

Articles

Habitat Patch Use by Fishers in the Deciduous Forest-Dominated Landscape of the Central Appalachian Mountains, USA

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Abstract

Fishers (*Pekania pennanti*) are often associated with the coniferous and mixed forests of the northern United States and central Canada, and their ecology has been studied extensively in portions of their distributional range. Recently, natural range expansion and reintroductions have led to recolonization by fishers to portions of the central Appalachian Mountains, where deciduous forest is the dominant vegetation type. We used noninvasive hair-snare surveys and microsatellite genetic analysis to detect fishers in the central Appalachian Mountains of Pennsylvania. We used these detections within an occupancy modeling framework to explore habitat patch use by fishers and the forest characteristics and land use features that influenced it. We found that the likelihood of patch use by fishers was related to forests with higher proportions of low-density residential areas. Our results also suggested that lower road densities might be related to higher likelihood of fisher patch use. Fishers in Pennsylvania tolerated some forms of land development. Patch use was not driven by forest type or canopy cover, at least within our deciduous forest-dominated study areas. Future research identifying threshold values at which forest cover and land development affect patch use by fishers in the central Appalachian Mountains will better inform management decisions with respect to sites for future reintroduction of fishers.

Keywords: detection probability; eastern deciduous forest; *Pekania pennanti*; occupancy model; Pennsylvania

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Introduction

Regional variation in resource availability, human activity, and climate can affect animal conservation and management plans because local resource requirements may vary spatially, especially for widely distributed species (Davis et al. 2007; Whittingham et al. 2007; Rhodes et al. 2008; Zielinski et al. 2010). The fisher (*Pekania pennanti*; Figure 1) is a widely distributed forest-dwelling carnivore that once occurred across most forested regions of Canada south of the subarctic, and in the United States its distribution extended southward along the Cascade and Coast ranges, Sierra Nevada, northern Rocky Mountains, Appalachian Mountains, and extensively throughout the midwestern region (Hagmeier 1956; Gibilisco 1994; Lewis et al. 2012). By the 1930s and 1940s, however, fishers were extirpated from large portions of their historic range because of habitat loss and overharvesting (Rand 1944; Powell 1993; Gibilisco 1994). Reintroduction efforts and natural recolonization have facilitated the reclamation by fishers of some of their former distribution. In eastern North America, reintroductions and natural recolonization of fishers appear to have been successful (Lewis et al. 2012), including reintroduction efforts in the central and southern Appalachian region of the United States, ranging from southern New York to eastern Tennessee (Lewis et al. 2012). Nonetheless, the current range of fishers in the southern and central Appalachian Mountains is noncontiguous, with large areas of deciduous forest yet to be recolonized (Lewis et al. 2012).

A prominent feature of previous research has been the examination of regional patterns in habitat associations across the distribution of fishers (e.g., Carroll et al. 1999; Zielinski et al. 1999; Davis et al. 2007; Weir and Corbould 2010). For example, fishers in the Pacific states and British Columbia, a distinct population segment listed as threatened pursuant to the U.S. Endangered Species Act (ESA 1973, as amended), are a habitat specialist, occupying low- to mid-elevation mixed forests that are characterized by structural complexity, the presence of

large snags or decadent live trees, and dense canopy cover (Allen 1983; Zielinski et al. 2004, 2010; Purcell et al. 2009; Raley et al. 2012; Aubry et al. 2013). In the eastern United States, information on fisher habitat associations largely comes from the Northeast and Great Lakes regions, where forests are predominantly mixed. There, fishers show seasonal preferences for several forest types, with dense conifer and mixed forest playing a primary role in structuring habitat use (Arthur et al. 1989; Powell 1994; Carr et al. 2007; Fuller et al. 2016). Farther south, in the Appalachian Mountain region, forests are predominantly deciduous, and conifer species are largely restricted to moist, rocky ravines, pine plantations, or barrens. Little is known about fisher-habitat relationships in the deciduous forests of the central Appalachian Mountains. There is some evidence that dense canopy in mixed or coniferous forest is less important for fishers in this region, and that fishers depend on large, deciduous trees for rest sites (Gess et al. 2013). Fishers have recolonized only portions of the region in the past few decades and their conservation status in Pennsylvania is considered vulnerable (Serfass and Dzialak 2010). As such, conservation and management plans for fishers and their habitats in this region would benefit from a more comprehensive understanding of how forest characteristics and land use patterns influence patch use by fishers.

Our goal was to examine the influence of landscape characteristics on patch use by fishers in the predominantly deciduous forests of the Appalachian Mountains in Pennsylvania. Using noninvasive hair snares and landscape-scale environmental covariates, we estimated the probability of patch use by fishers with a likelihood-based occupancy modeling framework that accounted for imperfect detection of fishers (MacKenzie et al. 2006). Fishers are thought to be forest obligates (Seton 1929; Allen 1983; Powell 1993); thus, we predicted that fisher presence in the deciduous forests of Pennsylvania would be associated with patches that had a relatively high proportion of forest and high canopy cover. Fishers used coniferous forest for hunting and resting in Maine



Figure 1. Fisher (*Pekania pennanti*) in the primarily deciduous forests of Pennsylvania. From left to right: deciduous forest of the Blue Knob study area (J.L. Larkin, September 2006); adult male fisher in tree cavity (S.W. Gess, July 2007); adult female fisher in tree (J.L. Larkin, September 2006).

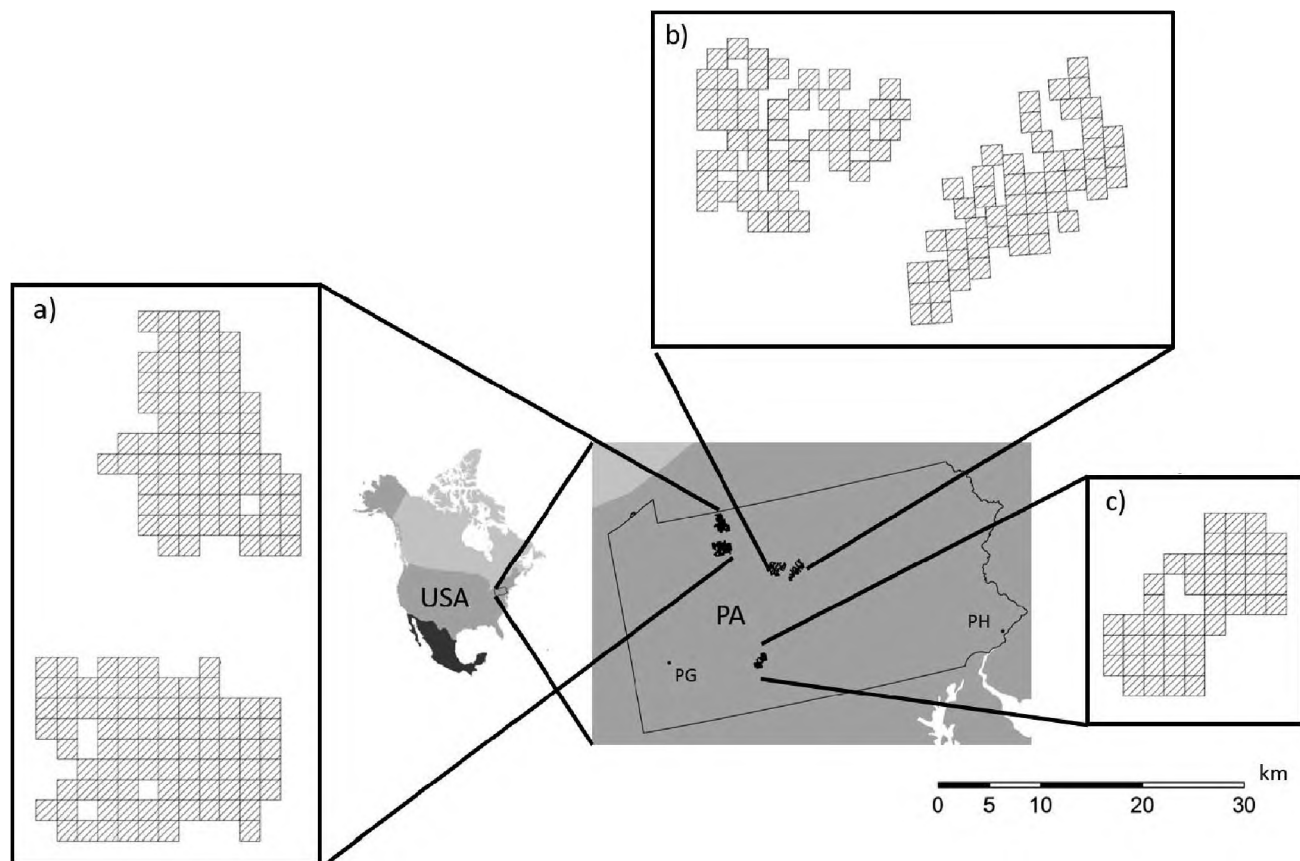


Figure 2. Survey grids used to survey fisher (*Pekania pennanti*) patch use in Pennsylvania from 2007 to 2008: a) Allegheny, b) Quehanna, and c) Blue Knob. PG and PH are the cities of Pittsburgh and Philadelphia, respectively, in Pennsylvania. The inset map shows the position of Pennsylvania within North America.

(Arthur et al. 1989) and Wisconsin (McCann et al. 2014), and thus we expected that patch use by fishers would be positively associated with the proportion of the canopy with a mix of coniferous and deciduous trees. We predicted that forest management, as indexed by land ownership, would influence patch use by fishers. Carroll et al. (1999) and Zielinski et al. (2010) found that publicly-owned forests in northern California managed for multiple purposes and thus consisting of mixed conifer and hardwood had higher value for fishers than private lands that tended to be managed for revenue from timber harvest. As such, we expected fisher patch use in Pennsylvania to be positively associated with publicly-owned land that was managed with and without timber harvest, and negatively associated with private and developed land. Little is known about the influence of roads on fisher habitat use. Road density has been shown to impede gene flow within a fisher population (Garroway et al. 2011), suggesting that roads either directly influence habitat suitability for fishers by posing a barrier to movement or indirectly through increased disturbance near roads. We predicted that patch use by fishers would be negatively associated with road density. Finally, we expected that edge habitat (between forest and open land cover) would be negatively associated with patch use by fishers, similar to the behavior of the

closely related American marten (*Martes americana*; Hargis et al. 1999, Potvin et al. 2000).

Methods

Study areas

We established survey grids at three study areas in three distinct regions of Pennsylvania: northern (Allegheny), central (Quehanna), and southern (Blue Knob; Figure 2). The topography was generally uneven with steep forested slopes, ridges, and valleys ranging in elevation from 240 to 725 m. Mean annual rainfall across the three study areas in 2007 and 2008 was 109 cm (SD = 5 cm), mean annual snowfall was 154 cm (SD = 63 cm), and mean temperature was -1.7°C (SD = 1.2°C) in January and 20.7°C (SD = 0.7°C) in July (PSC 2009). The landscape was largely forested (83–97% across study areas; Table 1), comprised mainly of deciduous forest (66–88% of the forest across study areas; Table 1). Dominant tree species included black cherry (*Prunus serotina*), maple (*Acer* spp.), American beech (*Fagus grandifolia*), and oak (*Quercus* spp.), with eastern hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*) interspersed throughout riparian areas and rocky ravines. The study areas encompassed historic and

Table 1. Mean (SD) value of covariates measured within all survey cells (4-km²) of patch use by fishers (*Pekania pennanti*) across three study regions in Pennsylvania, 2007–2008. For comparison, we included parameters that were omitted from the patch-use models because of strong multicollinearity. We present correlations among covariates and variance inflation factors in Table A1.

Covariate ^a	Allegheny (n = 158)	Blue Knob (n = 44)	Quehanna (n = 100)	All areas (n = 302)
Proportion of forest	0.96 (0.03)	0.83 (0.06)	0.94 (0.03)	0.94 (0.06)
Proportion of deciduous forest	0.73 (0.07)	0.66 (0.09)	0.71 (0.10)	0.71 (0.09)
Proportion of mixed forest	0.23 (0.06)	0.17 (0.08)	0.23 (0.10)	0.22 (0.08)
Proportion of sparse canopy	0.12 (0.06)	0.11 (0.07)	0.09 (0.06)	0.11 (0.06)
Proportion of medium canopy	0.11 (0.03)	0.14 (0.03)	0.23 (0.07)	0.16 (0.07)
Proportion of high canopy	0.77 (0.08)	0.74 (0.09)	0.68 (0.11)	0.74 (0.10)
Proportion of privately-owned land	0.21 (0.18)	0.36 (0.13)	0.07 (0.09)	0.19 (0.18)
Proportion of publicly-owned land with timber harvest	0.73 (0.19)	0.47 (0.17)	0.56 (0.31)	0.64 (0.26)
Proportion of publicly-owned land without timber harvest	0.02 (0.07)	0 (0)	0.32 (0.32)	0.12 (0.24)
Proportion of open land	0.02 (0.01)	0.14 (0.05)	0.05 (0.03)	0.05 (0.05)
Density of edge habitat	1.03 (0.44)	4.23 (0.81)	2.73 (0.92)	2.06 (1.37)
Proportion of developed land	0.01 (0.01)	0.02 (0.01)	0 (0)	0.01 (0.01)
Density of local roads	1.03 (0.43)	0.48 (0.22)	0.59 (0.25)	0.81 (0.42)
Density of major roads	0.20 (0.14)	0.25 (0.13)	0.13 (0.10)	0.19 (0.13)

^a Covariate descriptions can be found in Table 2.

ongoing agriculture, development of energy resources (natural gas wells and coal surface mines [active and reclaimed]), and human settlements. Although most of the forest characteristics were similar among study areas, Blue Knob was slightly less forested and more open than Allegheny or Quehanna (Table 1). One principal difference among study areas was the status of forest ownership: Quehanna had less privately-owned land and more publicly-owned forest without timber harvest than the other study areas (Table 1). Allegheny had the largest amount of forest that was publicly owned but with timber harvest, and Blue Knob had the largest amount of privately-owned land (Table 1). Blue Knob had a higher density of edge habitat than the other study areas, likely because much of the land was privately owned with a variety of land uses (Table 1). Allegheny had the highest density of local roads among the study areas, likely because of the large forestry road network within the publicly-owned forests (Table 1).

Survey and data collection

To estimate patch use by fishers, we collected hair samples from noninvasive hair snares and used genetic sequences from the control region of the mitochondrial deoxyribonucleic acid (DNA) to confirm detection of a fisher. We conducted hair-snare surveys from July 2007 to October 2008 during three survey bouts across the three study areas. Survey bouts were either 16 or 48 d long and the average number of days between survey bouts was 40 d (SD = 46), with a range from 0 to 120 d. We used a survey grid with 4-km² cells, which approximated the average size of a female fisher home range as estimated from telemetry data in the Blue Knob study area (4.31 km², SE = 0.61 km², $n = 9$; E.H. Ellington, Ohio State University, unpublished data). Using the home range as the scale of analysis minimized the likelihood of not detecting individuals with home ranges located between survey sites (Frery et al. 2011; Matthews

et al. 2011). Within each survey cell (4-km²), we placed two hair snares 200–500 m apart in a forested area. In total, we surveyed 302 cells (158 in Allegheny, 44 in Blue Knob, and 100 in Quehanna; Figure 2). In all three study areas, some cells within a grid were not sampled because they were inaccessible. After each survey bout, we moved the hair snares a short distance (approximately 200 m) within the cell, thus surveying each cell at multiple locations.

We used a modified version of the hair-snare design used by Zielinski et al. (2006): we used black, corrugated drainage pipe (24-cm diameter and 60 cm long) with one end permanently closed by a black plastic plate. We inserted three 0.30-caliber copper wire gun brushes into each pipe, approximately 20 cm from the open end of the pipe. We placed a commercial fisher-trapping olfactory lure (Gusto, Minnesota Trapline Products, Pennock, MN) within the hair snare and on a tree near each hair snare. We used bait (approximately 30 mL of pulverized sardines) in conjunction with the lure during the third survey bout in the Allegheny and Quehanna study areas and the second and third survey bouts in the Blue Knob study area; we did this to improve detection rate. We checked each hair snare for samples every 4 d. When hair was present, we collected all three gun brushes, reapplied the lure and bait if needed, and attached new wire brushes. We removed hair from wire brushes and stored them in paper envelopes containing silica desiccant.

We submitted all hair samples to the U.S. Forest Service Rocky Mountain Research Station Wildlife Genetics Laboratory in Missoula, Montana for analysis. Following established protocols in Riddle et al. (2003) and Schwartz (2007), 301 base pairs of the control region of the mitochondrial DNA were sequenced to determine if hair samples were from fisher. There was a 13 to 19% divergence in base pairs between fishers and other mustelids in our study area. Given the quality of genetic samples processed, we are confident that our samples

Table 2. Description of the six covariate sets that we used to model patch use by fishers (*Pekania pennanti*) across the central Appalachian Mountains of Pennsylvania, 2007–2008.

Covariate Set	Covariates	Description
Forest	Proportion of forest	Proportion of land cover classified as forest; derived from the Pennsylvania Land Cover data set (PSU 2007).
Forest type	Proportion of mixed forest	Proportion of land cover classified as forest with a canopy of >20% coniferous trees; derived from the Pennsylvania Land Cover data set (PSU 2007).
Canopy cover	Proportion of sparse canopy cover + proportion of medium canopy cover	Proportion of land cover classified as forest with a canopy of <30% (sparse) or between 30 and 70% (medium) cover. These data were derived from the U.S. Geological Survey canopy cover data set (Homer et al. 2004).
Forest management	Proportion of privately-owned land + proportion of publicly-owned land with timber harvest + proportion of publicly-owned land without timber harvest	Proportion of land cover classified as privately-owned forested land, publicly-owned forested land on which timber harvest could occur, or publicly-owned forested land on which timber harvest could not occur. These data were derived from the 2005 Stewardship Land data set (Bishop 1998) and the Pennsylvania Land Cover data set (PSU 2007).
Linear features	Density of edge habitat + density of local roads	Density (km/km ²) of edge habitat between forest and open land cover, derived from the Pennsylvania Land Cover data set (PSU 2007), or density of local roads (km/km ²), derived from the Pennsylvania 2007 Local Roads data et (PDOT 2011a).
Land development	Proportion of developed land + density of local roads + density of major roads	Proportion of land cover classified as developed land (residential, commercial, or industrial development, bare ground, and buildings associated with agriculture), derived from the Pennsylvania Land Cover data set (PSU 2007). Local and major road density (km/km ²) were derived from the Pennsylvania 2007 Local Roads data set (PDOT 2011a) and the Pennsylvania 2007 State Roads data set (PDOT 2011b), respectively.

were correctly identified to the species (Data A1). Genomic DNA was extracted from hair using the QIAGEN Dneasy Tissue Kit (Qiagen, Valenica, CA) with modifications for hair (Mills et al. 2000). Mitochondrial DNA was amplified using the polymerase chain reaction (PCR) using the primers MP-F' and MP-R4 (Drew et al. 2003) for the control region and CanidL1 and H15149 for the cytochrome *b* region. Both regions were amplified in reaction volumes of 50 μ L that contained 50–100 ng of DNA, 1 \times reaction buffer (Applied Biosystems, Waltham, MA), 2.5 mM MgCl₂, 200 μ M each deoxynucleotide triphosphate, 1 μ M each primer, and 1 U Taq polymerase (Applied Biosystems, Waltham, MA). The PCR protocol was 94°C for 5 min; 40 cycles of 94°C for 1 min, 55°C for 1 min, and 72°C for 1.5 min; and 72°C for 5 min. Polymerase chain reaction products were purified using ExoSap-IT (Affymetrix-USB Corporation, Cleveland, OH) according to manufacturer's instructions, sequenced using Big Dye sequencing kits (Applied Biosystems, Waltham, MA), and visualized on an ABI 3700 DNA analyzer (Applied Biosystems, Waltham, MA).

Occupancy modeling

We used likelihood-based single-season occupancy modeling (MacKenzie et al. 2006) to estimate the

probability of detecting the presence of a fisher (*p*) in each survey cell and the probability that a fisher used (Ψ) each survey cell. It was unlikely that our study areas were closed systems during the survey period; thus our estimates are more appropriately described as the probability of patch use by a fisher (MacKenzie et al. 2006). Henceforth, we use the term “patch use” to refer to the probability of a patch being used by a fisher in an occupancy model framework. We considered six covariate sets that we selected a priori, each representing one or several different environmental factors that could influence the probability of patch use by fishers: proportion of forest, forest type, forest management type, forest canopy cover, linear features, and land development (Table 2). We also included nonforest cover-based environmental metrics because these are typically negatively associated with fishers and other forest carnivores (Weir and Corbould 2010). To account for the variations in surveying protocol, we included two additional covariates in every model: length of sample bout (L.BOUT) and the use of bait (BAIT), on the probability of detecting fisher.

We focused our analysis on how landscape-scale forest variability influenced the probability of patch use by fishers because previous research has suggested that

Table 3. Models of fisher (*Pekania pennanti*) detection (p) and patch use (Ψ) from noninvasive hair-snare surveys conducted across the central Appalachian Mountains of Pennsylvania, 2007–2008. Italicized models failed to converge. QAIC_c, quasi-Akaike information criterion corrected for small sample size; w_i , model weight; ER, evidence ratio.

Model ^{a,b,c}	k	QAIC _c	Δ QAIC _c	w_i	ER (top model)	ER (null Ψ)
$p(L.BOUT + BAIT), \Psi(\text{land development})$	7	476.92	0.00	0.46	1:1	2,290:1
$p(L.BOUT + BAIT), \Psi(\text{linear features} + \text{land development})$	8	477.30	0.38	0.38	1:1.2	1,890:1
$p(L.BOUT + BAIT), \Psi(\text{forest type} + \text{canopy cover})$	7	481.50	4.58	0.05	1:9.9	232:1
$p(L.BOUT + BAIT), \Psi(\text{canopy cover})$	6	481.90	4.98	0.04	1:12.0	190:1
$p(L.BOUT + BAIT), \Psi(\text{forest} + \text{canopy cover})$	7	482.36	5.44	0.03	1:15.2	151:1
$p(L.BOUT + BAIT), \Psi(\text{forest} + \text{forest management type})$	8	482.45	5.53	0.03	1:15.9	144:1
$p(L.BOUT + BAIT), \Psi(\text{forest} + \text{forest type})$	6	485.38	8.46	0.01	1:68.7	33:1
$p(L.BOUT + BAIT), \Psi(\text{forest})$	5	486.05	9.13	0.01	1:96.1	24:1
$p(L.BOUT + BAIT), \Psi(\text{forest management type})$	7	486.63	9.71	<0.01	1:128.4	18:1
$p(L.BOUT + BAIT), \Psi(\text{forest type} + \text{forest management type})$	8	486.64	9.72	<0.01	1:129.0	17:1
$p(L.BOUT + BAIT), \Psi(\text{linear features})$	6	489.44	12.52	<0.01	1:523.2	4:1
$p(L.BOUT + BAIT), \Psi(\text{forest type} + \text{linear features})$	7	489.47	12.55	<0.01	1:531.1	4:1
$p(L.BOUT + BAIT), \Psi(\text{forest type})$	5	490.28	13.36	<0.01	1:796.3	3:1
$p(L.BOUT + BAIT), \Psi(NULL)$	4	492.39	15.47	<0.01	1:2,289.6	1:1
$p(NULL), \Psi(NULL)$	2	529.14	52.22	<0.01	1:2.2 $\times 10^{11}$	
$p(L.BOUT + BAIT), \Psi(\text{forest type} + \text{land development})$	8					
$p(L.BOUT + BAIT), \Psi(\text{forest} + \text{land development})$	8					
$p(L.BOUT + BAIT), \Psi(\text{canopy cover} + \text{linear features})$	8					

^a $p(L.BOUT + BAIT)$ represents a probability of detection scenario in which the probability of detection varied by the use of bait (BAIT) and by the length of survey bout (L.BOUT).

^b Covariates within each covariate set are described in Table 2.

^c In the model linear features + land development, we included density of local roads only once.

fisher habitat selection may be influenced by factors at scales larger than the home range (Carroll et al. 1999). Thus, we derived our environmental covariates from a 36-km² sample cell centered over the original 4-km² survey cell. Incidentally, our covariates and other land cover metrics within 4-km² cells were highly correlated with those same covariates and metrics measured within 36-km² cells (Ellington 2010; E.H. Ellington, Ohio State University, unpublished data). We transformed proportional covariates to continuous covariates using the logit transformation. We then standardized (z-score) all covariates, tested for correlations among them using a Pearson pair-wise correlation matrix, and generated a variance inflation factor for each covariate using the car package (Fox and Weisberg 2011) in R (R Core Team 2016; Text A1).

We used our six a priori covariate sets to create 21 models of patch use by fishers on the basis of all combinations of one and two covariate sets. We also included two null models, one that included the probability of detection covariates but not the probability of patch-use covariates, and one that did not have probability of detection or patch-use covariates. To limit overparameterization, we excluded models that had more than eight parameters ($n = 4$), which in practice meant a maximum of four patch-use covariates. All nonnull models included the two fisher detection probability covariates, a probability of detection error term, and a patch-use error term. We also excluded models that would have included correlated covariates ($n = 1$). For example, we did not generate the model $\Psi(\text{forest canopy cover} + \text{forest management type})$ because it would have had too many parameters.

Likewise, we did not generate the model $\Psi(\text{proportion of forest} + \text{linear features})$ because the covariate proportion of forest and density of edge habitat were correlated (Text A1). We conducted our occupancy modeling analysis and model selection procedures using the R packages unmarked (Fiske and Chandler 2011) and AICcmodavg (Mazerolle 2016). We omitted models that failed to converge or when convergence was suspect because of the nonsensical parameters (i.e., extremely large coefficient estimates with extremely small variance, $n = 3$; Table 3; Weller 2008). We ranked the remaining models ($n = 15$) using Akaike's information criterion corrected for small sample size (AIC_c; Burnham and Anderson 2002). We then calculated c-hat from the global model (a model that contained all of the covariates used in the analysis) using 10,000 simulations to determine if the data were overdispersed (Burnham and Anderson 2002). We used this c-hat value (1.04) to calculate quasi-AIC_c following Burnham and Anderson (2002). We also calculated evidence ratios for the top model compared with each subsequent model and for each model compared with the null patch-use model. We did not conduct model averaging because across the model set, the environmental covariates occurred at different frequencies. Our analysis code is available as Code A1.

Results

We collected 649 hair samples (338 in Allegheny, 210 in Blue Knob, and 101 in Quehanna) and of these, 96 were identified as fisher (35 in Allegheny, 43 in Blue Knob, and 18 in Quehanna). The remaining 552 hair

Table 4. Coefficients (β), 95% confidence intervals (CI) for each Ψ covariate, and the Ψ intercept term in the two competing models (quasi-Akaike information criterion corrected for small sample size [QAIC_c] model weight [w_i] > 0.05) of fisher (*Pekania pennanti*) detection (p) and patch use (Ψ) from noninvasive hair-snare surveys conducted across the central Appalachian Mountains of Pennsylvania, 2007–2008. Coefficients with 95% CI that do not overlap zero are in bold font.

Model	Covariate	β	95% CI	
			Lower	Upper
Land development (QAIC _c w_i = 0.46)	Intercept	0.91	–1.10	2.96
	Proportion of developed land	1.85	0.10	3.58
	Density of local roads	–0.82	–2.20	0.55
	Density of major roads	–0.92	–2.20	0.35
Land development + linear features (QAIC _c w_i = 0.38)	Intercept	0.89	–1.40	3.18
	Proportion of developed land	1.78	–0.20	3.73
	Density of local roads	–0.72	–2.30	0.87
	Density of major roads	–0.87	–2.20	0.48
	Density of edge habitat	0.18	–0.70	1.09

samples were identified as species other than fisher ($n = 483$) or the DNA quality was too poor to analyze ($n = 69$). We detected fishers in 72 of 302 (24%) survey cells (Data A2). The use of bait in addition to a lure had a strong, positive relationship with detection probability of fishers (in the top model, $\beta = 1.21$, $SE = 0.28$). Specifically, when we examined bait alone (i.e., null patch use and not considering the length of the survey bout), the probability of detection increased from 0.09 to 0.32 with the use of bait. We were also more likely to detect fishers during the longer survey bout in the Blue Knob study area (in the top model, $\beta = 1.84$, $SE = 0.43$). Specifically, when we examined survey bout length alone (i.e., null patch use and not considering the use of bait), the probability of detection increased from 0.09 to 0.19 with a longer survey bout (48 days).

Only two models had a weight (w_i) > 0.05 (Table 3). The only difference between these two models was the inclusion of density of edge habitat: this covariate had a weak and highly variable effect on patch use by fishers (Table 4), indicating that this parameter was uninformative (sensu Arnold 2010). Thus, we focused our analysis on the coefficients of the covariates from the top model only. In the top model, patch use by fishers was positively correlated with the proportion of developed land ($\beta = 1.85$, $SE = 0.89$). In this model, patch use by fishers also displayed a negative correlative trend with the density of local ($\beta = -0.82$, $SE = 0.70$; Table 4) and major roads ($\beta = -0.92$, $SE = 0.64$; Table 4) with high variance (95% confidence interval [CI] spans 0). Although neither of our top models was unanimously supported and the 95% CI around the effect of several of our covariates spanned 0 (Table 4), both competing models were more likely to explain patch use than the null patch-use model (Table 3).

Discussion

We know little about habitat use by fishers in the predominantly deciduous forests of Pennsylvania, where coniferous forest with dense canopy cover, typically thought to be suitable as habitat for fishers, is uncommon. Within these forests, we found a positive relationship between the proportion of human-dominat-

ed land uses and patch use by fishers. We also found a negative correlative trend between road density and patch use, although this relationship was only weakly supported. We draw two primary conclusions from our findings: 1) we did not detect a relationship between patch use by fishers and the proportion of forest, forest type, or canopy cover; and 2) fishers in our study areas appeared to be tolerant of some forms of land development.

Many studies in eastern North America have found that fishers selected for coniferous forests, forests with a mixture of coniferous and deciduous trees, and forests with dense canopy cover (Allen 1983; Arthur et al. 1989; Powell 1994; Carr et al. 2007; Fuller et al. 2016). Although we did not detect a strong relationship between patch use and the proportion of forest, forest type, or canopy cover, these features were fairly ubiquitous within and among our study areas (Table 1) and thus we may not have sampled conditions in which an effect could have occurred. We do not interpret our results to suggest that these features are not important for fishers in the central Appalachian Mountains; indeed, our study areas had relatively high proportions of forested land with high canopy cover and we detected fishers at all three study areas. Rather, these landscape features might influence fisher patch use in parts of the central Appalachian Mountains where they are limiting. In New York, just north of Pennsylvania, for example, a statewide occupancy model analysis found that proportion of coniferous and mixed forest was positively related to fisher occupancy (Fuller et al. 2016). Future research could expand on our study by assessing patch use in this region where there is more variability in the proportion and type of forest, as well as variability in canopy cover.

Fisher populations appear to be well established in the deciduous-dominated forests of the central Appalachian Mountains: we detected fishers in all three of our study areas and the reintroduction of fishers to the region has been considered successful (Lewis et al. 2012). In fact, a legal fisher trapping season was recently initiated within several wildlife management units throughout Pennsylvania (PGC 2016). Although fishers occurred in this region historically (Powell 1993; Gibilisco 1994), we are only beginning to learn about fisher ecology in these

deciduous forests. Studies from across the distribution of fishers show that their diet is diverse (Arthur et al. 1989; Weir et al. 2005; Aubry and Raley 2006; Raley et al. 2012), and in deciduous-dominated forests, fishers appear to be capitalizing on prey that tend to occur in these forests. The population density of snowshoe hare (*Lepus americanus*), a common prey item for fishers in coniferous forests (Powell 1993), is relatively low in Pennsylvania (Diefenbach et al. 2005) and was not observed in the stomachs of a sample of 90 fishers in Pennsylvania (McNeil et al. 2017). However, Bowman et al. (2006) found that fishers in Ontario, Canada switched to alternate prey during lagomorph population lows. The most common prey items observed in the stomachs of fishers in Pennsylvania were white-tailed deer (*Odocoileus virginianus*) carrion, cottontail (*Sylvilagus* spp.), and members of the Cricetidae and Scuiridae families (McNeil et al. 2017). Most of these species are typically associated with deciduous forests (Althoff et al. 1997; Morin et al. 2005; Gibbes and Barrett 2011). In addition to utilizing prey species found in deciduous forests, fishers also appear to be successfully utilizing features of deciduous forests for rest and den sites: fishers used large, deciduous trees for resting in Pennsylvania (Gess et al. 2013) and female fishers used cavities of large, deciduous trees as natal dens in the mixed deciduous forests of south-central Maine (Paragi et al. 1996) and Ontario (Koen 2005).

We know little about how fishers respond to forest disturbance in the eastern parts of their distribution. Powell (1993) suggested that fishers that occur in the second-growth deciduous forests of eastern North America may be tolerant of small clear-cuts of forest if those cuts are interspersed within large tracts of connected, uncut forests. Fishers appeared to have mixed responses to land development in our study. We found that patch use by fishers was positively related to the proportion of developed land, which in our study areas was most frequently described by low-density residential areas. Future work could examine how particular features of low-density residential areas might lead to increased patch use by fishers, such as increased prey availability or decreased predator occurrence. Fisher populations in other parts of northeastern North America do not appear to be limited by forest fragmentation (Arthur et al. 1989; Koen et al. 2007), and Lancaster et al. (2008) detected fishers in landscapes with as little as 40% forest cover. Human-developed landscapes that retain some forest cover might be adequate as habitat for fishers. For example, LaPoint et al. (2013) found that fishers persisted in the suburban areas of Albany, New York. We found that patch use by fishers might be inversely related to density of both local and major roads, although support for this finding was limited. Nonetheless, negative relationships between fisher patch use and road density have been found by other researchers; for example, Fuller et al. (2016) found a negative relationship between fisher occupancy and road density in New York. The impact of road density and habitat fragmentation on fishers could be the focus of future research, particularly as infrastructure associated

with the central Appalachian Mountains' burgeoning energy development (i.e., wind, oil, and gas) continues to fragment much of the region's deciduous forests (Drohan et al. 2012).

The current distribution of fishers in the southern and central Appalachian Mountains is not contiguous (Lewis et al. 2012) and we did not detect fishers in all of our survey cells. It is possible that fisher patch use is limited in this region by forest features that we were unable to measure at the landscape scale of our study, such as the presence of rest structures and large, deciduous trees with cavities for denning. We found that there was little variation in the proportion of forest, forest type, and canopy cover among and within our study areas, but it is possible that other forest cover characteristics that we were unable to quantify did vary spatially. Specifically, diversity and interspersed of various seral stages of forest succession may be an important predictor of patch use by fisher. Stands of larger size classes dominate much of Pennsylvania's forests (McWilliams et al. 2007), and a lack of early successional forests is known to influence the diversity and abundance of many vertebrate prey species (Litvaitis 1993, Askins 2001). It is also possible that elements of finer-scale forest structure varied among and within our study areas, which could explain some of the variation in patch use by fishers that we observed. The increased availability of LiDAR remote sensing data provides researchers with an opportunity quantify aspects of forest structure at scales ranging from the landscape to microsite (Lefsky et al. 2002). Future research that incorporates this technology could assess whether spatial variation in forest age class interspersed or microhabitat features known to be important to fishers in Pennsylvania (i.e., rest site structures; Gess et al. 2013) limits patch use by fishers. Such studies should provide insight that would help forest managers to retain or create sufficient forest structure for fishers.

We conducted hair-snare surveys using both unbaited and baited designs. We found that the use of bait increased our ability to detect fishers that were present. As such, we recommend that future studies use bait in conjunction with scent lures when estimating fisher patch use. We also note that in California and New York, the probability of detection of fishers varied by season (Fuller et al. 2016; Sweitzer et al. 2016); considering this in future studies might lead to more precise and accurate estimates of how environmental conditions relate to fisher patch use.

Finally, although we found a positive relationship between patch use by fishers and developed land, there is likely some threshold at which the probability of patch use will begin to decrease as the amount of developed land within an area increases (sensu Lancaster et al. 2008). This threshold of landscape heterogeneity represents a balance between the benefits of increased prey availability associated with ecotonal habitat (Moore and Swihart 2005) and forested habitat required for successful reproduction (Paragi et al. 1996) and shelter (Gess et al. 2013), and could be an important focus of future research. Ultimately, a better understanding of the

factors that influence the distribution and abundance of fishers in eastern North America will inform management decisions with respect to sites for future reintroduction of fishers and resource extraction.

Supplemental Material

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Reference S1. Allen AW. 1983. Habitat suitability index models: fisher. U.S. Fish and Wildlife Service FWS/OBS-82/10.45.

Found at DOI: <http://dx.doi.org/10.3996/012016-JFWM-006.S1> (1467 KB PDF).

Reference S2. Aubry KB, Raley CM. 2006. Ecological characteristics of fisher (*Martes pennanti*) in the Southern Oregon Cascade Range. Report on the Rogue River Fisher Study. Olympia, Washington: USDA Forest Service, Pacific Northwest Research Station.

Found at DOI: <http://dx.doi.org/10.3996/012016-JFWM-006.S2> (1021 KB PDF).

Reference S3. Diefenbach DR, Rathbun S, Vreeland JK. 2005. Distribution and coarse-scale habitat association of snowshoe hare in Pennsylvania.

Found at DOI: <http://dx.doi.org/10.3996/012016-JFWM-006.S3> (416 KB PDF).

Reference S4. McWilliams WH, Cassel SP, Alerich CL, Butler BJ, Hoppus ML, Horsley SB, Lister AJ, Lister TW, Morin RS, Perry CH, Westfall JA, Wharton EH, Woodall CW. 2007. Pennsylvania's Forest 2004. Resource Bulletin NRS-20. Newtown Square, Pennsylvania, USA: USDA Forest Service, Northern Research Station. Found at DOI: <http://dx.doi.org/10.3996/012016-JFWM-006.S4> (7 MB PDF).

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Code A1. Analysis code for program R that we used to: 1) generate models of fisher (*Pekania pennanti*) detection and patch use in Pennsylvania from July 2007 to October 2008 within an occupancy-modeling framework; and 2)

compare these models using Akaike's information criterion. Found at <http://dx.doi.org/10.5061/dryad.p397k/1> (15 KB DOCX).

Data A1. DNA sequence file of samples identified as fisher (*Pekania pennanti*) from hair-snare surveys conducted in Pennsylvania from July 2007 to October 2008. Found at <http://dx.doi.org/10.5061/dryad.p397k/5> (31 KB FAS).

Data A2. Capture history and coordinates of survey cells used to examine fisher (*Pekania pennanti*) detection and patch use in Pennsylvania from July 2007 to October 2008 within an occupancy-modeling framework. Found at <http://dx.doi.org/10.5061/dryad.p397k/2> (13 KB DOCX).

Text A1. Description of correlations among covariates in the analysis and other measurements of forest characteristics and land uses within the survey cells. Found at <http://dx.doi.org/10.5061/dryad.p397k/4> (12 KB DOCX).

Table A1. Variance inflation factors and Pearson's correlations among covariates in the analysis patch use by and detection of fishers (*Pekania pennanti*) across the central Appalachian Mountains of Pennsylvania, 2007–2008. Found at <http://dx.doi.org/10.5061/dryad.p397k/3> (13 KB DOCX).

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