is beneficial to capture rates for a least some species (Grayson & Roe, 2007; Liebgold & Carleton, 2020). The use of light bait can be especially important when studying species of conservation concern that are rare on the landscape. For example, eastern newts are a state threatened species in Kansas and a county record was discovered with the use of a glowstick-baited trap (Buckardt et al., 2021; Rohweder, 2015). Although this record was an adult newt, the use of glowstick-baited traps for this species of conservation concern helped to find a population that may not have otherwise been detected. Even so, the use of a glow stick was not a universal attractant; we did not detect differences in CPUE between baited and un-baited traps for *Hyla* spp., smallmouth salamander, and Blanchard's cricket frog. Future research could lead to improved techniques for effectively sampling larvae of these species.

Changes to the effectiveness of glowsticks over the year are likely influenced by multiple factors. The length of the chemical reaction in glow sticks can change based on the temperature, with a longer glow time in colder temperatures. The longer light source in the trap may increase the number of individual larvae captured. Wetland characteristics such as vegetation may limit the visibility of the light source, as more vegetation may block the light and decrease the chances of it being seen by individuals that are farther away.

Although our study included two years of data across multiple sites, there are limitations that should be considered. The wetlands in this study were primarily pond-like habitats with limited emergent vegetation and relatively long hydroperiods. These features may increase the likelihood that species such as spring peeper or crawfish frog may be found (Babbitt et al., 2003). Additionally, we designed this survey (i.e., number and placement of traps) with the goal of examining the entire larval amphibian community. Although each dipnet and trap location was considered independent, adjacent locations may have been influenced by similar factors such as presence of a single artificial light source. Since we were examining the larval communities, we did not test the use of light bait on capture rates of adult amphibians, which likely differs due to their mobility and diet changes after metamorphous.

The use of dipnets, baited traps and un-baited traps for sampling larval amphibians should be carefully considered, as capture rates of individual species may differ. Using both methods when examining the entire community instead of a single species may provide a more complete understanding of a wetland community. The choice of methodology should be decided by the research question, logistics, and other factors like wetland habitat characteristics.

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APPENDIX

Appendix I. Location name, survey group, and coordinates for each call survey site. Asterisks indicate private property. Coordinates for sites on private property have been removed to protect the landowner's privacy.

Location	Latitude	Longitude
Buche Wildlife Area	37.316974	-94.682287
Ford N*	-	-
Ford S*	-	-
MLWA 10	37.266704	-94.809560
MLWA 11	37.265700	-94.837879
MLWA 12N	37.258830	-94.815712
MLWA 12W	37.252120	-94.823983
MLWA 13	37.251744	-94.800832
MLWA 14	37.244293	-94.814228
MLWA 16	37.236934	-94.832718
MLWA 17S	37.287281	-94.894402
MLWA 17W	37.294005	-94.904708
MLWA 18E	37.274680	-94.908684
MLWA 18N	37.278798	-94.922917
MLWA 18S	37.266982	-94.914834
MLWA 19	37.278018	-94.895768
MLWA 1E	37.477094	-94.692814
MLWA 1N	37.482111	-94.702619
MLWA 1S	37.470528	-94.702748
MLWA 21E	37.246787	-94.960168
MLWA 21S	37.237713	-94.961258
MLWA 21W	37.245497	-94.976008
MLWA 22E	37.231035	-94.983160
MLWA 22S	37.223694	-94.990934
MLWA 23	37.236269	-94.973374
MLWA 24E	37.208764	-95.001307
MLWA 24W	37.212982	-95.011926
MLWA 25	37.193670	-95.059222
MLWA 26	37.332893	-94.800483
MLWA 27	37.202004	-95.050163
MLWA 28	37.202911	-95.031941
MLWA 29	37.201895	-95.013651
MLWA 3	37.443976	-94.617400

MLWA 30	37.208272	-95.022606
MLWA 32	37.208684	-94.977684
MLWA 33	37.224965	-95.031858
MLWA 35E	37.223696	-95.002268
MLWA 35W	37.225870	-95.013272
MLWA 36	37.244601	-95.037620
MLWA 38E	37.251762	-94.926703
MLWA 38W	37.248576	-94.940461
MLWA 39	37.252689	-94.984668
MLWA 40	37.264013	-94.976427
MLWA 41	37.261499	-94.958279
MLWA 42E	37.259492	-94.924293
MLWA 42W	37.257327	-94.936826
MLWA 44	37.267074	-94.934636
MLWA 45	37.283367	-94.912269
MLWA 4E	37.433128	-94.617333
MLWA 4W	37.438060	-94.630769
MLWA 5	37.411957	-94.768700
MLWA 6N	37.423991	-94.754964
MLWA 6S	37.415987	-94.758231
MLWA 7N	37.396332	-94.778641
MLWA 7S	37.388040	-94.783519
MLWA 8	37.389996	-94.772590
MLWA 9	37.287609	-94.772275
Monahan Outdoor Education Center	37.350972	-94.801386
Natural History Reserve	37.374343	-94.781406
Pittsburg Bike Park	37.428762	-94.693380
Pittsburg High School	37.409146	-94.670453
Pittsburg Industrial Park	37.433169	-94.683672
Pittsburg State University	37.391364	-94.697968
Stefanoni*	-	-
Wilderness Park	37.454764	-94.713891

Appendix II. Detections of nine anuran species heard calling from 65 sites. Detection at each site is indicated as the following: blank = not detected, 21 = only detected in 2021, 22 = only detected in 2022, and X = detected in 2021 and 2022.

Survey Point	American bullfrog	American toad	Blanchard's cricket frog	Boreal chorus frog	Cope's gray treefrog	Crawfish frog	Gray treefrog	Southern leopard frog	Spring peeper
Buche Wildlife Area	22	X	X	X	X	X	22	X	X
Ford N	X	22	X	X	X		X	X	
Ford S	X	X	X	X	X			X	
MLWA 10	X	X	X	21	X		21	X	X
MLWA 11	X	X	X	X	X			X	X
MLWA 12N	22	X	X	X	X			X	X
MLWA 12W	X	X	X	X	X		X	X	X
MLWA 13	X	X	X	X	X			X	X
MLWA 14	X	X	X	X	X	22	21	X	X
MLWA 16		X	X	X	X			X	X
MLWA 17S	X	X	X	X	X			X	
MLWA 17W	X	X	X	X	X			X	
MLWA 18E	X	X	X	X	X	22		X	
MLWA 18N	X	X	X	X	X			X	
MLWA 18S	X	X	X	X	X	X	21	X	
MLWA 19	X	X	X	X	X	22		X	
MLWA 1E	X	X	X	X	X			X	X
MLWA 1N	22	X	X	X	X	21	22	X	22
MLWA 1S		22	X	X	X		22	22	X
MLWA 21E	X	X	X	X	X			X	
MLWA 21S	X	X	X	X	X			X	
MLWA 21W	X	X	X	X	X		X	X	
MLWA 22E	X	X	X	X	X			X	
MLWA 22S	22	22	X	X	X		21	X	
MLWA 23	X	22	X	X	X			X	

MLWA 24E	X	X	X	X	X		22	X	
MLWA 24W	X	22	X	X	X			X	
MLWA 25	21	X	X	X	X			X	
MLWA 26	X	X	X	X	X			X	X
MLWA 27	X	X	X	X	X	X	22	X	
MLWA 28	X	X	X	X	X	X	X	X	
MLWA 29	X	22	X	X	X	X	22	X	
MLWA 3	X	X	X	X	X	21		X	X
MLWA 30	X	22	X	X	X		22	X	
MLWA 32	22	X	X	X	X			X	
MLWA 33	X	X	X	X	X		21	X	
MLWA 35E	X	X	X	X	X	X		X	
MLWA 35W	X	X	X	X	X	21	22	X	
MLWA 36	X	X	X	X	X			X	
MLWA 38E	X	X	X	X	X	X		X	
MLWA 38W	X	X	X	X	X	22		X	
MLWA 39	X	X	X	X	X		X	X	
MLWA 40	X	22	X	X	X	22	21	X	
MLWA 41	22		X	X	22			X	
MLWA 42E	X	X	X	X	X			X	
MLWA 42W	X	X	X	X	X			X	
MLWA 44	22	X	X	X	X	22		X	
MLWA 45	X	X	X	X	X			X	22
MLWA 4E	X	X	X	X	X			X	X
MLWA 4W	X	X	X	X	X	21		X	X
MLWA 5	X	X	X	X	X			X	X
MLWA 6N	X	X	X	X	X	X		X	X
MLWA 6S	X	X	X	X	X			X	X
MLWA 7N	X	X	X	X	X	21		X	X
MLWA 7S	X	X	X	X	X	21		X	22
MLWA 8	X	X	X	X	X	X		X	X
MLWA 9	22	X	X	X	X	21		X	X

Monahan Outdoor Education Center	22	22	X	X	X			X	X
Natural History Reserve	X	X	X	X	X	X		X	X
Pittsburg Bike Park		X	X	X	X		22	X	X
Pittsburg High School	X	X	X	X	X			X	X
Pittsburg Industrial Park		X	X	X	X	21	22	X	X
Pittsburg State University		22	X						
Stefanoni		X	X	X	X	21	22	X	X
Wilderness Park		X	X	X	X	21	22	X	X

Appendix III. Occupancy models estimating the probability that American bullfrogs (*Lithobates catesbeianus*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + obs) ψ(water + built)	8	0	0.92
p(day + obs) ψ(water + built + grass)	9	5.06	0.07
p(day + obs) ψ(built + watershed)	8	9.78	0.01
p(day + obs) ψ(water + built +watershed)	9	13.26	0
p(day + obs) ψ(built)	7	18.17	0
p(day + obs) ψ(built + wetland)	8	20.45	0
p(day + obs) ψ(built + forest)	8	20.45	0
p(day + obs) ψ(built + year)	8	21.62	0
p(day + obs) ψ(watershed)	7	23.60	0
p(day + obs) ψ(water + built + wetland)	9	29.29	0
p(day + obs) ψ(forest)	7	29.43	0
p(day + obs) ψ(crop)	7	31.80	0
p(day + obs) ψ(.)	6	32.35	0
p(day + obs) ψ(year)	7	32.73	0
p(day + obs) ψ(wetland)	7	34.48	0
p(day + obs) ψ(built + crop)	8	40.56	0
p(day + obs) ψ(grass)	7	47.88	0
p(day + obs) ψ(water)	7	47.89	0
p(day + obs) ψ(built + grass)	8	50.15	0
p(day + obs) ψ(water + built + year)	9	52.47	0
p(day + obs) ψ(water + built + crop)	9	52.47	0
p(day + obs) ψ(water + built + forest)	9	52.48	0
p(.) ψ(.)	2	322.31	0

Appendix IV. Occupancy models estimating the probability that crawfish frogs (*Lithobates areolatus*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + obs) ψ(crop + watershed + year)	9	0	0.65
p(day + obs) ψ(crop + watershed)	8	4.88	0.06
p(day + obs) ψ(crop + watershed + built)	9	5.74	0.04
p(day + obs) ψ(crop)	7	5.80	0.04
p(day + obs) ψ(crop + watershed + wetland)	9	5.86	0.03
p(day + obs) ψ(crop + watershed+ grass)	9	6.75	0.02
p(day + obs) ψ(crop + watershed + water)	9	6.81	0.02
p(day + obs) ψ(crop + watershed + forest)	9	7.18	0.02
p(day + obs) ψ(crop + wetland)	8	7.49	0.02
p(day + obs) ψ(watershed)	7	7.56	0.01
p(day + obs) ψ(crop + year)	8	7.59	0.01
p(day + obs) ψ(crop + grass)	8	7.74	0.01
p(day + obs) ψ(crop + built)	8	7.86	0.01
p(day + obs) ψ(water)	7	7.87	0.01
p(day + obs) ψ(crop + water)	8	7.98	0.01
p(day + obs) ψ(crop + forest)	8	8.01	0.01
p(day + obs) ψ(.)	6	9.23	0.01
p(day + obs) ψ(built)	7	10.07	0
p(day + obs) ψ(year)	7	10.43	0
p(day + obs) ψ(grass)	7	10.56	0
p(day + obs) ψ(forest)	7	10.85	0
p(day + obs) ψ(wetland)	7	11.02	0
p(.) ψ(.)	2	45.07	0

Appendix V. Occupancy models estimating the probability that gray treefrogs (*Hyla versicolor*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + obs) ψ(grass)	7	0	0.3
p(day + obs) ψ(built)	7	0.42	0.24
p(day + obs) ψ(wetland)	7	1.65	0.13
p(day + obs) ψ(year)	7	2.62	0.08
p(day + obs) ψ(.)	6	2.76	0.08
p(day + obs) ψ(forest)	7	2.78	0.07
p(day + obs) ψ(water)	7	4.27	0.04
p(day + obs) ψ(crop)	7	4.61	0.03
p(day + obs) ψ(watershed)	7	4.73	0.03
p(.) ψ(.)	2	45.24	0

Appendix VI. Occupancy models estimating the probability that spring peepers (*Pseudacris crucifer*) would occupy a call site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 1.2 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day + noise) ψ(watershed+ built + crop)	10	0	0.83
p(day + noise) ψ(watershed + built + water)	10	4.36	0.09
p(day + noise) ψ(watershed+ built + wetland)	10	7.23	0.02
p(day + noise) ψ(watershed + built)	9	7.81	0.02
p(day + noise) ψ(watershed+ built + forest)	10	8.53	0.01
p(day + noise) ψ(watershed + built + grass)	10	8.99	0.01
p(day + noise) ψ(watershed + built + year)	10	9.45	0.01
p(day + noise) ψ(watershed + forest)	9	10.12	0.01
p(day + noise) ψ(watershed + wetland)	9	12.82	0
p(day + noise) ψ(watershed + water)	9	13.48	0
p(day + noise) ψ(watershed + crop)	9	13.52	0
p(day + noise) ψ(watershed)	8	13.75	0
p(day + noise) ψ(watershed + grass)	9	13.93	0
p(day + noise) ψ(watershed + year)	9	15.39	0
p(day + noise) ψ(water)	8	76.26	0
p(day + noise) ψ(wetland)	8	89.20	0
p(day + noise) ψ(.)	7	97.95	0
p(day + noise) ψ(grass)	8	98.34	0
p(day + noise) ψ(built)	8	99.41	0
p(day + noise) ψ(year)	8	99.94	0
p(day + noise) ψ(crop)	8	100.22	0
p(day + noise) ψ(forest)	7	100.45	0
p(.) ψ(.)	2	210.19	0

Appendix VII. Site names, coordinates, and mining history for each survey site in southeast Kansas. Coordinates for sites on private property have been removed to protect the landowner's privacy.

Site	Latitude	Longitude	Mining History
Buche Wildlife Area†	37.31900	-94.68000	Non-mined
Buche Wildlife Area 2º	37.31967	-94.68082	Non-mined
Ford E*	-	-	Non-mined
Ford W*	-	-	Non-mined
MLWA 1	37.47519	-94.69988	Revegetated
MLWA 10	37.26732	-94.81289	Revegetated
MLWA 14	37.24484	-94.81422	Revegetated
MLWA 17	37.28233	-94.89190	Revegetated
MLWA 18	37.27416	-94.90721	Revegetated
MLWA 23 N	37.23625	-94.96997	Revegetated
MLWA 23 S	37.2305	-94.97710	Revegetated
MLWA 24	37.21294	-95.01171	Revegetated
MLWA 25	37.19983	-95.05648	Revegetated
MLWA 28	37.20794	-95.03116	Revegetated
MLWA 30	37.20951	-95.02092	Managed
MLWA 35	37.22534	-95.01129	Managed
MLWA 36	37.24368	-95.03973	Managed
MLWA 38	37.24885	-94.94020	Managed
MLWA 39	37.25316	-94.97762	Managed
MLWA 4 E	37.25583	-94.97166	Revegetated
MLWA 4 W	37.2681	-94.93485	Managed
MLWA 40	37.43318	-94.61997	Managed
MLWA 44	37.43891	-94.62923	Managed
MLWA 6 N	37.42294	-94.75732	Managed
MLWA 6 S	37.41605	-94.75536	Revegetated
MLWA 7	37.38795	-94.78133	Revegetated
Monahan Outdoor Education Center	37.34896	-94.80429	Managed
O'Malley Prairie	37.35270	-94.79471	Non-mined
Pittsburg High School	37.40999	-94.67033	Non-mined
Natural History Reserve	37.37444	-94.77864	Revegetated
Stefanoni*	-	-	Non-mined

† Site only surveyed in 2021
º Site only surveyed in 2022
* Site on private property

Appendix VIII. Amphibian species captured by dipnet and trapping at 31 sites from 2021 and 2022 in southeast Kansas. Captures are indicated as the following: blank = not captured, 21 = only captured in 2021, 22 = only captured in 2022, and X = captured in 2021 and 2022. Buche was only surveyed in 2021 and Buche 2 was only surveyed in 2022.

Common Name	**Scientific Name**	Buche2	Buche	Ford E	Ford W	HS	ML1	ML10	ML14	ML17	ML18	ML23 N
American bullfrog	*Lithobates catesbeianus*	22	21		x			x	x	x	22	22
American toad	*Anaxyrus americanus*	22		21	21				22			
Blanchard's cricket frog	*Acris blanchardi*	22	21	22	x	x	x	x	x	22	22	22
Boreal chorus frog	*Pseudacris maculata*			x	22		22					21
Crawfish frog	*Lithobates areolatus*	22		22	x							
Eastern newt	*Notophthalmus viridescens*											
Gray treefrog complex	*Hyla chrysoscelis/versicolor*			x	x		x		x			
Smallmouth salamander	*Ambystoma texanum*	22		21								
Southern leopard frog	*Lithobates sphenocephalus*	22		x	x		x	22	x		22	x
Spring peeper	*Pseudacris crucifer*	22					22		22			

Common Name	**Scientific Name**	ML23 S	ML24	ML25	ML28	ML30	ML35	ML36	ML38	ML39	ML4 E	ML4 W
American bullfrog	*Lithobates catesbeianus*	x		x		x		x	21	22	21	
American toad	*Anaxyrus americanus*							22				
Blanchard's cricket frog	*Acris blanchardi*	x	x	x	x	x		x	x		21	22
Boreal chorus frog	*Pseudacris maculata*	21	x			22		x				
Crawfish frog	*Lithobates areolatus*											
Eastern newt	*Notophthalmus viridescens*											
Gray treefrog complex	*Hyla chrysoscelis/versicolor*							x		22		
Smallmouth salamander	*Ambystoma texanum*							x				
Southern leopard frog	*Lithobates sphenocephalus*	x	22	x	x	x	22	x		x	x	22
Spring peeper	*Pseudacris crucifer*										22	

Common Name	**Scientific Name**	ML40	ML44	ML6 N	ML6 S	ML7	Monahan	O'Malley	Reserve	Stefanoni
American bullfrog	*Lithobates catesbeianus*	22			22	22		21		
American toad	*Anaxyrus americanus*							21		22
Blanchard's cricket frog	*Acris blanchardi*	x	x		x	x	22	22	22	x
Boreal chorus frog	*Pseudacris maculata*	21	21	22				x		x
Crawfish frog	*Lithobates areolatus*									
Eastern newt	*Notophthalmus viridescens*			x	x					
Gray treefrog complex	*Hyla chrysoscelis/versicolor*	x	22	22	21	x		22		x
Smallmouth salamander	*Ambystoma texanum*	x		x				21		x
Southern leopard frog	*Lithobates sphenocephalus*	x	x	x	x	x	x	x		x
Spring peeper	*Pseudacris crucifer*			22						22

Appendix IX. Occupancy models estimating the probability that American bullfrog (*Lithobates catesbeianus*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(type) ψ(.)	5	0	0.17
p(.) ψ(.)	3	0.58	0.13
p(type) ψ(pH)	6	1.09	0.10
p(type) ψ(hydro)	6	1.60	0.08
p(type) ψ(emveg)	6	1.64	0.08
p(type) ψ(area)	6	1.82	0.07
p(type) ψ(pH + hydro)	7	2.00	0.06
p(type) ψ(DO)	6	2.30	0.05
p(type) ψ(pH + emveg)	7	2.75	0.04
p(type) ψ(fish)	7	2.83	0.04
p(type) ψ(cond)	7	3.01	0.04
p(type) ψ(pH + area)	7	3.46	0.03
p(type) ψ(pH + fish)	8	3.83	0.03
p(type) ψ(pH +hydro + emveg)	8	4.26	0.02
p(type) ψ(pH +cond)	8	4.91	0.01
p(type) ψ(type)	7	5.02	0.01
p(type) ψ(year)	6	5.06	0.01
p(type) ψ(pH + hydro +fish)	9	5.25	0.01

Appendix X. Occupancy models estimating the probability that Blanchard's cricket frog (*Acris blanchardi*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(day) ψ(hydro)	5	0	0.36
p(day) ψ(hydro + area)	6	1.29	0.19
p(day) ψ(.)	4	2.45	0.11
p(day) ψ(hydro + site)	7	2.66	0.10
p(day) ψ(area)	5	2.84	0.09
p(day) ψ(hydro + fish)	7	3.48	0.06
p(day) ψ(hydro + cond)	7	4.90	0.03
p(day) ψ(type)	6	5.14	0.03
p(day) ψ(cond)	6	6.29	0.02
p(day) ψ(fish)	6	6.85	0.01
p(day) ψ(year)	5	7.16	0.01
p(day) ψ(emveg)	5	12.11	0
p(day) ψ(DO)	5	12.49	0
p(day) ψ(pH)	5	12.49	0
p(.) ψ(.)	3	21.61	0

Appendix XI. Occupancy models estimating the probability that boreal chorus frog (*Pseudacris maculata*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the ΔAIC_c, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	ΔAIC_c	Model Weight
p(day) ψ(emveg)	4	0	0.46
p(day) ψ(emveg + DO)	5	1.40	0.23
p(day) ψ(emveg + fish)	6	3.29	0.09
p(day) ψ(DO)	4	3.61	0.08
p(day) ψ(emveg + type)	6	3.83	0.07
p(day) ψ(type)	5	5.47	0.03
p(day) ψ(fish)	5	7.13	0.01
p(day) ψ(.)	3	7.33	0.01
p(day) ψ(area)	4	8.01	0.01
p(day) ψ(hydro)	4	8.73	0.01
p(day) ψ(cond)	5	9.21	0
p(day) ψ(pH)	4	9.62	0
p(day) ψ(year)	4	10.70	0
p(.) ψ(.)	2	24.47	0

Appendix XII. Occupancy models estimating the probability that *Hyla* spp. (*Hyla chrysoscelis/versicolor*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(day) ψ(fish)	6	0	0.23
p(day) ψ(fish + cond)	8	1.02	0.14
p(day) ψ(fish + area)	7	1.69	0.10
p(day) ψ(fish + emveg)	7	1.95	0.09
p(day) ψ(fish + pH)	7	2.51	0.06
p(day) ψ(fish + hydro)	7	2.57	0.06
p(day) ψ(fish + cond + emveg)	9	2.93	0.05
p(day) ψ(fish + cond + hydro)	9	2.94	0.05
p(day) ψ(fish + cond + pH)	9	3.06	0.05
p(day) ψ(fish + type)	8	3.13	0.05
p(day) ψ(fish + area)	9	3.52	0.04
p(day) ψ(fish + cond + type)	10	4.14	0.03
p(day) ψ(.)	4	6.04	0.01
p(day) ψ(emveg)	5	6.42	0.01
p(day) ψ(pH)	5	6.93	0.01
p(day) ψ(area)	5	6.99	0.01
p(day) ψ(hydro)	5	7.54	0.01
p(day) ψ(DO)	5	8.13	0
p(day) ψ(cond)	6	8.24	0
p(day) ψ(year)	5	8.46	0
p(day) ψ(type)	6	8.54	0
p(.) ψ(.)	3	11.95	0

Appendix XIII. Occupancy models estimating the probability that southern leopard frog (*Lithobates sphenocephalus*) would occupy a wetland site during 2021 and 2022 in southeast Kansas. Null models have been included, along with the $\Delta QAIC_c$, model parameters (K), and weights. See Table 2.1 for variable definitions.

Model	K	$\Delta QAIC_c$	Model Weight
p(type + day) ψ(hydro)	7	0	0.22
p(type + day) ψ(.)	6	0.85	0.14
p(type + day) ψ(fish)	8	2.14	0.08
p(type + day) ψ(hydro + area)	8	2.34	0.07
p(type + day) ψ(hydro + emveg)	8	2.61	0.06
p(type + day) ψ(hydro +DO)	8	2.65	0.06
p(type + day) ψ(hydro + pH)	8	2.67	0.06
p(type + day) ψ(emveg)	7	2.95	0.05
p(type + day) ψ(pH)	7	3.27	0.04
p(type + day) ψ(hydro + fish)	9	3.34	0.04
p(type + day) ψ(DO)	7	3.34	0.04
p(type + day) ψ(area)	7	3.40	0.04
p(type + day) ψ(year)	7	3.95	0.03
p(type + day) ψ(hydro + fish + DO)	10	5.49	0.01
p(type + day) ψ(hydro + fish + pH)	10	5.85	0.01
p(type + day) ψ(cond)	8	5.91	0.01
p(type + day) ψ(type)	8	6.01	0.01
p(type + day) ψ(hydro + fish + emveg)	10	6.20	0.01
p(type + day) ψ(hydro + fish + area)	10	6.22	0.01
p(.) ψ(.)	3	12.48	0

Appendix XIV. Candidate set of models of the effects of wetland characteristics on the amphibian species richness in wetlands across southeast Kansas during 2021 and 2022. Null models have also been included, along with each models' parameters (K), ΔAIC_c, and weights. See Table 2.1 for variable definitions.

Models	K	ΔAIC_c	Model Weight
fish +cond + emveg	7	0	0.41
fish + cond	6	0.69	0.29
fish + cond + do	7	0.88	0.26
fish + cond + type	8	5.14	0.03
fish + emveg	5	11.64	0
fish + type	6	12.48	0
fish	4	13.09	0
fish + do	5	14.32	0
cond	4	18.27	0
emveg	3	20.21	0
do	3	21.13	0
type	4	22.78	0
area	3	23.86	0
ph	3	25.14	0
null	2	26.85	0
year	3	27.88	0
hydro	3	28.89	0

Appendix XV. Candidate set of models of the effects of wetland characteristics on the Shannon diversity index of amphibian in wetlands across southeast Kansas during 2021 and 2022. Null models have also been included, along with each models' parameters (K), ΔAIC_c, and weights. See Table 2.1 for variable definitions.

Model	K	ΔAIC_c	Model Weight
cond + fish + hydro	7	0	0.75
cond + fish	6	3.51	0.13
cond + fish + area	7	5.67	0.04
cond	4	6.57	0.03
cond + hydro	5	7.31	0.02
cond + area	5	7.34	0.02
fish	4	11.79	0
area	3	12.61	0
hydro	3	12.87	0
null	2	13.23	0
year	3	13.49	0
ph	3	14.28	0
do	3	15.14	0
emveg	3	15.18	0
type	4	17.58	0

DENSITY AND NEST SUCCESS OF SHRUB-DEPENDENT BIRDS ON FORMERLY STRIP-MINED LANDS

A Thesis Submitted to the Graduate School in Partial Fulfillment of the Requirements for the Degree of Master of Science

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March 2023

DENSITY AND NEST SUCCESS OF SHRUB-DEPENDENT BIRDS ON FORMERLY STRIP-MINED LANDS

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DENSITY AND NEST SUCCESS OF SHRUB-DEPENDENT BIRDS ON FORMERLY STRIP-MINED LANDS

An Abstract of the Thesis by
Luke A. Headings

As bird populations continue to decline across North America, it is important to understand the benefits that disturbed habitats can have for breeding birds. One of the major land disturbances and causes of habitat loss in the United States is surface mining, which often results in altered vegetative communities. The primary goal of this study was to evaluate the relationships between bird populations, habitat, previous and current land use, and densities of invasive plant species on formerly strip-mined land. Due to the proliferation of invasive shrub species in post-mined landscapes, we sought to determine the effects of post-mined habitat features on three shrub-nesting bird species: Bell's Vireo (*Vireo bellii*), Northern Cardinal (*Cardinalis cardinalis*), and Indigo Bunting (*Passerina cyanea*). In addition to assessing their densities, we estimated each species' reproductive success to understand future population trends. We conducted point count surveys, and searched for and monitored nests of these shrubland birds at 84 sites varying in land use and mining history. Overall, we detected 7,999 individuals from 87 bird species. Forested mined lands had the most diverse bird communities. We found that habitat type (i.e., forest, grassland, or rangeland) best described patterns in each focal species' density, with densities differing by habitat type for all three shrub-dependent species. We located 178 nests, the majority of which belonged to Bell's Vireos and Northern Cardinals. Logistic exposure models predicted daily nest survival for Bell's Vireos as a function of habitat type between post-mined grasslands and rangelands, while Northern Cardinals

daily nest survival was a function of nest age. If demographic rates were consistent across the study region, Bell's Vireo reproductive rates were not high enough to maintain their populations. Particularly as woody invasion continues, invasive shrub populations grow, and land cover changes occur in the Midwest, both species' breeding success may be negatively impacted, resulting in their population declines. This information will be useful for creating a more informed management plan for non-game birds and exotic plant species on reclaimed mined lands.

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CHAPTER I

DIVERSITY AND DENSITIES OF THREE SHRUB-DEPENDENT BIRD SPECIES ON FORMERLY STRIP-MINED LANDS

ABSTRACT

Shrubland habitats occupy a crucial position in ecological succession and host a wide variety of species, including shrub-dependent birds. Unfortunately, these habitats are often overlooked and understudied. Understanding the causes of bird declines in response to landscape changes is imperative, especially in the era of biodiversity loss. In this study, we examined the relationships between habitat features and densities of three shrub-dependent bird species on previously strip-mined land. We used fixed radius point counts to survey the bird communities on 84 locations in southeast Kansas and fit generalized linear mixed models to estimate densities of Bell’s Vireos, Northern Cardinals, and Indigo Buntings. We detected 7,999 individual birds from 87 species, including 13 species of conservation concern in Crawford and Cherokee counties. Habitat type was the best-supported model for predicting densities of all focal species. Bell's Vireos, Northern Cardinals, and Indigo Buntings had the highest densities in rangelands, forests, and grasslands, respectively. We demonstrated that formerly mined areas can support a diverse range of species, with the most diverse areas being the forested

sections. Management that creates a habitat matrix of multiple habitat types may support the greatest diversity of bird species.

INTRODUCTION

Birds in North America have suffered persistent and widespread population declines over the past 50 years. In a highly publicized paper, Rosenberg et al. (2019) reported trends that indicated 2.9 billion birds have disappeared from the continent since 1970, representing 29% of all individuals. These declines represent birds across multiple habitat types and life history traits. Grassland birds were the most affected group, exhibiting a 53% total loss and 74% of species in decline. Eastern forest birds have not shown declines as steep as grassland birds, but the trend is still concerning, with a 17% population decrease. The decline in populations of generalist species may be even more telling of population trends. A group of 38 habitat generalist species showed a 23% decrease in the same time frame, suggesting that even the most adaptable species are having trouble conforming to human-altered landscapes. Many factors influence these population trends, the most consistent across all regions being habitat loss (Rosenberg et al., 2019). Other important factors include increased pesticide use, exotic species, building collisions, predation by cats, emerging diseases and global climate change (Faaborg et al., 2010).

In the Midwest, the two primary threats to breeding birds are habitat loss and fragmentation, which are closely associated (Robinson et al., 1995). Fragmented habitats disrupt the interconnectivity of populations and may serve as population sinks for some specialist species. Agriculture is a main driver of habitat loss in this region, but urbanization and industrial land uses, such as mining, also have high impacts. For example, more than 95% of all tallgrass prairies were converted to agricultural land during the 19th and early 20th centuries, and remaining patches are often too small and isolated to support grassland specialists (Johnson & Igl, 2001; Powell, 2008).

Surface mining is a major form of land disturbance in the United States. Surface mining has resulted in the destruction of over 2.4 million hectares of terrestrial habitat since the 1930s (Lemke et al., 2013). Mining is distinct from most other disturbance types because of its comprehensive impact on ecosystems. Surface mining, in particular, changes the entirety of the ecosystem structure starting at the soil level. Soil horizons and pH levels in mined soils can take decades or centuries to return to suitable conditions for the original plant community (Skousen et al., 1994). The long-term impacts of mining on vegetation and wildlife communities are determined by the initial reclamation efforts on the mined site, which are highly variable depending on when the mining occurred. Land mined before the passing of the Surface Mining and Control Act (SMCRA) in 1977 was more likely to be abandoned to natural succession (Holl et al., 2018; Skousen et al., 1994; SMRCA, 1977). Following the passage of the SMCRA, the key reclamation objectives are typically to restore soil horizons and vegetation structure to the original status after mining operations are completed. Mined lands are often reclaimed with herbaceous plants because soil conditions and compaction from large machinery prevent tree regeneration (Lautenbach et al., 2020). These grasslands are dominated by seeded plants, usually exotic cool season grasses and legumes, for at least 20 years after reclamation (Rummel & Brenner, 2003). Percent biomass of seeded species is positively related to topsoil depth during the reclamation process (Pinchak et al., 1985). This suggests that older pre-

SMCRA mined lands are more vulnerable to aggressive successional species with high environmental tolerances.

As land uses, such as agriculture and mining, and habitat fragmentation continue to change the landscape, it is important to understand the response of bird communities to habitat disturbance. Reclaimed mined lands can benefit a wide variety of wildlife including birds, small mammals, reptiles, and amphibians (Carrozzino et al., 2011; Rummel & Brenner, 2003). Restoration principles implemented for post-mined landscapes include establishing suitable soil for the target plant species, providing seed sources for recolonization, using non-aggressive ground cover and planting a variety of species (Holl et al., 2018). Mined lands are difficult to restore to original habitat conditions because of the scope of the disturbance and due to their poor soil conditions (Wali, 1999). Examples of indicators of successful grassland reclamation in the Midwest include tall ground vegetation, dense ground cover (40–85%), low canopy cover (< 40%), and patch size minimums for target species (Rummel & Brenner, 2003). In reclaimed forests, heterogeneity of the vegetation structure may be the most important factor affecting bird species diversity (Karr, 1968). Reclamation goals for each habitat type are necessary for creating adequate habitat to support associated bird populations and communities (Reiley & Benson, 2020).

Habitat restoration and management is essential to maintain native plant communities, especially in the forest-prairie ecotone of the Midwest. Invasive plants exhibit characteristics that make them highly competitive, such as growth under variable moisture conditions, clonal growth, extended flowering periods, and allelopathy (Cadotte et al., 2006). Mined lands are especially vulnerable to invasion because invasive plants respond positively to disturbance, early successional environments, low diversity of native species and high environmental stress (Lemke et al., 2013). Additionally, woody encroachment continues to threaten grassland ecosystems in the Midwest due to fire suppression, heavy grazing, climate change, and introduction of exotic species (Anadon et al., 2014). Woody cover provides perches for birds, which encourages a positive feedback loop of encroachment through seed defecation from perches (Lautenbach et al., 2020). Eastern Red Cedar (*Juniperus virginiana*) has had a particularly prolific expansion in the forest-prairie ecotone. Though a native species, the growing stock volume of red cedar increased in Kansas by 15,000% from 1965–2010. Eastern Red Cedar not only encroaches on grasslands, but also into forests, suppressing the oak-dominated forests that constitute just 5% of Kansas's land base (Galgamuwa et al., 2020).

Avian community composition often changes dramatically with succession following disturbance. In some cases, reclaimed mined lands can support similar diversity of birds to unmined areas and provide quality habitat for grassland, shrub- and forest-dependent birds (Carrozzino et al., 2018; Graves et al., 2010; Karr, 1968). However, when the percentage of woody cover increases and distance to woodlands decreases, grassland obligate birds are quickly replaced by shrubland species, such as Bell's Vireos (*Vireo bellii*), Northern Cardinals (*Cardinalis cardinalis*), and Indigo Buntings (*Passerina cyanea*) (Graves et al., 2010). Shrubland bird species may remain in these habitats between 10–12 years post-disturbance, though soil loss from surface mining may delay the transition from shrublands to forests (Hollie et al., 2020).

The primary goal of this study was to evaluate the relationships between bird communities, vegetation structure, land use, and mining history on strip-mined land. We described bird diversity and evenness, and modeled species responses to vegetation structure. We tested the impacts of land cover type, the presence of exotic plants, and overall plant structure on the densities of three common shrub-dependent bird species that occur at high densities on mined areas: Bell's Vireos, Northern Cardinals, and Indigo Buntings. We predicted that bird densities would be positively related to shrub vegetation structure, but negatively related to invasive plant cover. Information on bird use of abandoned strip-mined land should guide the prioritization of habitat features in formerly mined landscapes.

METHODS

Study Area

We studied shrub-dependent birds on abandoned mined lands in southeast Kansas, which is part of the Cherokee Lowlands ecoregion. This ecoregion spans Bourbon, Crawford, Cherokee, and Labette counties, totaling about 259,000 hectares (Buchanan & McCauley, 2010). The variable climate is characterized by cold winters and hot, dry summers. Monthly average temperatures ranged from 0.66ºC in January (coldest month; average daily min. -4.55ºC, max 5.94ºC) to 26.72ºC in July (hottest month; average daily min. 21.17ºC, max 32.33ºC) (NOAA, 2023). Average annual precipitation was 121.64 cm, with the most precipitation falling in spring (39.19 cm) and summer (36.09 cm; NOAA 2023).

The native ecosystems in this region included tallgrass prairie with smaller patches of oak-hickory forests. However, over 90% of historical prairie habitat has been converted to row crop agriculture, creating a diverse matrix of croplands, grasslands, and forest. Strip mining activity also played a prevalent role in land use change for this region. Strip mining for coal occurred from the 1860s to the early 1970s using a variety of methods, with the majority using large electric draglines (Kansas Geological Survey, 2021). All mining activity in this region ceased in the face of incoming federal legislation for reclamation and restoration of mined lands (SMCRA, 1977), so many of these areas were abandoned to natural succession. The enormous electric shovels used for strip mining created a landscape of alternating overburden piles and water-filled pits that is still prevalent on mined lands today. The pits and overburden piles range in size from 2 m to 20 m deep/tall. The variability in terrain, in conjunction with dense vegetation, makes many mined areas impractical or unsafe to traverse by foot for ecological surveys.

After the cessation of mining activity, the Pittsburg & Midway Coal Company donated a large portion of their land to the state of Kansas, which resulted in the creation of the Mined Land Wildlife Areas (MLWAs). The majority of our study sites were on the MLWAs and other public lands in southeast Kansas (Figure 1.1). Sites were primarily located in Cherokee (n=63) and Crawford (n=9) counties in Kansas. In addition, we selected sites in the adjacent Barton (n=9) and Jasper (n=3) counties in Missouri. The MLWAs consist of 47 individual units totaling 5,868 hectares, including 1,619 hectares of grassland, 3,642 hectares of forest, and 607 hectares of open water. All but 809 hectares of the property was mined (KDWP, n.d.). We determined the mining history of these areas with a combination of characteristics. The most obvious indicator of mining history being the presence of strip pits. Other indicators included lack of topsoil and location on geological maps (Kansas Geological Survey 2021). The MLWAs included a wide range of successional stages due to the 100-year range of mining activity and varied restoration practices. We classified habitat types across the study region as forests, grasslands, or rangelands. Rangelands were classified as any area that was observed to be pasture for livestock during any part of the study. Forests were characterized by having thin rows of overburden piles and pits running through the entire area (Figure 1.2). Grasslands and rangelands were typically graded flat and had much deeper and wider pits (Figure 1.3). Now under the management of the KDWP, a variety of management practices were used on the majority of study sites including prescribed burns, native grass restoration, water level management, mowing, food plots, and livestock grazing (KDWP, n.d.).

Site Selection

We identified 84 point count locations, twenty of which were located in forests, 37 in grasslands and 18 in rangelands. The distribution of sites between habitat types was determined by availability, with far fewer rangelands available and many forests MLWAs unsuitable for this study. Sites were selected to achieve representative spatial coverage of the region while allowing for accessibility. Prior to sampling, each site was visited to evaluate the location for accessibility, noise, habitat type, and any other factors affecting suitability for the project. To select point count sites, we overlaid a 100 x 100 m grid on Google Earth satellite view, assigned a number to each box on the grid, and used a random number generator to select the box where the point count location would be located. Grids were placed 100 m from any habitat borders to prevent bias from adjacent habitats. Points were placed 200 m apart in forests and 250 m apart in grasslands and rangelands to avoid double-counting individuals (Hutto et al., 1986). We placed sampling locations farther apart in grasslands and pasture because noise carries further in those habitats.

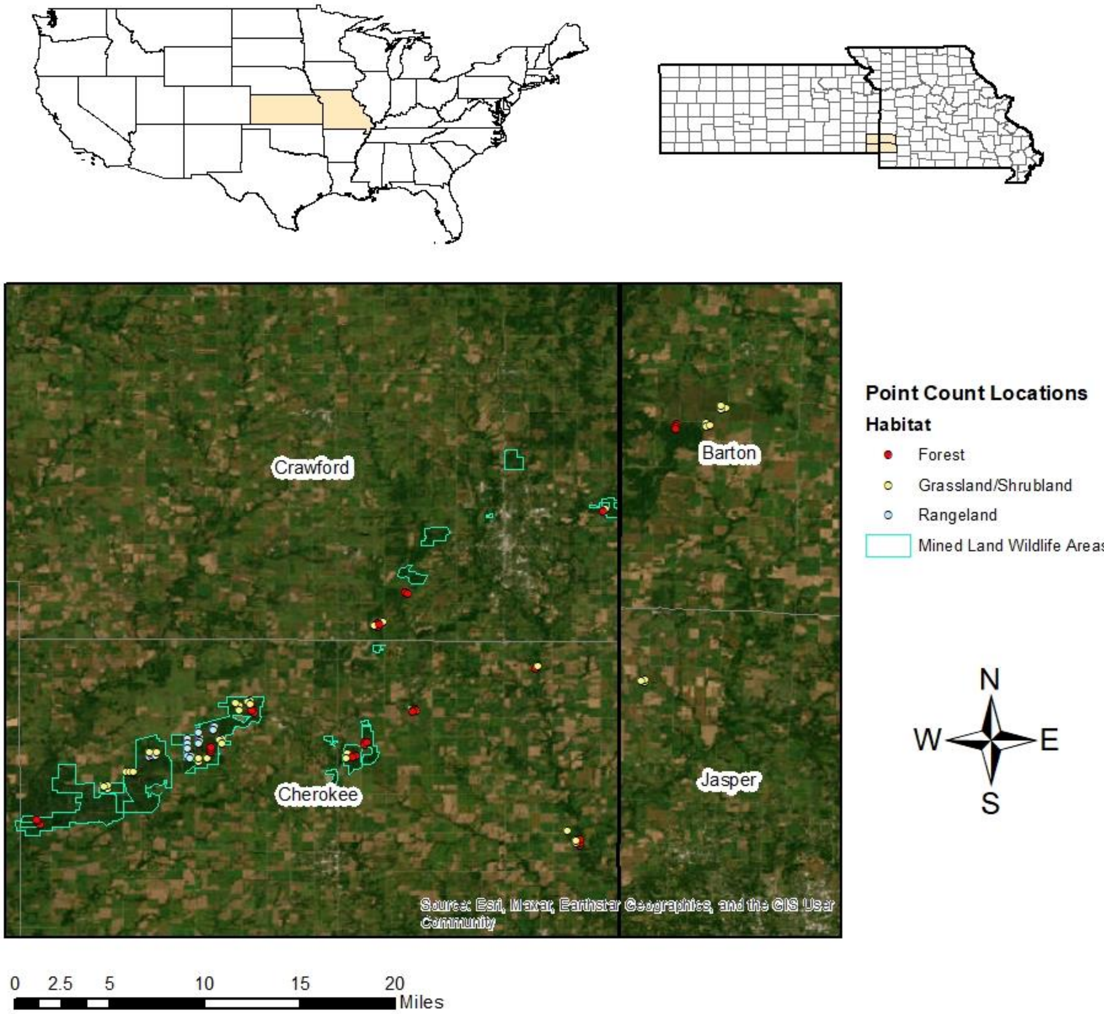

Figure 1.1. Map of surveyed mined land areas in southeast Kansas (Crawford and Cherokee counties) and southwest Missouri (Barton and Jasper counties). Point count locations are represented by their habitat categories.

Figure 1.2. Aerial view of the typical landscape of a forested unit on the Mined Land Wildlife Areas. Photograph from LJWorld.com by Mike Belt, 2007.

Figure 1.3. Typical MLWA habitats: MLWA 40 grassland (top photograph, foreground), MLWA 21 rangeland (top photograph, background), and MLWA 17 managed grassland, with MLWA 17 forest fragments in the distance (bottom photograph).

Field Methods

We performed five-minute fixed-radius point counts at each site three times during the breeding season (May 16–June 30) for three years (2020–2022) (Bibby et al., 2000). Counts were conducted from sunrise until four hours post sunrise. We did not conduct point counts if wind speeds surpassed 8 km/h, during sustained rain, in temperatures above 35ºC, or if other noisy conditions occurred (i.e., construction or road noise; Buckland, 2006). Counts were broken into four distance classes 0–24 m, 25–49 m, 50–99 m, and 100+ m (Ralph et al., 1995). We recorded the following detection variables with every point count: site ID, date, observer, start time, end time, cloud cover, wind speed, air temperature, visit number, and any additional notes specific to that visit. Before conducting a point count, we became familiar with the site by identifying landmarks for each of the distance classes with a rangefinder. Sites were approached as quietly as possible to avoid flushing birds. If any birds were flushed, a note was made of species and distance from point count location. The environmental readings, wind speed, air temperature, and cloud cover were collected prior to the count to give the area time to quiet down and accustom birds to the presence of the observer, typically two minutes. For each bird detected, we recorded time of detection, species (alpha code), distance class from observer, type of detection (i.e., fly through, seen, or heard), cardinal direction of the detection, and additional notes such as breeding behavior. If a flock was too large to count, we recorded an estimated range. During collection periods with multiple observers, we alternated point count sites between observers to minimize bias.

We used a variety of methods to collect vegetation data within a 11.3-m radius circle centered on each point count site (James & Shugart, 1970). We sampled five random locations using a Daubenmire frame (30 x 50 cm) to estimate ground cover of artificial surface, bare soil, forbs, grass, leaf litter, rock, shrubs, trees, woody litter, and water (Bonham et al., 2004). We identified trees (> 8 cm DBH) and shrubs (< 8 cm DBH, > 1 m height) to species, and classified the species as exotic or invasive according to the Kansas Forest Service's invasive species list (Kansas Forest Service, n.d.) and the Kansas Department of Agriculture's noxious weed list (Kansas Department of Agriculture, n.d.). We measured tree diameter-at-breast-height (DBH) using a D-tape (Metric Fabric Diameter Tape, Forestry Suppliers Inc., 205 W Rankin St, Jackson, MS 39201). Snag trees were included in these measurements because they are important habitat features for cavity nesting species. We visually estimated percent shrub cover of the plots and measured vertical vegetation density using a Nudd's board (Nudds, 1977). The board was placed in the center of each plot and the observer recorded how much of the board's five sections was covered by vegetation when viewed from 11.3 m away and from each cardinal direction. Tree canopy cover was estimated with a spherical densiometer (Spherical densiometer, Model-A, Forest Densiometers, 10175 Pioneer Ave Rapid City, SD 57702-4756). We also measured grass or other dominant ground cover height with a meter stick.

Statistical Analysis

We used generalized linear mixed models within an information theoretic framework to examine relationships between habitat features and densities of our focal species (Burnham and Anderson 2002, Bolker et al 2009). To account for differences in detection probability, we first estimated p (availability) and q (perceptibility) for each species and visit using time-removal models and distance models, respectively, based on the raw point count values binned by distance and time interval (Sólymos et al., 2013). The log product of both parameters was then included as an offset term in Poisson models with raw point counts as the response variable and different combinations of habitat characteristics as the predictor variables. We included a random intercept in all models to account for non-independence of point counts from the same sites. The candidate models represented *a priori* hypotheses regarding effects of grass height, invasive shrub cover, invasive forb cover, shrub cover, basal area, vertical vegetation density, habitat type (i.e., grassland, rangeland, forest), and past mining occurrence (i.e., mined or unmined). Correlated variables ($r > 0.7$) were excluded from the same models. We then ranked and sorted models using Akaike's Information Criteria (AICc) and model weights. Informative models with ΔAICc < 2 were considered supported (Burnham and Anderson, 2002, Arnold, 2010). We also calculated Shannon diversity of each habitat using package vegan (Oksanen et al., 2022). All analyses were conducted in Program R, version 2021.09.0 (R Core Team, 2021).

RESULTS

We recorded 87 bird species from 7,999 total detections during point counts (Appendix I), including 13 species of conservation concern for southeast Kansas counties (Appendix II). Our most frequently detected species was the Dickcissel *(Spiza americana*), with a total of 1286 detections. We detected 276 Bell's Vireos, 757 Northern Cardinals, and 496 Indigo Buntings. Across habitats, forests had the highest Shannon diversity score (H' = 2.81) (Figure 1.4).

Our best-supported density models for all three shrub-dependent species indicated that habitat type was the leading variable in predicting their densities (Table 1.1). Bell's

Vireos occurred at similar densities on grasslands and rangelands, with no individuals detected in forest habitats (Figure 1.5; Table 1.1). Northern Cardinals had the highest densities in forest habitat, intermediate densities on grassland habitat, and the lowest densities on rangeland habitat (Figure 1.6; Table 1.2). Indigo Buntings had highest densities on grasslands, intermediate densities in forests, and lowest densities on grasslands (Figure 1.7, Table 1.2).

Vegetation structure in forested locations differed from that in either rangeland or grassland, but was similar between grassland and rangeland. Forested areas had the greatest canopy cover, invasive shrub cover, basal area, vertical vegetation cover, and overall shrub cover (Table 1.3). Grassland and rangeland had similar vegetation makeup with a few notable differences. Grassland had the tallest grass height, double that of rangeland, as well as the greatest invasive forb cover, which was made up of over 90% Sericea Lespedeza (*Lespedeza cuneata*) (Table 1.3). The most common invasive shrub species overall were Bush Honeysuckle (*Lonicera maackii*), Japanese Honeysuckle (*Lonicera japonica*), and Autumn Olive (*Elaeagnua umbellate*). Grassland also had greater vertical vegetation density and shrub cover than rangeland (Table 1.3).

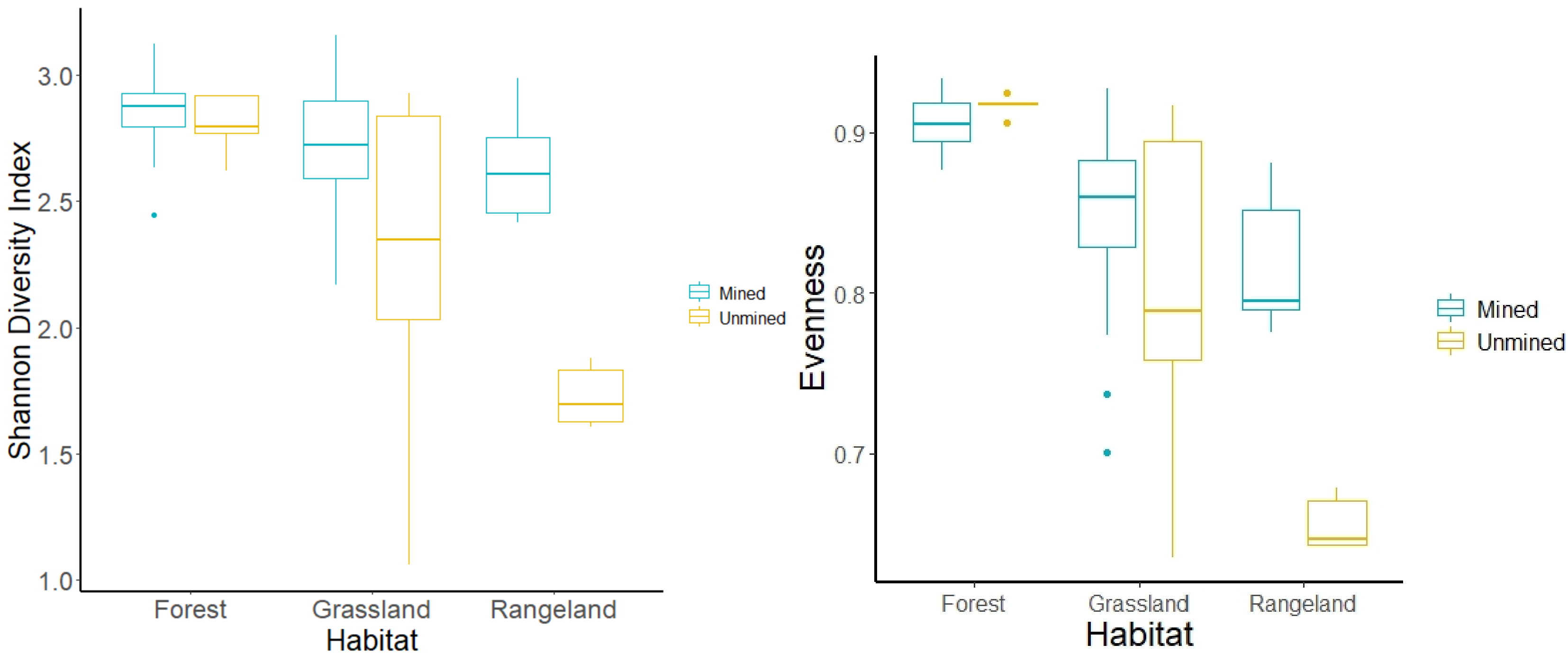

Figure 1.4. Shannon diversity index and species evenness of point counts across habitat type and mining history. Forests (mined n=26, unmined n=4), Grassland (mined n=25, unmined n=11), Rangeland (mined n=15, unmined n=3).

Table 1.1. A priori candidate models for Bell's Vireo, Northern Cardinal, and Indigo Bunting densities, estimated from point count data. Models included habitat types (i.e., forest, grassland, rangeland), basal area, canopy coverage (out of 100%), grass height ("Grass"), mining history (i.e., mined vs. unmined), shrub coverage ("Shrub"), invasive shrub coverage ("Invasive"), invasive forb coverage ("Forbs"), vertical vegetation density ("Nudds"), and forb ground coverage (i.e. forb, grass, shrub). Each model's ΔAIC, parameters (K) and weights (wi) are included.

Species	Model	ΔAICc	K	w*i*
Bell's Vireo	Habitat	0	4	0.87
	Basal area	3.77	3	0.13
	Canopy	16.34	3	0.00
	Soil depth	32.26	3	0.00
	Invasive	40.31	3	0.00
	Mining	50.19	3	0.00
	Grass	50.49	3	0.00
	Shrub	52.42	3	0.00
	Nudds	53.01	3	0.00
	Forbs	53.06	3	0.00
	Null	53.77	2	0.00
Northern Cardinal	Habitat	0	4	1.00
	Nudds	24.63	3	0.00
	Basal area	36.53	3	0.00
	Invasive	51.20	3	0.00
	Mining	53.08	3	0.00
	Grass	57.34	3	0.00
	Forbs	58.95	3	0.00
	Soil depth	58.95	3	0.00
	Null	58.96	2	0.00
Indigo Bunting	Habitat	0	4	0.74
	Mining	3.23	3	0.15
	Grass	6.40	3	0.03
	Forbs	8.01	3	0.01
	Basal area	8.02	3	0.01
	Soil depth	8.03	3	0.01
	Nudds	8.08	3	0.01
	Shrub	8.08	3	0.01
	Null	8.08	2	0.01

Table 1.2. Coefficients of the best-supported models for Bell's Vireo, Northern Cardinal, and Indigo Bunting, based on point count data. The beta coefficients, standard errors (SE), and the lower and upper 95% confidence intervals (CI) for each parameter are included. Forest habitat is the reference level.

Species		Coefficient	SE	Lower 95% CI	Upper 95% CI
Bell's Vireo	Intercept	-6.96	1.05	-9.02	-4.90
	Grassland	4.84	1.07	2.74	6.94
	Rangeland	5.00	1.10	2.84	7.16
Northern Cardinal	Intercept	-0.67	0.08	-0.83	-0.51
	Grassland	-0.62	0.11	-0.84	-0.40
	Rangeland	-1.31	0.15	-1.60	-1.02
Indigo Bunting	Intercept	-1.23	0.11	-1.45	-1.01
	Grassland	0.21	0.14	-0.06	0.48
	Rangeland	-0.29	0.18	-0.64	0.06

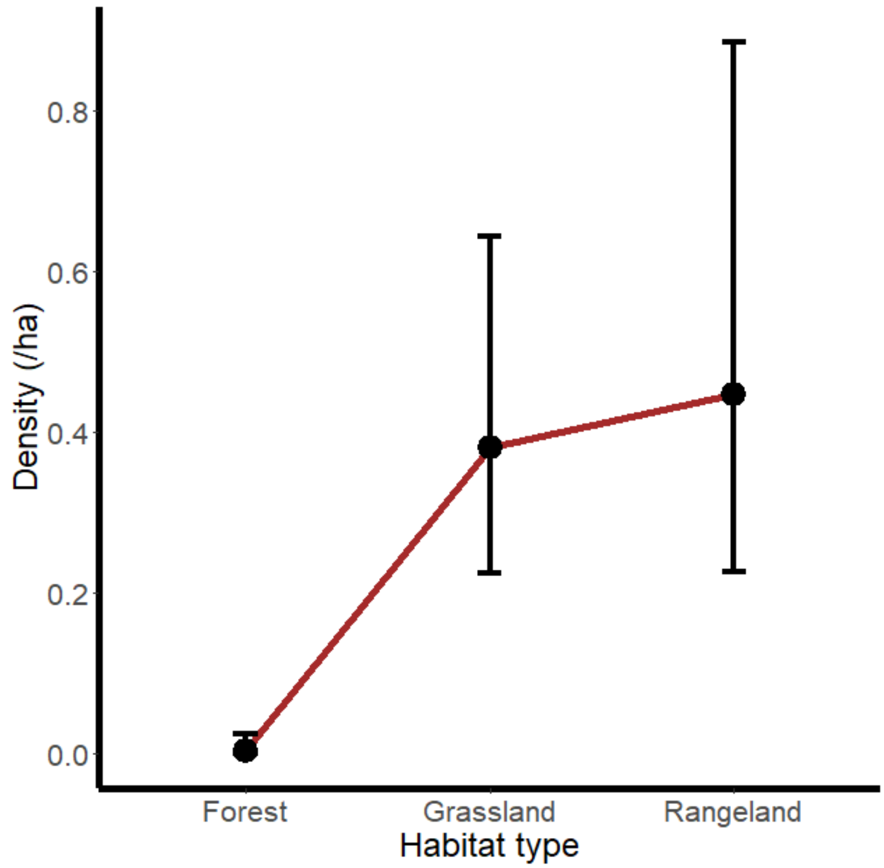

Figure 1.5. Model predictions from the best-supported models of the effects of habitat type on Bell’s Vireo densities in southeast Kansas. Error bars indicate 95% confidence intervals.

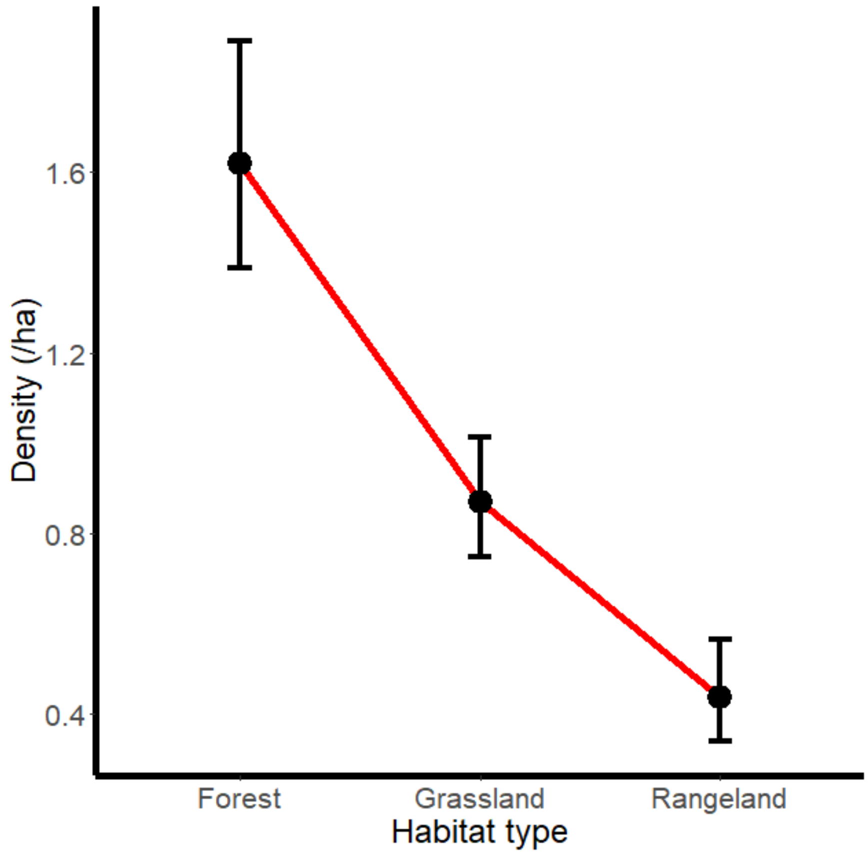

Figure 1.6. Model predictions from the best-supported models of the effects of habitat type on Northern Cardinal densities in southeast Kansas. Error bars indicate 95% confidence intervals.

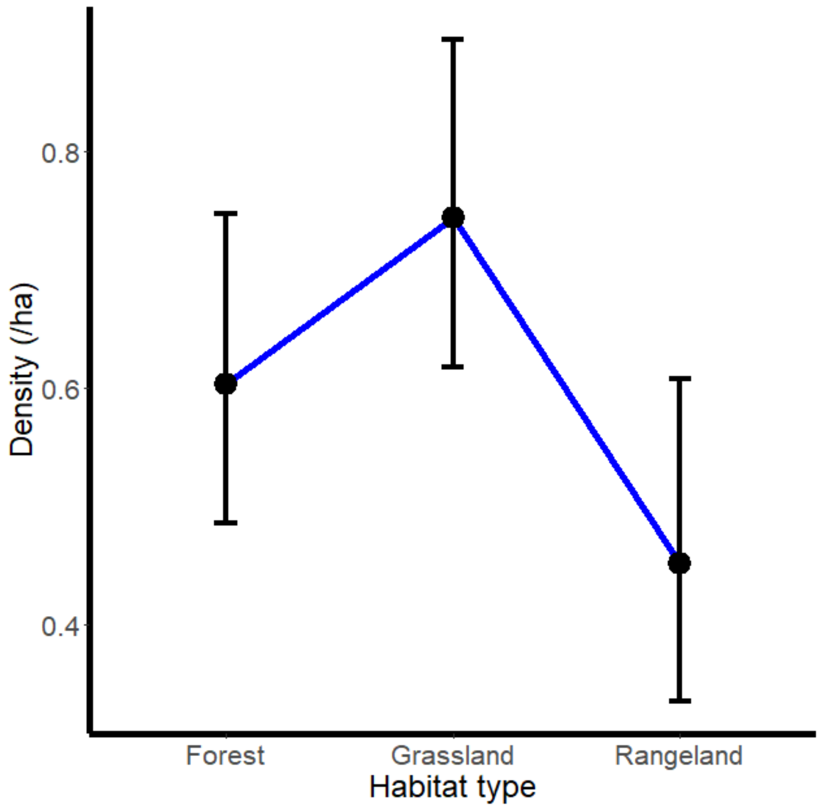

Figure 1.7. Model predictions from the best-supported models of the effects of habitat type on Indigo Buntings densities in southeast Kansas. Error bars indicate 95% confidence intervals.

Table 1.3. Mean vegetation values sampled each year at point count locations across three sampled habitat types. The number of point count locations surveyed for each habitat type are indicated in the first row.

Habitat Feature	Forest (n=30)	Grassland (n=39)	Rangeland (n=15)
Canopy Cover (%)	80.18 ± 26.84	4.55 ± 14.71	0.98 ± 4.43
Grass Height (cm)	32.84 ± 39.96	74.64 ± 48.33	38.82 ± 20.72
Basal Area	72.09 ± 39.11	3.63 ± 13.32	2.77 ± 10.41
Invasive Tree Cover (%)	0.77 ± 2.02	0.14 ± 0.64	0.06 ± 0.31
Shrub Cover (%)	42.64 ± 25.38	13.54 ± 13.87	9.53 ± 14.46
Invasive Shrub Cover (%)	17.92 ± 22.99	1.88 ± 4.20	0.57 ± 1.46
Invasive Forb Cover (%)	4.19 ± 9.01	5.44 ± 12.69	3.37 ± 6.33
Vertical Vegetation Density (%)	63.81 ± 24.05	38.97 ± 22.13	18.56 ± 16.70

DISCUSSION

We found that habitat type was the best predictor of population density for all three focal shrub-dependent bird species. Models with mining history were not supported, suggesting that whether a site was mined or not was less important than other characteristics. Habitat type, as determined by a variety of vegetation metrics, was the best explanation for bird diversity. Understanding how past land use history and current habitat conditions influence bird communities is imperative for habitat managers to effectively manage specific species or groups of birds.

There have been extensive studies globally of the relationships between bird community diversity and habitat (Goetz et al., 2014; Reif et al., 2022; Tu et al., 2020). In our study, forests had higher estimates of Northern Cardinal densities and overall diversity. One explanation for the lower species diversity on grasslands is the overall size and fragmentation of the grasslands in the study region (Herkert, 1994). Grassland units on the MLWAs were typically between 20 and 50 ha, while the minimum recommended size for a continuous grassland patch to support sensitive grassland species (e.g., Henslow's Sparrows, *Ammodramus henslowii*, and Grasshopper Sparrows, *Ammodramus savannarum*), range from 10–200 ha (Herkert, 1994; Vickery et al., 1994). The exception to this rule is Dickcissels, as they are less sensitive to grassland patch size (Herkert, 1994). This explains why Dickcissels were our most detected bird and also why evenness was lower on grasslands, as high densities of Dickcissels may have resulted in overall lower diversity. If area effects are contributing to the lower bird diversity that we observed on formerly mined grasslands, then management to increase the size and connectivity of grassland patches, may support more grassland obligate birds. Many mined areas had either hedgerows of trees running through grasslands or sparse numbers of trees scattered throughout. Decreasing the amount of woody vegetation would be beneficial to a number of species that we detected in low densities, such as Eastern Meadowlarks (*Sturnella magna*), Grasshopper Sparrows and Henslow's Sparrows. For forest habitats, there are limited management options for abandoned mined sites in this region. The difficult terrain and soil conditions make fire and mechanical techniques difficult and other approaches, such as regrading the surface to reseed vegetation may not be economically viable.

We observed major differences in vegetation characteristics between forested units and both grasslands and rangelands, with only minimal differences between grasslands and rangelands. Habitat designations for this project included a wide range of vegetation characteristics in each habitat. The designation of habitat types was difficult to distinguish between heavily shrubbed prairie, late stage shrubland and early successional forest. Using this designation system contributed to some of the high variation we observed in diversity differences between habitat types. Invasive shrubs made up a large portion of shrub cover in forests. However, we did not observe relationships between invasive plant densities and shrubland bird densities. Invasive plant species have complicated interactions with native bird species, with both positive, neutral and negative interactions (Maresh Nelson et al., 2017). Species such as bush honeysuckle, which was the most common invasive species in our forested habitats, often create monocultures. Even so, monocultures of invasive shrubs may not be harmful to bird species like Northern Cardinals, which often use bush honeysuckle as a food source (Ingold & Craycraft, 1983).

Northern Cardinals are typically associated with habitats characterized by shrubs and small trees, such as forest edges and openings within patch interiors (Halkin & Linville, 2021). Our findings support this habitat association, as we observed the greatest Northern Cardinal densities in forested units, and lower densities on grasslands and rangelands. Of the three focal species, Northern Cardinals have the least conservation and habitat concerns, as they adapt well to altered and anthropogenic landscapes and select for shrubby forest habitats like those found throughout previously mined lands.

Bell's Vireos were equally abundant on grassland and rangeland sites and absent from forested sites, which was expected, as they are a shrubland obligate species (Budnik et al., 2000). Bell's Vireos rely on grassland-shrub habitat that has largely been removed from the landscape with the removal of associated prairie habitat (Budnik et al., 2000). Early-successional wildlife habitat is largely overlooked and has become increasingly uncommon (King and Schlossberg 2014, DeGraaf & Yamasaki, 2003). Management of abandoned mined lands may provide opportunities to protect large areas of early-successional habitat. Formerly mined areas of the Midwest may be especially responsive to management because of the matrix of grassland and adjacent forested sites that harbor large populations of difficult to manage shrubs. Management that prioritizes the conservation of shrubland habitats will benefit shrub-dependent species such as Bell's Vireos. Approaches such as rotational prescribed burns and mechanical control could create a habitat matrix ranging from grassland to early successional forests.

Of our three focal species, densities of Indigo Buntings varied the least between habitat types, with the highest densities in grasslands. Indigo Buntings are associated with forest edges, and they prefer habitats with complex patch shapes. Thus, it is not surprising that Indigo Buntings showed the lowest variation among habitats; smaller, irregularly shaped patches were abundant throughout the formerly mined sites in our study area (Weldon & Haddad, 2005). Similar to Northern Cardinals, Indigo Buntings are a common songbird species throughout the Midwest and on mined lands.

CONCLUSION

Reclamation efforts following intense human disturbances can supply habitat for a wide variety of wildlife. Even with minimal restoration efforts, the strip-mined land in our study region hosts considerable habitat variation and associate species diversity. We observed that habitat type was the best model for predicting density of three shrub-dependent bird species. While managing for shrubs in restored mined lands may not be suitable for all species, focusing efforts to improve habitats for shrub-dependent species of conservation concern could benefit bird diversity overall. Formerly mined lands provide an excellent opportunity to manage a diverse habitat matrix that may benefit a wide range of species throughout the region.

CHAPTER II

NEST SUCCESS OF SHRUB-NESTING BIRDS IN A POST-MINED LANDSCAPE

ABSTRACT

As bird populations continue to decline in North America, it is important to understand the benefits that disturbed habitats can have for breeding birds. In this study, we tested how daily nest survival (DSR) responded to land use, habitat type, and vegetation characteristics across forest, grassland, and rangeland sites in a formerly strip-mined landscape. We searched for and monitored nests of shrubland birds at 84 sites in southeast Kansas and southwest Missouri. We located 178 nests, the majority of which belonged to our focal species: Bell's Vireo (*Vireo bellii*, 69% of nests), Northern Cardinal (*Cardinalis cardinalis*, 16%), and Indigo Bunting (*Passerina cyanea*, 4%). Logistic exposure models estimated daily nest survival for Bell's Vireo as a function of habitat type, with grasslands having a DSR of 94% and rangelands a DSR of 89%. This relationship could be the result of 10% more invasive plant cover and 17% higher chance of Brown-headed Cowbirds (*Molothrus ater*) brood parasitism in rangeland habitats. Northern Cardinals had an average DSR of 91%, with nest age negatively associated with their DSR. If demographic rates are consistent across the study region, Bell's Vireo reproductive rates are not high enough to maintain their populations.

INTRODUCTION

Surface mining is one of the most destructive forms of land use, and has resulted in over 2.4 million hectares of terrestrial habitat disturbance in the United States since the 1930s (Lemke et al., 2013). Mining is distinct from most other disturbance types because of the comprehensive impacts on ecosystems. Surface mining disturbance is thorough and persisting because it destroys both the soil and vegetation. Soil horizons and pH levels can be altered to the point that it takes decades or centuries for conditions to be suitable for the original plant community to reestablish, which has direct consequences on wildlife habitat (Skousen et al., 1994).

The long-term impacts of mining on vegetation and wildlife communities are determined by the initial site conditions following the conclusion of active mining. A wide range of reclamation efforts may occur on previously mined lands, some of which are a result of the passing of the Surface Mining and Control Act in 1977 (Holl et al., 2018; Skousen et al., 1994; SMCRA, 1997). Current reclamation efforts implemented for mined lands include establishing suitable soil for vegetation regrowth, providing seed sources for recolonization, using non-aggressive ground cover, and planting a variety of species (Holl et al., 2018). Reclaimed grasslands are typically dominated by seeded plants, usually exotic cool season grasses and legumes, for at least 20 years post-reclamation (Rummel & Brenner, 2003). Land mined before the passing of the SMRCA was more likely to be abandoned to natural succession due to the lack of guidelines for its remediation and reclamation. Thus, older pre-SMCRA mined lands are more vulnerable to aggressive successional species with high environmental tolerances.

Most mined lands are reclaimed by seeding the area with quick-growing ground cover species mixed with a wide range of planted tree species, such as hardwoods and pines (Holl et al., 2018). If reclaimed mined lands are not managed as grasslands post-restoration, natural succession will proceed and the area will transition to shrublands and forests due to fire suppression, heavy grazing, and the introduction of exotic species (Anadon et al., 2014). Introduced and naturalized species have increasing impact in encroachment due to their highly competitive traits, tolerance of a variety of moisture conditions, clonal growth, extended flowering periods, and allelopathy (Cadotte et al., 2006). Mined lands are especially vulnerable to invasion because they exhibit habitat attributes that coincide with invasive species establishment such as heavily disturbed soils, early successional environments, low diversity of native species, and high environmental stress (Lemke et al., 2013). To compound this problem, many reclamation efforts included seeding with exotic species that were eventually listed as invasive, including Autumn Olive (*Elaeagnus umbellate*; Oliphant et. al., 2016) and Sericea lespedeza (*Lespedeza cuneata*; Zipper et al., 2011). Other common invasive species on mined lands included, Bush Honeysuckle (*Lonicera maackii*), Japanese Honeysuckle (*Lonicera japonica*), Tree of Heaven (*Ailanthus altissima*), Multiflora Rose (*Rosa multiflora*) and Silktree (*Albizia julibrissin*; Adams et al., 2019; Holl et al., 2018). Eastern Red Cedar (*Juniperus virginiana*), which is a native species, has had particularly prolific expansion in the central and eastern Great Plains. Considered a pioneer species for mined lands, it grows quickly, is well adapted to drought conditions, and has a long growing season (Burns & Service, 1990). For example, the growing stock volume of Easter Red Cedar increased by 15,000% in Kansas between 1965 and 2010 (Galgamuwa et al., 2020). Eastern Red Cedar not only encroaches on grasslands, but also into forests, suppressing native hardwood species (Galgamuwa et al., 2020).

As land use legacies and invasive plant species establishment continue to change the landscape, it is important to identify positive and negative biodiversity impacts of previously disturbed areas. Invasive plants in post-mined lands have a number of concerning ecological effects, as they alter habitat structure, change ecosystem processes and decrease native biodiversity (McNeish & McEwan, 2016). The primary concern is that these aggressive and prolific species will outcompete slower growing native species that have higher ecological value, lowering the overall diversity of the area. Invasive plants can also provide less adequate habitat and forage for invertebrates, resulting in decreased food availability for organisms at higher trophic levels (Love & Anderson, 2020; George et al., 2013). For example, Bush Honeysuckle was consumed by larval insects ten times less than native shrubs in the same environment (Love & Anderson, 2020). Even so, reclaimed mined lands can be beneficial to a wide variety of wildlife, including birds, small mammals, reptiles, and amphibians (Carrozzino et al., 2011; Rummel & Brenner, 2003). While numerous wildlife species do use reclaimed mined lands, habitat use does not necessarily indicate habitat quality (Stauffer et al., 2011). Assessing a species' reproductive effort is imperative to understand its future population trends.

Relationships between exotic plant species and native birds are complex and cannot be easily summarized, as they can have negative, neutral, or positive outcomes, depending on the situation. The ecological trap hypothesis is one of the most discussed ideas as to why invasive plant species can potentially decrease songbird productivity.

This hypothesis describes decreased nest success in exotic plants due to differences in leaf phenology and insect biomass, compared to their native counterparts (Donovan & Thompson, 2001; McChesney & Anderson, 2015; Rodewald et al., 2010). However, other studies indicate that birds may nest in invasive plant species without any negative effects to nesting success (Gleditsch and Carlo, 2014). A recent meta-analysis showed that bird species richness was negatively related to invasive plant densities, while some birds preferred to nest in invasive shrubs, and nest success typically remained neutral (Nelson et al., 2017). Additional ecosystem-specific information is needed to continue developing our understanding of how invasive plant species affect native bird reproduction.

Due to the proliferation of invasive shrub species in post-mined landscapes, we sought to estimate reproductive rates of three shrub-nesting bird species: Bell's Vireos (*Vireo bellii*), Indigo Buntings (*Passerina cyanea*), and Northern Cardinals (*Cardinalis cardinalis*). Several studies have examined relationships between vegetation or habitat relationships and reproductive rates of Northern Cardinals and Indigo Bunting in the Midwest, but few have examined the reproductive success of Bell's Vireos outside of the southwestern United States (Budnik et al., 2002; Chapa-Vargas & Robinson, 2013). Our goal was to identify the habitat and vegetation variables that were most likely to affect reproductive success for our target species in a post-mined landscape. We predicted that areas with higher densities of invasive plants would have lower reproductive rates and areas with high vertical vegetation density would have higher reproductive.

METHODS

Study Area

We studied bird nesting success on post-mined lands in southeast Kansas, which is part of the Cherokee Lowlands ecoregion. This region covers about 259,0002 hectares in Bourbon, Crawford, Cherokee, and Labette counties (Buchanan & McCauley, 2010). The variable climate is characterized by cold winters and hot, dry summers. Monthly average temperatures ranged from 0.66ºC in January (coldest month; average daily min. -4.55ºC, max 5.94ºC) to 26.72ºC in July (hottest month; average daily min. 21.17ºC, max 32.33 ºC) (NOAA, 2023). Average annual precipitation was 121.64 cm with the most precipitation falling in spring (39.19 cm) and summer (36.09 cm) (NOAA, 2023).

Southeast Kansas contains a number of different ecotones, particularly with the transition from oak hickory forests in the east to the tallgrass prairie regions to the west. This ecotone also coincides with increases in intensive row crop agriculture, which has replaced over 90% of the native prairie habitat. In addition to habitat conversion due to agriculture, southeast Kansas was strip mined for coal from 1860–1974 (Kansas Geological Survey, 2021). The mining operations were performed with enormous electric draglines, which created a landscape of alternating overburden piles and water-filled pits that have now revegetated to grasslands, shrublands, or forest. All mining activity in the region ended with new federal legislation for reclamation or restoration of mined lands in the process of being enacted, so many of these areas were abandoned to natural succession (SMRCA 1977). Even so, some restoration actions have occurred to grade the overburden piles and pits to a relatively flat surface, leaving a smaller number of large deep pits instead of numerous narrow pits. After mining activity was completed, the mining companies donated 5,867 ha of land to Kansas Department of Wildlife and Parks, which resulted in the creation of the Mined Land Wildlife Areas (MLWAs) across Cherokee, Crawford and Labette counties.

The majority of the MLWAs were characterized by dense rows of forest, split by shallow strip pits. Forests (3,642 ha) were dominated by Pin Oak (*Quercus palustris*), Black Walnut (*Juglans nigra*) and Eastern Red Cedar. The remaining area was split into 607 ha of open water and 1,618 ha of prairie, shrubland, and rangeland. We classified habitat types on the MLWAs as either forest (35%), grassland (44%), or rangeland (21% sites). The MLWAs are managed using a variety of standard approaches including prescribed fire, livestock grazing, mowing, native plant restorations, wetland restoration and mechanical removal of vegetation. However, the majority of the forested sites remain inaccessible to management due to the complex strip pit topography.

We surveyed 84 sites for bird nesting activity. Most of our study sites were located in the MLWAs and other public lands in southeast Kansas. In Kansas, we surveyed sites in Cherokee (n = 63) and Crawford (n = 9) counties, including focused efforts on MLWAs 4, 9, 12, 14, 17, 21, 38, 40 41, 42, and 45, as well as the Buche Wildlife Area, Monahan Outdoor Education Center, and Spring River Wildlife Area. We also surveyed sites in Missouri in Barton (Prairie State Park, n = 9) and Jasper (Wah-Sha-She Prairie State Natural Area, n = 3) counties. To identify study sites, we first scouted the location to evaluate each area for safety and accessibility. Then, we identified areas with high densities of our three focal species early in the point count season and from previous years' experience (Chapter 1). Sites were selected on a year-to-year basis due to the high occurrence of prescribed fire and brush clearing activity, which caused shifts in local densities of shrub-nesting birds, especially Bell's Vireo.

Field Methods

We searched for and monitored nests according to the BBird protocol (Martin et al., 1997). We searched for nests from mid-May through late July in 2020–2022. We used two primary methods to find nests: systematic searching of nesting habitat and observing bird behavior. The behaviors we looked for during nest searching included carrying nesting material or food items, carrying fecal sacs and defending a territory. We also found nests opportunistically during point count surveys or by flushing birds.

When a nest was found, we recorded the species, a location description to aid in finding the nest again, GPS coordinates, search method, date, time, observer, nesting stage, contents of nest and any comments. Once found, the nest was checked every 2–3 days. On each subsequent check, we recorded the date, time, observer, time at nest, nest stage, nest contents, adult location and activity, and the minimum and maximum age of the nestlings, if any. When possible, we checked the nests from a distance to minimize potential stress and likelihood of abandonment. We took different paths during each nest check to avoid creating a physical or scent trail leading to the nest. Upon nest fate completion (i.e., fledging of young, depredation, or abandonment), we attempted to determine nest fate and collected vegetation data. The fate clues we primarily looked for were presence of fledglings in the area, aggression from parents, and fecal sacs on the rim of the nest or below it.

We used a Daubenmire frame (30 x 50 cm) to estimate ground cover in vegetation plots at each nest (Bonham et al., 2004). The cover classes included artificial surface, bare soil, forbs, grass, leaf litter, rock, shrubs, trees, woody litter, and water. Trees (> 8 cm DBH) and shrubs (< 8 cm DBH, > 1 m height) were identified to species. Exotic and invasive plants were classified according to the Kansas Forest Service's invasive species list (Kansas Forest Service, n.d.) and the Kansas Department of Agriculture's noxious weed list (Kansas Department of Agriculture, n.d.). Each tree's diameter-at-breast-height (DBH) was measured with a D-tape (Metric Fabric Diameter Tape, Forestry Suppliers Inc., 205 W Rankin St, Jackson, MS 39201). Shrubs were not counted individually; instead, the percent total cover of the plot was visually estimated. We measured vertical vegetation density using a Nudd's board (Nudds, 1977). The board was placed in the center of each plot and the observer recorded how much of each section of the board was covered by vegetation standing 11.3 m away in each cardinal direction. Tree canopy cover was estimated with a spherical densiometer (Spherical densiometer, Model-A, Forest Densiometers, 10175 Pioneer Ave Rapid City, SD 57702-4756). Grass or other dominant ground cover height was recorded as well. We also recorded the nest substrate species and any other climbing or intertwined plant associated with the substrate plant, substrate height, nest height, and nest orientation. Nest cup visibility was measured using a 6.5 cm plastic disc divided into eight alternating black and white sections. We took measurements by placing the disc in the nest cup and recording how many of the sections were covered from directly above and from 1 m away for each of the four cardinal directions (Stauffer et al., 2011).

Statistical Analysis

We estimated daily survival rates (DSR) of nests following methods from Rotella et al. (2004). Estimates of nest success are an important metric for evaluating habitat management strategies and an essential component of demographic modeling (Jehle et al., 2004; Rotella et al., 2004). There are many ways to estimate nest success. Daily survival rate (DSR) calculates the probability that an individual nest will survive any given day. To be included in nest survival analysis, nests had to contain at least one host species egg for two or more nest checks. We used logistic exposure and program MARK to examine the relationships between DSR and habitat characteristics. Models were then ranked and sorted using Akaike's Information Criteria (AICc). Models within 2 ΔAICc were considered informative (Arnold, 2010). We included both null and global models in candidate model sets. We tested a variety of predictor variables including habitat type, nest age, proportion of shrub coverage on the nest vegetation plot, proportion of invasive shrub cover on the plot, invasive or native nest substrate, nest height, and year (Table 2.1). Correlated predictor variables ($r > 0.7$) were excluded from the same models. All models were fit and ranked in Program R, version 2021.09.0 (R Core Team, 2021) using package MARK (White and Burnham, 1999).

Table 2.1. Description of daily nest survival model variables.

Variable	Habitat Characteristic
NestAge	Age of nest in days
Time	Time series since day 1 of monitoring
PpnInv	Proportion of invasive vegetation within nest plot
NuddsAVG	Average vertical vegetation density as measured with a Nudds board
PpnShrub	Proportion of shrub coverage within nest plot
Year	Year
Habitat	Habitat Type (i.e., grassland, rangeland, forest)

RESULTS

From 2020–2022, we monitored 178 nests that included 159 nests of our three focal species (Table 2.2). We only found eight nests from Indigo Buntings, none of which were successful. Thus, we excluded Indigo Buntings from the analysis due to a lack of statistical power.

The leading cause of failure for Bell's Vireo nests was depredation, followed closely by nest parasitism (Figure 2.1; Table 2.2). Of all observed species, Bell's Vireo had the most nest failures due to livestock disturbance (Figure 2.1; Table 2.2). Bell's Vireo nests were found in only two of the three habitat designations for our study: grassland (n = 90) and rangeland (n = 32) (Table 2.3). Bell's Vireo nests had an average DSR of 93% (Figure 2.2). Habitat plus nest age was the best-supported model for DSR (Table 2.4). Nests in grasslands had an average DSR of 94%, while nests in rangeland had a DSR of 89% (Figure 2.3). On average, Bell's Vireo nests had 23.1% shrub cover on the nest vegetation plot, 13.7% of which was invasive shrub cover. Only ten (8%) of nests were placed in an invasive substrate (i.e., Bush Honeysuckle, Black Locust [*Robinia pseudoacacia*], and Autumn Olive). Brown-headed Cowbirds parasitized 50% of Bell's Vireo nests with at least one egg (Table 2.2). Nest parasitism was more likely to occur on rangeland habitat, with 66.7% of nests being parasitized compared to 46.7% on grassland. Damage from livestock was the cause of 4% of Bell's Vireo nest failures, all of which occurred on rangeland habitat. Additionally, rangelands had 9.7% greater invasive plant cover around the nests than those in grassland habitats.

Table 2.2. Number of nests detected with our nest searching efforts 2020–2022. Values include the number of nests that succeeded in fledging young and those that failed, listed by cause of failure.

		Failure Cause					
Species	Successful	Depredation	Parasitized	Livestock	Weather	Unknown	Total
Bell's Vireo	21	58	33	5	3	2	122
Northern Cardinal	6	19	3	0	0	1	29
Dickcissel	3	8	1	0	0	0	12
Indigo Bunting	0	6	1	1	0	0	8
Common Nighthawk	1	2	0	0	0	0	3
Scissor-tailed Flycatcher	1	1	0	0	0	0	2
Lark Sparrow	0	1	0	0	0	0	1
Kentucky Warbler	0	1	0	0	0	0	1
Total	32	96	38	6	3	3	178

Figure 2.1. Examples of a Bell's Vireo nest failure due to livestock disturbance on MLWA 21 in 2022 (left) and Brown-headed Cowbird nest parasitism and depredation on MLWA 17 in 2021 (right).

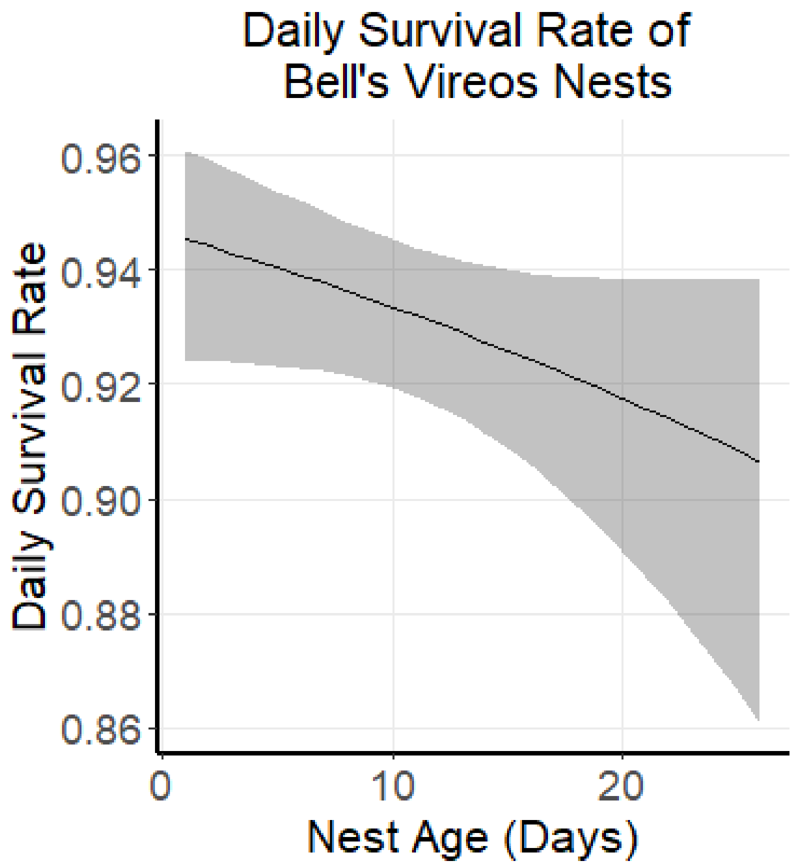

Figure 2.2. Daily survival rate of Bell’s Vireo nests (n=122) over the 26-day nesting period in southeast Kansas. Shaded regions represent 95% confidence interval for daily survival rate.

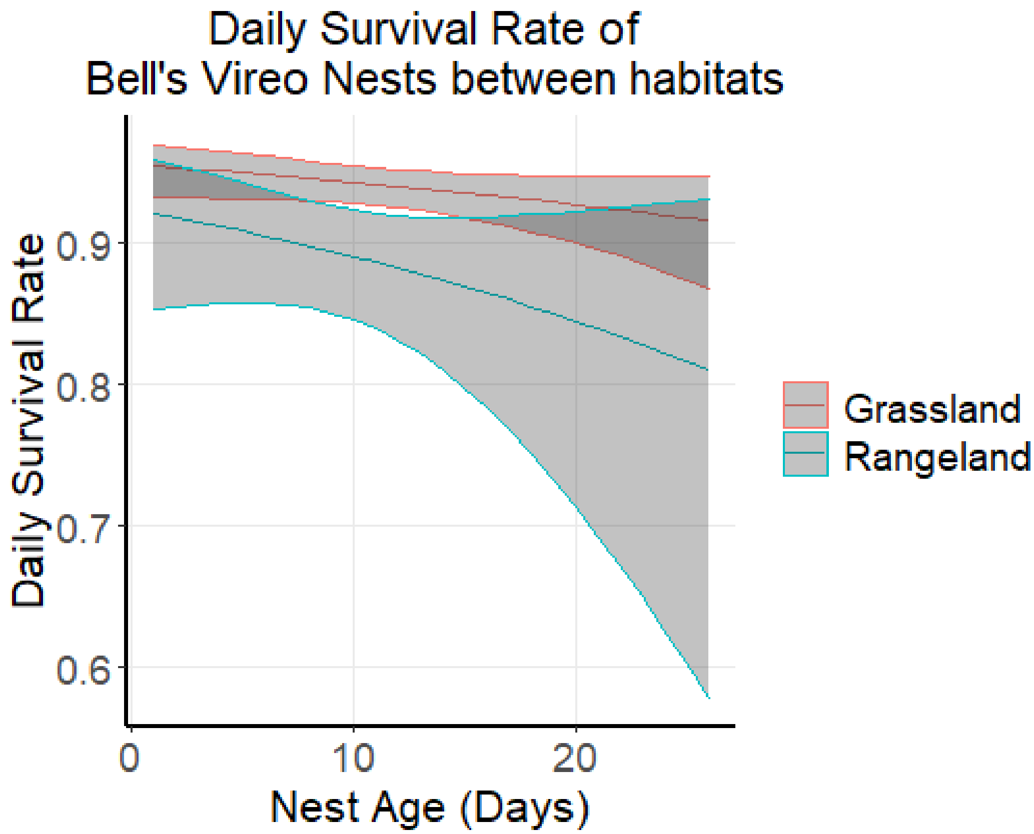

Figure 2.3. Daily survival rate by habitat type for Bell's Vireo nests across a 26-day nesting cycle. More nests were found in grasslands (n=90) versus rangelands (n=32). Error bars represent 95% confidence intervals for daily survival rate.

Table 2.3. Mean vegetation characteristics of Bell's Vireo nests (± standard deviation). No Bell's Vireo nests were found in forested habitats. Shrub coverage and invasive vegetation represent plants within the 11.3 m plot surrounding the nest. Brown-headed Cowbird parasitism represents the percentage of all nests with at least one Brown-headed Cowbird egg.

Habitat	n	Shrub Coverage (%)	Vertical Density (%)	Invasive Shrub Coverage (%)	Invasive Vegetation (% of plot)	Nest Height (m)	Brown-headed Cowbird Parasitism (%)
All Nests	122	23.16 ± 16.68	62.99 ± 21.22	8.19 ± 5.52	13.70 ± 21.30	1.02 ± .45	50.00
Grassland	90	22.56 ± 23.91	63.47 ± 8.43	10.00 ± 2.06	11.17 ± 16.94	1.05 ± .52	45.56
Rangeland	32	24.88 ± 2.08	61.63 ± 8.37	3.13 ± 10.55	20.84 ± 6.56	0.93 ± .69	62.50

Table 2.4. Model variables for Bell's vireo nest daily survival rate, including number of model parameters (npar) and model weights (w*i*).

Model	npar	AICc	ΔAICc	w*i*
(NestAge +Habitat)	3	495.57	0	0.57
(Habitat)	2	497.14	1.56	0.26
(NestAge)	2	501.56	5.99	0.03
(NestAge + PpnShrub)	3	501.65	6.07	0.03
(Null)	1	501.87	6.29	0.02
(PpnShrub)	2	502.30	6.72	0.02
(NestAge + PpnInv)	3	503.30	7.73	0.01
(NestAge + NuddsAVG)	3	503.33	7.76	0.01
(Year)	2	503.61	8.03	0.01
(PpnInv)	2	503.65	8.08	0.01
(NuddsAVG)	2	503.71	8.13	0.01
(Time)	2	503.87	8.29	0.01

Of the total 29 Northern Cardinal nests, 19 were depredated, three failed due to parasitism, and one failed due to an unknown reason. Only six were successful in fledging at least one young (Table 2.2). Northern Cardinals had an overall DSR of 91% (Figure 2.4). Northern Cardinals nested in all three habitat types, with the most nests observed in grasslands (Table 2.5). Of these nests, 28% were placed in invasive substrates, with 20% in Bush Honeysuckle, 4% in Autumn Olive, and 4% in Black Locust. Nest age was the best predictor of DSR. Northern Cardinals had higher DSR during their incubation period (96%) (i.e., the first 12 days of the nesting period) than Bell’s Vireos (94%).

On average, Northern Cardinal nests had 37% shrub coverage on their nest plots, 75% vertical vegetation density, and 28% were placed in an invasive substrate, 13% of the plots were covered by invasive plant species, and 44% of nests were parasitized by at least one Brown-headed Cowbird egg (Table 2.2). Nests on grasslands had a greater likelihood of being placed in an invasive substrate (53%), compared to rangelands (33%) and forests (18%); however, the proportion of invasive substrates in which Northern Cardinals nested did not vary between habitats (Table 2.6).

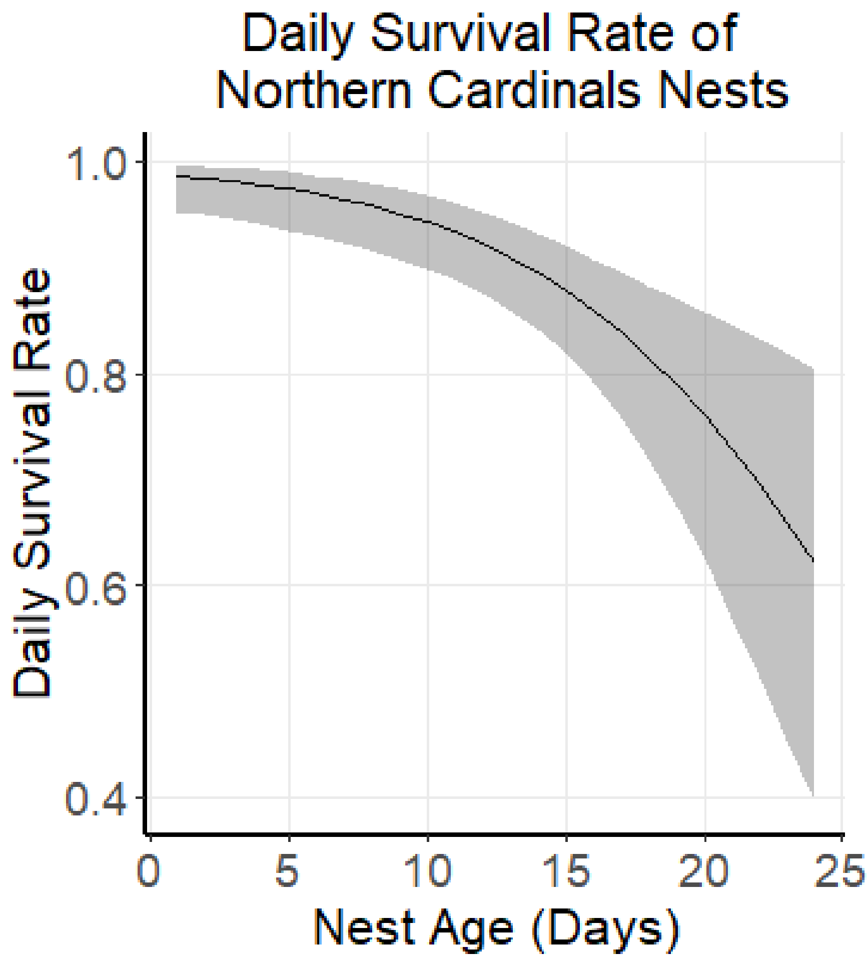

Figure 2.4. Daily survival rate of Northern Cardinal nests in southeast Kansas across a 24-day nesting cycle. Shaded regions indicate 95% confidence intervals of daily survival rate.

Table 2.5. Model variables for the daily survival rates of Northern Cardinal nests, including number of model parameters (npar) and model weights (w*i*).

Model	npar	AICc	ΔAICc	w*i*
(NestAge)	2	94.79	0.00	0.41
(NestAge + PpnInv)	3	95.98	1.18	0.22
(NestAge + PpnShrub)	3	96.76	1.96	0.15
(NestAge + NuddsAVG)	3	96.84	2.04	0.14
(NestAge + Habitat)	4	98.86	4.06	0.05
(Null)	1	108.97	14.17	0.01
(Time)	2	109.88	15.09	0.01
(PpnInv)	2	110.83	16.03	0.01
(NuddsAVG)	2	110.89	16.09	0.01
(PpnShrub)	2	110.94	16.15	0.01
(Year)	2	111.00	16.21	0.01
(Habitat)	3	113.05	18.25	0.01

Table 2.6. Mean vegetation characteristics of Northern Cardinal nests (± standard deviation). Shrub coverage and invasive vegetation represented plants within the 11.3 m plot surrounding the nest. Brown-headed Cowbird parasitism represents the percentage of all nests with at least one Brown-headed Cowbird egg.

Habitat	n	Shrub Coverage (%)	Vertical Density (%)	Invasive Shrub Coverage (%)	Invasive Vegetation (% of plot)	Nest Height (m)	Brown-headed Cowbird Parasitism (%)
All Nests	29	36.65 ± 21.59	74.81 ± 13.81	37.93 ± 9.55	12.82 ± 13.97	1.32 ± 0.49	44.82
Grassland	15	35.86 ± 23.92	79.76 ± 8.43	53.33 ± 10.06	16.06 ± 16.94	1.23 ± 0.52	46.66
Rangeland	3	39.33 ± 2.08	69.08 ± 8.37	33.33 ± 1.15	6.00 ± 6.56	1.20 ± 0.69	66.66
Forest	11	37.00 ± 22.37	69.60 ± 18.56	18.18 ± 9.35	10.27 ± 9.95	1.48 ± 0.40	36.36

DISCUSSION

We found that habitat type plus nest age and nest age were the leading variables for predicting the DSR of Bell's Vireos and Northern Cardinals, respectively, in a disturbed, post-mined landscape. As woody encroachment continues throughout the world, invasive shrub species and vegetative succession will alter wildlife habitat, especially for shrub-nesting birds. Understanding the continuing larger role of these issues is imperative for wildlife managers presently and in the future.

The presence of livestock and grazed pastures may be an important feature affecting Bell's Vireo nest success. Previous literature has mixed findings regarding livestock presence and stocking rates on daily survival rates of bird nests. The majority of literature indicates that livestock are responsible for less than 2% of all nest failures and do not significantly influence daily nests survival rates of bird nests (Bleho et al., 2014, Johnson et al., 2012) However, some studies have shown drastic decreases in daily survival rates in some species of grassland birds in pastures with high stocking rates (Fromberger et al., 2020). All highly publicized research has been completed on obligate grassland bird species, so improved knowledge base is necessary to understand the impacts of livestock stocking rates on shrubland nesting birds. We observed that Bell's Vireo nests placed in rangelands had on average 5% less daily survival than nests placed in grasslands. Nests in grasslands were less likely to be parasitized by Brown-headed Cowbirds and had fewer incidental failures from livestock. Cattle could either snap off the branches on which the nests were located or knock down the entire nest substrate. Increased invasive plant cover is often associated with livestock presence (Hobbs, 2001). Livestock can act as transmission vectors for plant seeds as they are moved from pasture to pasture, by grazing and path creation (Chuong et al., 2016). Assemblages of livestock can also influence the abundance and distribution of Brown-headed Cowbirds, potentially influencing the likelihood of nest parasitism (Goguen & Mathews, 1999).

The leading causes of nest failure reported in the literature for Bell's Vireo are nest depredation and parasitism by Brown-headed Cowbirds (Budnik et al., 2000; Parker, 1999; Rivers et al., 2010). Bell's Vireos are one of the preferred host species for Brown-headed Cowbirds and rates of parasitism vary from 29% to 70.5% (Budnik et al., 2000; Rivers et al., 2010). Nest parasitism can influence host productivity in numerous ways. Parasite eggs or nestling presence can reduce incubation of host eggs and divert food resources away from host young. Adult cowbirds also regularly remove host eggs or destroy established nests to induce renesting (Kosciuch & Sandercock, 2008). We have little evidence to suggest why parasitism rates were higher on rangeland than on grasslands in our study. Brown-headed Cowbirds are associated with livestock, but they are known to feed, breed, and roost in distinct habitats that are separated by up to 10 km (Chace et al., 2005). Thus, local abundances of cowbirds on rangeland may not always indicate higher rates of local parasitism. Research on Plumbeous Vireos (*Vireo plumbeus*) indicated that brood parasitism rates were associated with distance to pasture, with parasitism rates greater than 80% in actively grazed pastures to 33% in areas 8–12 km away from a pasture (Goguen & Mathews, 2000). In the landscape matrix of southeast Kansas, all of our sites were within close proximity (<10 km) to pasture, forest and grassland habitat.

Bell's Vireos were by far the most sampled nesting species we monitored. With an overall DSR of 93%, Bell's Vireo nests were slightly more productive on grassland units compared to rangeland units. The habitat variables that differed the most between rangelands and grassland were invasive species coverage and rate of nest parasitism. Rangelands had more invasive cover with more parasitized nests. In central Missouri, Bell’s Vireo had a DSR of 95.6%, which was too low to compensate for annual mortality, resulting in a shrinking local population (Budnik et al., 2000). Thus, while nationwide populations of Bell’s Vireo are rebounding from a record low in the 1980s (Ziolkowski et al., 2022), local populations in our sampled region and Missouri may decline in the near future due to low nest daily survival rates.

Nest age was the most informative variable for predicting Northern Cardinal nest daily survival rates. Northern Cardinal nests were placed in invasive substrates at a much higher rate than Bell’s Vireo. Nest placement might be attributed to the habitat preferences, or lack thereof, for Northern Cardinals. Northern Cardinals were more likely to place nests in habitats with more tree and shrub coverage. Many nests were found on formerly mined lands that have transitioned to mature forests. Northern Cardinal nests were surrounded by high vertical vegetation densities, particularly on grassland habitats. Rates of brood parasitism were less than those of Bell’s Vireos, but still high at 45%. Northern Cardinal DSR dropped consistently over time with major decreases occurring after the completion of incubation. This pattern may be due to increased activity around the nest during the nestling stage, which may alert predators to the nest location.

Our results indicated a lack of a relationship between the proportion of invasive shrub species and daily survival rates of nests. Invasive shrubs could influence the nest success of in a variety of ways, both positively and negatively. The ecological trap hypothesis, which is one of the most common explanations why invasive shrubs negatively affect bird species, may not apply to some of our study sites. For instance, most studies that support this hypothesis describe increased depredation rates in patches of invasive shrubs (Gleditsch & Carlo, 2014). The invasive vegetation found throughout mined lands differ, as bush honeysuckle created such large continuous monocultures within our study sites. Thus, the ecological trap hypothesis may not apply. It is difficult to determine underlying reasons for decreasing DSR rates without further study in this system.

CONCLUSION

While Bell's Vireo populations are increasing nationwide, they remain a species of conservation concern in Kansas (Rohweder, 2022). Our management recommendations to increase Bell's Vireo nest success are to remove livestock or reduce stocking densities from sites that have high densities of birds during peak breeding season. This change could reduce direct livestock destruction of nest substrates and potentially reduce Brown-headed Cowbird abundance. We also recommend continuing efforts to mitigate woody encroachment, as Bell's Vireo prefer early successional shrublands to early forest succession. While Northern Cardinals are a common songbird species, maintaining nesting habitat may support their persistence in this region. Removing invasive shrub species on all habitat types may reduce ecological traps for shrub dependent birds and improve overall breeding bird diversity in a disturbed landscape.

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APPENDIX

Appendix I. Bird species and individuals observed at point count sampling locations across three sampling years. Values indicate species counts. Focal species are bolded.

Species		2020	2021	2022
Acadian Flycatcher	*Empidonax virescens*	0	1	10
Alder Flycatcher	*Empidonax alnorum*	0	0	7
American Crow	*Corvus brachyrhynchos*	63	105	114
American Goldfinch	*Spinus tristis*	48	54	35
American Kestrel	*Falco sparverius*	1	0	0
Baltimore Oriole	*Icterus galbula*	5	7	10
Barred Owl	*Strix varia*	5	0	3
Barn Swallow	*Hirundo rustica*	6	8	13
Black-and-white Warbler	*Mniotilta varia*	0	2	0
Bell's Vireo	***Vireo belli***	**57**	**116**	**103**
Belted Kingfisher	*Megaceryle alcyon*	0	0	1
Blue-gray Gnatcatcher	*Polioptila caerulea*	29	54	49
Brown-headed Cowbird	*Molothrus ater*	82	102	143
Blue Grosbeak	*Passerina caerulea*	4	0	2
Blue Jay	*Cyanocitta cristata*	27	62	41
Brown Thrasher	*Toxostoma rufum*	7	4	16
Carolina Chickadee	*Poecile carolinesis*	41	54	74
Carolina Wren	*Thryothorus ludovicianus*	95	42	45
Canada Goose	*Branta canadensis*	6	50	8
Cedar Waxwing	*Bombycilla cedrorum*	0	0	6
Chipping Sparrow	*Spizella passerina*	0	2	3
Common Gallinule	*Gallinula galeata*	2	0	0
Common Grackle	*Quiscalus quiscula*	5	2	1
Common Nighthawk	*Chordeiles minor*	2	3	2
Common Yellowthroat	*Geothlypis trichas*	24	48	56
Chuck-will's-widow	*Antrostomus carolinensis*	0	2	0
Dickcissel	*Spiza americana*	284	496	523
Downy Woodpecker	*Picoides pubescens*	12	15	25
Eastern Bluebird	*Sialia sialis*	1	1	5
Eastern Kingbird	*Tyrannus tyrannus*	6	4	6
Eastern Meadowlark	*Sturnella magna*	21	31	32
Eastern Phoebe	*Sayornis phoebe*	1	1	2
Eastern Towhee	*Pipilo erythrophthalmus*	35	31	35
Eastern Wood Peewee	*Contopus virens*	52	64	51
Eurasian Collared Dove	*Streptopelia decaoto*	0	0	1
European Starling	*Sturnus vulgaris*	1	0	1
Fish Crow	*Corvus ossifragus*	8	23	26
Field Sparrow	*Spizella pusilla*	146	137	106
Great Blue Heron	*Andrea herodias*	12	3	3
Great-crested Flycatcher	*Myiarchus crinitus*	30	57	46
Gray Catbird	*Dumetella carolinensis*	5	21	10
Great Egret	*Andrea alba*	2	1	4
Hairy Woodpecker	*Leuconotopicus villosus*	0	1	4
Henslow's Sparrow	*Ammodramus henslowii*	0	7	15

House Sparrow	*Passer domesticus*	1	0	0
Indigo Bunting	***Passerina cyanea***	**159**	**156**	**181**
Kentucky Warbler	*Geothlypis formosa*	2	1	7
Killdeer	*Charadrius vociferus*	15	12	6
Lark Sparrow	*Chondestes grammacus*	1	0	1
Least Flycatcher	*Empidonax minimus*	0	0	4
Louisiana Waterthrush	*Parkesia motacilla*	0	0	1
Mourning Dove	*Zenaida macroura*	80	74	65
Mourning Warbler	*Geothlypis philadelphia*	0	0	1
Northern Bobwhite	*Colinus virginianus*	52	80	41
Northern Cardinal	***Cardinalis cardinalis***	**215**	**281**	**261**
Norther Flicker	*Colaptes auratus*	5	8	1
Northern Mockingbird	*Mimus polyglottos*	21	7	13
Northern Parula	*Setophaga americana*	20	33	28
Orchard Oriole	*Icterus spurius*	12	11	12
Painted Bunting	*Passerina ciris*	0	0	2
Pileated Woodpecker	*Dryocopus pileatus*	9	25	33
Prothonotary Warbler	*Protonotaria citrea*	11	12	11
Purple Martin	*Progne subis*	0	1093	5
Red-bellied Woodpecker	*Melanerpes carolinus*	68	100	95
Red-eyed Vireo	*Vireo olivaceus*	30	32	34
Red-headed Woodpecker	*Melanerpes erythrocephalus*	14	5	6
Red-shouldered Hawk	*Buteo lineatus*	5	5	8
Red-tailed Hawk	*Buteo jamaicensis*	11	5	5
Ruby-throated Hummingbird	*Archilochus colubris*	4	7	6
Red-winged Blackbird	*Agelaius phoeniceus*	47	87	70
Song Sparrow	*Melospiza melodia*	2	0	0
Scissor-tailed Flycatcher	*Tyrannus forficaatus*	3	11	3
Summer Tanager	*Piranga rubra*	13	16	17
Tree Swallow	*Tachycineta bicolor*	3	4	1
Tufted Titmouse	*Baeolophus bicolor*	73	121	100
Turkey Vulture	*Cathartes aura*	13	7	4
Warbling Vireo	*Vireo gilvus*	12	10	13
White-breasted Nuthatch	*Sitta carolinesis*	3	1	11
White-eyed Vireo	*Vireo griseus*	5	20	23
Willow Flycatcher	*Empidonax traillii*	7	3	1
Wild Turkey	*Meleagris gallopavo*	2	0	1
Wood Duck	*Aix sponsa*	0	1	1
Wood Thrush	*Hylocichla mustelina*	4	1	3
Yellow-breasted Chat	*Icteria virens*	66	115	120
Yellow-billed Cuckoo	*Coccyzus americanus*	107	116	77
Yellow Warbler	*Setophaga petechia*	1	0	3
Yellow-throated Vireo	*Vireo flavifrons*	1	3	0

Appendix II. List of detected Kansas bird species of conservation concern (Rohweder 2015). Each species is listed by the number of individuals observed during point count sampling across all sampling years.

Species		2020	2021	2022	Total
Baltimore Oriole	*Icterus galbula*	5	7	10	22
Bell's Vireo	*Vireo belli*	57	116	103	276
Chuck-will's-widow	*Antrostomus carolinensis*	0	2	0	2
Dickcissel	*Spiza americana*	284	478	524	1286
Eastern Kingbird	*Tyrannus tyrannus*	6	4	6	16
Eastern Meadowlark	*Sturnella magna*	21	31	32	84
Eastern Wood-Pewee	*Contopus virens*	52	64	51	167
Kentucky Warbler	*Geothlypis formosa*	2	1	7	10
Lark Sparrow	*Chondestes grammacus*	1	0	1	2
Northern Bobwhite	*Colinus virginianus*	52	80	41	173
Prothonotary Warbler	*Protonotaria citrea*	11	12	11	34
Red-headed Woodpecker	*Melanerpes erythrocephalus*	14	5	6	25
Scissor-tailed Flycatcher	*Tyrannus forficaatus*	3	10	3	16