Final Report

AM11043, Project 5

Determine Causative Factors of Changes in Northern New York Fisher Populations

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Executive Summary

The Adirondack region remained the last foothold in NY State for fishers in the late 19th century after severe depletion of forests and game elsewhere (Krohn 2012). Following decades of forest regrowth and sustainable harvest regulations, fisher populations have recently recovered and reclaimed much of their historic range. However, a decline of fisher catch-per-unit-effort (CPUE) over the past 10-15 years has been apparent in the Adirondack region, which motivated the current study. Our objectives were to quantify the survival, productivity, and recruitment of fisher within the central Adirondacks and compare those population vital rates to a nearby population in the Tug Hill Plateau to reveal drivers of population change in the region. We expected declines in prey availability, aligned with an east-west gradient of increasing forest stand age, to correlate with lower survival and productivity in Adirondack fisher compared to Tug Hill fisher. We further expected the inter-annual pattern of beech nut masting events to drive changes in food availability (directly through nuts, indirectly through effects on small mammal populations) that would drive interannual patterns in survival and productivity. Lastly, in partnership with neighboring states, we investigated fisher exposure to anticoagulant rodenticides across NY State.

This work was the primary focus of a PhD dissertation at SUNY ESF (Stephanie Cunningham '23), relied heavily on data and assistance from the NY DEC and licensed trappers, and involved intensive monitoring of GPS-collared female fisher in den trees (to determine productivity and kit survival), extensive monitoring of VHF-collard fisher by airplane (to determine survival), and samples collected from harvested fisher statewide (to determine potential productivity and exposure to anticoagulant rodenticides). The study was initiated with our first live capture of fisher in January 2019, with monitoring of radio-collared animals continued through December 2022, and with statistical analysis and write-up completed (and Ph.D. successfully defended) summer 2023. This report summarizes the main research efforts and findings.

Annual survival rates tended to be higher for female versus male fishers, and higher for adults versus juveniles. Surprisingly, survival rates (pooled across sex and age classes) were 1.4x higher in the Adirondack population than the Tug Hill population. Tug Hill fisher sustained 5.2x higher harvest mortality than Adirondack fisher, with harvest mortality increasing during years of mast failure. We estimated parturition (birth) dates to range 14 March – 9 April. Maximum observed litter sizes ranged 1-4 kits (average 2.42), being slightly higher and more variable in Tug Hill versus Adirondack fishers. On average 76% of kits survived from birth to 11 weeks of age (when kits began moving independently from their mother), ranging 15 to 99% among individuals and years, declining with stand age, and being lower overall in for Adirondack versus Tug Hill fisher. A pattern of alternate year breeding coincided with the beech nut mast cycle, with 75-77% of our collared- and breeding-aged female fishers producing kits in years following mast production (2019, 2021) and only 17% producing kits the year following mast failure (2020)—most notably in Adirondack fisher. Overall, we provide evidence for resource limitations in Adirondack fisher, with greater space requirements and lower productivity, rather than lower survival, underlying observed differences in population abundance compared to the Tug Hill Plateau.

Lastly, we detected residues of 1-5 anticoagulant rodenticide compounds in 83% of fisher harvested across NY State, with exposure rates highest in males, middle-aged fisher (peaking ~3.5-4.5 years of age after territory establishment and coincident with prime breeding age), where low-density housing is interspersed within forest-dominated landscapes, and in years following beech nut masting events.

Introduction

Fishers (*Pekania pennanti*) are medium-sized carnivores in the weasel family. Globally, fisher range spans much of the boreal forest, extending coast-to-coast across Canada and southward into the northern United States within the eastern and mid-western states as well as the mountainous regions of western states. Across this range, fisher populations have exhibited both widespread expansion in recent decades and regional declines (Lewis et al. 2012, LaPoint et al. 2015).

Fishers, like many other mammalian carnivores, suffered severe range contractions after the arrival of Europeans in North America due to a combination of deforestation and unregulated trapping (Aubry and Lewis 2003, Lewis et al. 2012). Only within the last 75 years have fishers begun recolonizing their former range (Krohn 2012). In the Northeast, fisher recovery has been facilitated by regrowth of the region's forests over the past century, implementation of sustainable harvests via regulated trapping seasons (Lancaster et al. 2008, Fuller et al. 2016), and use of translocations and reintroductions (Lewis et al. 2012). The rate of fisher population expansion in the northeastern states has been high relative to western states, and their current range in the east, including within New York State, is approaching its pre-colonial extent (LaPoint et al. 2015).

Yet, the contemporary forests occupied by fisher differ considerably from their historic counterparts. Eastern forests have experienced shifts in species dominance due to a series of disease and pest outbreaks and are facing continued shifts driven by climate change (Bose et al. 2017). The current forest is also highly fragmented (Adams et al. 2019), with the Northeast containing extensive wildland-urban interface (developments adjacent to wild vegetation; Radeloff et al. 2005), with an ever-expanding urban footprint (Shifley et al. 2014). Under such conditions wildlife "spill over" into residential and agricultural areas creating human-wildlife conflict while toxic residues (Köhler and Triebskorn 2013, Rattner et al. 2014), domestic animals (Brearley et al. 2013, Hassell et al. 2017), and humans themselves (Grignolio et al. 2011, Coppes et al. 2017) spill over into wildlife habitats posing disturbances and mortality risks. Tolerance for human disturbance varies among wildlife species and the type of disturbance, restricting available habitat for some species more than others (Lesmeister et al. 2015).

Certain species, including fishers, have adapted to (Randa and Yunger 2006, LaPoint et al. 2015) or even thrived in the fragmented eastern forests. Timber harvest and land use conversion has resulted in forests composed predominately of early- to mid-successional stages (Thompson et al. 2013), which may offer greater prey diversity and abundance for fisher compared to more mature forests. Given spatial variation in forest conditions, it is not surprising that even within a given state, where fisher management has been harmonized, fisher may be rapidly recolonizing parts of their former range while exhibiting evidence of population decreases in other areas (New York State Department of Environmental Conservation 2015).

The research reported herein involves multiple sources of data (e.g., camera traps, radio-collared individuals) to primarily monitor the fate and reproductive success of fisher across an area spanning the central Adirondacks (low density population) to the Tug Hill (high density population). Secondarily, we quantified potential productivity and exposure to anticoagulant rodenticides statewide using harvest data. This research benefitted enormously from the contributions of licensed trappers who recorded the necessary effort and harvest data to estimate population trends, provided carcasses from which we could sample fisher health, and assisted in the live-capture and radio-collaring of fisher in our focal

areas. Herein we report on the population vital rates of fisher in NY, and potential drivers of change in those vital rates, to help guide effective management actions.

Whereas each of the following sections provides a summary of the main methods and results, full details are available in Cunningham (2023).

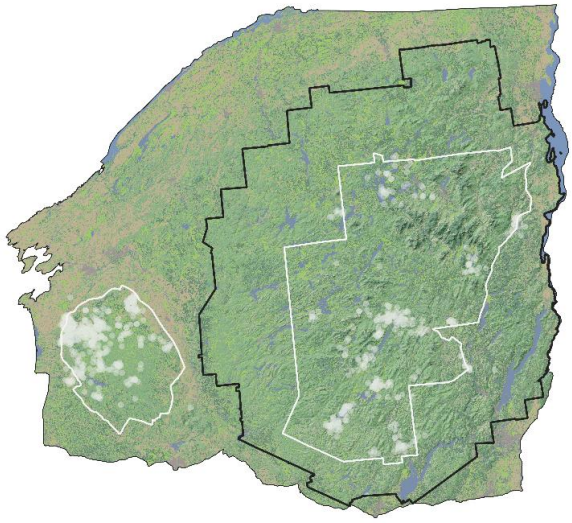

Figure 1. Map of northern New York State indicating the two primary study areas (white outline), the Adirondack Park (black outline), and approximate locations collected from radio-collared fisher tracked in this study (white dots).

Fisher Survival

We investigated fisher survival within two focal areas spanning northern New York State (Figure 1). Our focal areas (central Adirondacks and Tug Hill Plateau) shared many ecological characteristics (e.g., a predominantly northern hardwood forest, harsh winters) but also differed in forest management history, abiotic conditions, human land use intensity, and apparent fisher population density (New York State Department of Environmental Conservation 2015). Catch-per-unit-effort (CPUE) data based on the regulated fall harvest indicated a lower density fisher population in the central Adirondacks region compared with the Tug Hill Plateau (New York State Department of Environmental Conservation 2015, Linden et al. 2017; Figure 2). We hypothesized that the observed patterns in CPUE were due to differences in demographic rates, especially survival. Therefore, our objectives were to 1) determine age-, sex-, and focal area-specific survival rates, 2) identify factors contributing to differences in mortality risk, and 3) examine drivers of harvest mortality risk.

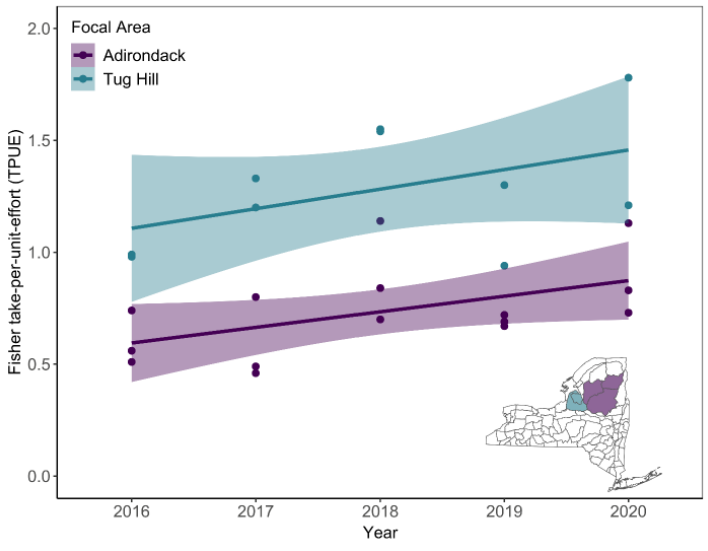

Figure 2. Take-per-unit-effort (TPUE) of fishers in northern New York State, 2016–2020, according to wildlife management unit boundaries most closely associated with Tug Hill Plateau and Adirondack Park focal areas.

State wildlife biologists, assisted by licensed recreational trappers, live-trapped and collared 177 individual fisher for this study (Table 1). Trapping occurred between approximately 25 October and 7 March each winter between January 2019 and March 2022. Tomahawk box traps (model 108) were enclosed by black coroplast, lined with sheep's wool, baited with beaver or moose and a commercial scent lure, and set near roads or trails, generally in pairs. Trapped fisher were chemically immobilized at the capture site when

Table 1. The number of individual fishers captured for this study by sex and age class, and sorted by focal study area. Immature fisher were <2 years old, adults ≥2 years old. The number of individuals radio-collared and meeting the criteria for inclusion in the survival analysis are shown in parentheses.

Sex & Age Class	Tug Hill Plateau	Adirondack Park
Immature male	39 (29)	30 (24)
Immature female	42 (35)	22 (17)
Adult male	9 (7)	18 (18)
Adult female	9 (8)	8 (8)
All fishers	99 (79)	78 (67)
Recaptures[a]	4	7

[a]11 individuals were recaptured and fitted with new collars after 1 year.

possible or transported to an indoor location for processing when necessary. We extracted a tooth (pre-molar 1) to determine fisher age, drew a blood sample, recorded body measurements, affixed a permanent metal ear tag, and fitted animals with a VHF- or GPS-collar prior to release (adult females were fitted with GPS collars for the productivity study described in next section). We successfully determined age for 121 fisher (83%) and used field-based aging (based on tooth eruption and wear) for the remaining 25 fishers. The radio-collars were equipped with mortality signals that activated after eight consecutive hours of no movement.

We tracked the status and location of each collared fisher at least once per month from initial release until their death was detected or we stopped hearing their signal for several months. While we conducted some ground-based telemetry, and collected camera-based sightings of collared fisher, the most efficient way to keep track of fishers was via fixed-wing aircraft. We attached antennas to wing struts to listen for radio signals from collars (Figure 3). We recorded a location at the point where a telemetry signal was first detected, and assumed the collared animal was within 3.2–4.0 km of that location based on test collars. When a mortality signal was detected, efforts were made to physically find the animal as soon as possible, generally within three days (but typically much longer than 3 days since the animal died, which made determination of mortality cause difficult). Intact carcasses were sent to the DEC Pathology Lab in Delmar, NY or to the Wildlife Health Lab at Cornell University in Ithaca, NY for necropsy, pathology, and toxicology screening. Given the often-poor evidence available at carcass sites, we were conservative in our attribution of mortality to specific sources, with the majority classified as “unknown”.

Prior to analyzing survival rates, we excluded individuals if they died within 7 days of capture to eliminate bias due to capture effects (Smith et al. 2022). We had two male fishers die while under anesthesia or shortly after, likely due to rodenticide poisoning as indicated from the necropsy. We further eliminated those that were collared but never relocated after release (N = 16). Half of these (N=8) were lost during the Covid-19 lockdown that precluded flights several months immediately following capture.

Figure 3. View of the Tug Hill focal area from a fixed-wing aircraft with radio telemetry antenna attached to the wing strut.

Ultimately, our final sample size was 146 individuals (67 in the Adirondack focal area, 79 in the Tug Hill area).

We documented a total of 57 deaths (22 in central Adirondacks, 35 in Tug Hill; Figure 4), due collectively to harvest (28), vehicle strikes (4), rodenticide poisoning (1), coyote predation (1), disease (1) and unknown causes (22). Overall, our estimates of survival were at the lower end, but not outside of, previously published survival rates for fisher in the northeastern U.S. (Lewis et al. 2022)—a somewhat surprising result given the fairly rapid rates of fisher population expansion in recent decades documented across the southern tier of NY State. Female fishers tended to have ~17% higher annual survival probabilities (0.67, 95% credible interval 0.54-0.76) than males (0.57, CRI 0.46-0.68), although the difference was not statistically significant. As expected, adult fisher had ~37% higher annual survival probabilities (0.67, CRI 0.55-0.77) than fisher 2 years or younger (0.49, CRI 0.36-0.63). Unexpectedly, when we examined fisher survival by focal area, which required pooling all sex and age classes within a study area, central Adirondack fishers exhibited ~39% greater annual survival (0.71, CRI 0.60-0.80) than Tug Hill fishers (0.52, CRI 0.40-0.63; Figure 5). Regarding cause of death, Tug Hill fishers were nearly 3 times more likely to die from harvest than Adirondack fishers, which is evident in the rapid drop observed in survival rate for Tug Hill fisher during the hunting season (Figure 5). In contrast, Adirondack fishers were 5x more likely to die than Tug Hill fishers from non-harvest causes, and their mortalities were more spread out across the year (Figure 5). Unfortunately, we could not separate natural causes (e.g., predation) from other-than-harvest human-caused (e.g., roadkill, poisoning) to further investigate this pattern.

Figure 4. DEC Biologist Tim Watson (Region 5) holding a deceased fisher extracted from an underground burrow.

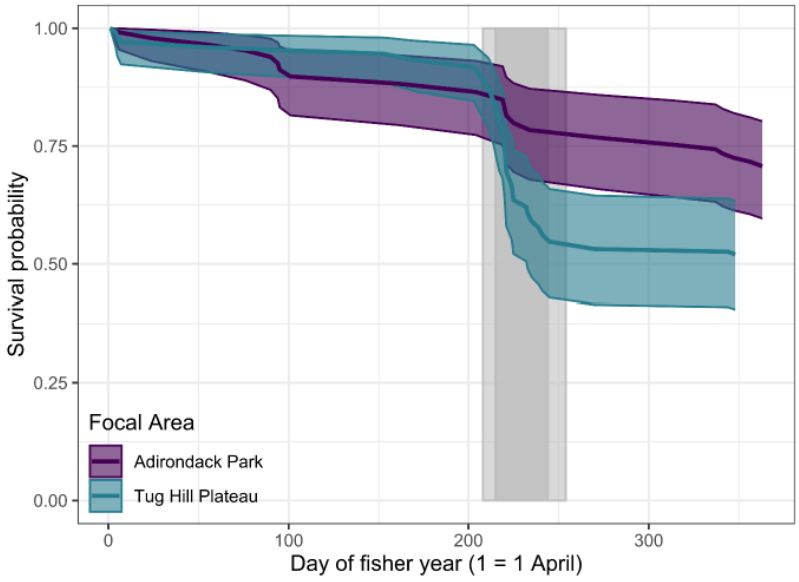

Figure 5. Annual (1 Apr–31 Mar) survival curves with 95% credible intervals for fishers in Tug Hill Plateau and central Adirondack Park, 2019–22. Vertical shading indicates fisher trapping season in the Adirondack Park (dark gray, 1–30 Nov) versus Tug Hill Plateau (light gray, 25 Oct–10 Dec).

Generally, immature individuals are expected to represent a greater proportion of harvest than adults, which is considered largely compensatory to natural mortality experienced by immature fishers, and therefore consistent with sustainable harvest management (Krohn et al. 1994). The median age

at harvest for fishers (aged via cementum annuli) was 1.6 years old (N = 24). The ages of the four animals harvested in the central Adirondacks area skewed older, with three estimated via cementum annuli to be 1.6 years old or older and the remaining animal estimated at capture to be an adult. Similarly, our entire sample of live-captured individuals in the central Adirondacks skewed towards older individuals, whereas we captured many more animals <1 year old in the Tug Hill Plateau (Table 2). Lower overall catch per unit effort in the central Adirondacks, combined with the lower ratio of juveniles:adults, indicates the potential for harvest of fisher in the Adirondack focal area to be more additive than compensatory (Lebreton 2005). While fisher populations are generally thought to be robust to an annual take of 15–25% (Douglas and Strickland 1987), small and low-density fisher populations can be at risk of extinction

Table 2. Number of fishers captured alive per unit effort (100 trap nights) by age group across northern New York, split into focal areas (Tug Hill Plateau [THP] and Adirondack [ADK]), 2019–2022. Juvenile fishers were <1 year old, subadult fishers were ≥1 year and <2 years old, and adults were ≥2 years old. For survival analyses, we classified juveniles and subadults together as immatures.

	Juvenile		Subadult		Adult		All individuals	
Trapping Season	THP	ADK	THP	ADK	THP	ADK	THP	ADK
2019	0.08	0.06	0.24	0.06	0.00	0.12	0.32	0.24
2019–2020	0.71	0.15	0.26	0.11	0.19	0.23	1.17	0.48
2020–2021	0.32	0.08	0.11	0.40	0.21	0.18	0.64	0.65
2021–2022	0.40	0.19	0.09	0.36	0.11	0.33	0.60	0.87
All years	0.41	0.13	0.14	0.25	0.15	0.23	0.71	0.61

when even 10% of the population is removed (Fogarty et al. 2022). We expect that the combination of vehicle strikes, harvest, and anticoagulant rodenticide-related mortalities together may well put the Adirondack population within that range.

With respect to potential drivers of fisher survival, we did not observe strong evidence that the relative abundance of coyotes, stand age, amount of human-wildlife interface, or overall habitat quality influenced survival rates across this region. However, we did observe an increase in harvest following mast failure years (2020 and 2022 in our study), indicating that mast crops are likely an important food source for fishers. In years when mast crops fail, fishers likely need to range further to find food and likely encounter greater harvest risk.

Fisher Productivity

In species like fisher, adult survival rates are generally high and stable whereas recruitment—the survival of offspring to the age where they become independent and subsequently join the breeding population—is often much more variable (Gaillard et al. 1998). As

Figure 6. Image of female fisher moving a kit from one den tree to another.

a result, juvenile survival has less predictable but no less important impacts on population dynamics compared to adult survival (Finkelstein et al. 2010).

Fishers give birth from early March to early April (Frost et al. 1997, Powell et al. 2003, Facka et al. 2016) and use tree cavities to raise their young (Powell 1993). Fisher kits are reared in 1–5 reproductive dens for approximately ten weeks (Paragi et al. 1996). This period of vulnerability is a critical life history stage for fishers, with kit survival hypothesized to be more limiting than kit productivity (Jensen and Humphries 2019). Litter mortality prior to weaning has been estimated to be up to 60% in fishers (Matthews et al. 2013, 2019, Green et al. 2018), though individual kit mortality is likely even greater (Matthews et al. 2019). Threats to kit survival include exposure to harsh temperatures, malnutrition due to low prey density, predation of the mother (Green et al. 2017, Matthews et al. 2019), and predation by males (Matthews et al. 2019, Smith et al. 2020).

In order to support themselves and offspring, females must forage efficiently (i.e., intake energy that exceeds the energy expended to capture prey; Powell and Leonard 1983, Jeanniard-du-Dot et al. 2017). Thus, habitat choices underlie the demographic rates that drive population growth (Morris 2003). Ideally, animals select habitats that have the highest quality and will yield the greatest fitness benefits (Fretwell 1972), with females expected to maintain a home range only as large as necessary to meet energetic requirements for their own maintenance and provisioning of offspring (Sandell 1989, Arthur and Krohn 1991, Katnik et al. 1994). Home range size will likely increase, and population density decrease, where prey distribution is less predictable and prey density is low (Clutton-Brock and Harvey 1978), such as in highly seasonal or unpredictable environments (Herfindal et al. 2005). Thus, availability and predictability of prey, including the effort required to obtain it, is linked tightly to reproductive potential (McNab 2006).

Our objective was to investigate recruitment in fishers, that is the annual production and survival of kits, along with patterns of maternal den attendance that may reveal habitat limitations. Female fishers use tree cavities to keep their kits safe when they are very small (usually the first 10-12 weeks after birth). Importantly, females move their kits to a new den tree every few weeks. Thus, to assess kit survival, we used trail cameras set at den trees to count the number of kits each female moved each time she moved her den (Figure 6). Further, kits are vulnerable to the elements and other predators when their mother leaves the den to hunt for food, and the duration of female hunting bouts should be driven by prey encounter rate. Thus, we expected that local environmental conditions should drive variation in the daily number of hunting trips (i.e., foraging bouts) as well as the duration of foraging bouts, which in turn is likely to impact kit survival.

Of the 81 female fishers collared as part of this overall study, we determined 44 to be potentially sexually mature (aged 2+ years), but ultimately were successful in tracking

Table 3. Number of female fishers ≥2 years old as of April each year that were monitored (left columns) and observed with kits (central columns) in the Tug Hill Plateau (THP) and central Adirondack (ADK) focal areas. The proportion of females detected with kits are given in the last columns.

Year	Females monitored			Observed with kits			Proportion with kits		
	THP	ADK	Total	THP	ADK	Total	THP	ADK	Total
2019	1	2	3	0	0	0	0	0	0
2020	5[a]	3	8	3[a]	3	6	0.60	1.0	0.75
2021	10	8	18	3	0	3	0.30	0	0.17
2022	5	8	13	4	6	10	0.80	0.75	0.77

19 (10 in Tug Hill, 9 in central Adirondacks) individuals known to have produced at least 1 kit. The percent of these 19 collared females observed to have produced kits ranged from 0 to 100% depending upon the year and study area (Table 3).

Despite our small sample sizes, we noted an alternative-year pattern in number of denning female fishers in our focal areas, especially within the Adirondack Park (Table 3), which may be indicative of a response to mast cycles. American beech (*Fagus grandifolia*) is the primary hard mast-producing species in the Adirondack region (McNulty and Masters 2005), and SUNY ESF researchers at the Huntington Wildlife Forest near Newcomb, NY have recorded data on beechnut production since 1988. Their records indicated mast production in August–November 2019 and 2021, which immediately preceded denning periods where 75–77% of our collared, breeding-age female fishers were observed with kits. In contrast, following the mast failure recorded in 2020, we observed only 3 of 18 breeding-age females with kits (17%). Fisher populations have been shown to respond to mast crops both in harvest vulnerability (Jensen et al. 2012, also see our survival study) and numerical increases related to increases in small mammal abundance (Jensen et al. 2012, Greenhorn et al. 2021). However, small mammal populations do not increase until the spring or summer after mast crops, and therefore we speculate that this pattern could be a result of increased winter food availability of the beechnuts themselves, as over-winter prey availability can influence breeding propensity (Orlando et al. 2023), if not juvenile survival. Further, we monitored three female fishers over two breeding seasons each, and each was observed denning in only one of the years they were observed, providing anecdotal evidence for alternate-year breeding schedules.

We attempted to locate collared females 2-3 times/week beginning mid-March via aerial and ground telemetry. When a given female was found in the same tree on ≥2 consecutive occasions, we set up 2-6 Reconyx PC800 or Hyperfire 2 Covert (Reconyx Inc, Holmen, WI, USA) or Browning Spec Ops BTC8 (Browning Trail Cameras, Birmingham, AL, USA) cameras facing the tree. We attempted to download GPS and acceleration data from the collar whenever the female was found in the den and continued to relocate her at least once per week after kits were weaned and moving on their own. At each den tree we checked cameras every 2–10 days and initiated a search for a new den after observing photos of the female moving kits away from the focal tree. Occasionally, we misidentified a rest site as a den, and after failing to detect a fisher at the site for several consecutive days we moved the cameras. During each den transfer we counted the number of kits moved by each female, using photo timestamps to differentiate the transfer of individual kits as well as temporary movements (i.e., mom carrying a kit out and then back in or exploratory movements by kits) versus den relocation. From our sample of collared females, we monitored 78 dens and obtained kit counts at 62 dens, with 47 of those being occasions when the female carried the kits to a new den. The litter sizes we observed ranged between 1 and 4 kits per female (Table 4), averaging 2.42 (0.90 SD) and being slightly higher for the Tug Hill population (2.70 ± 0.94 SD) than the central Adirondack population (2.11 ± 0.78 SD).

We estimated parturition date (i.e., birth date) for each litter based on multiple lines of evidence from the female's GPS and accelerometer data, e.g., decreased travel distances and clustered locations (following Hooven et al. 2022). Estimated parturition dates ranged from 14 March to 9 April, with an average date of 25 March (Table 4). We estimated the average initial litter size, at the time of parturition, to be 3.40 kits on average (95% Credible Interval 2.37-3.50). Studies of captive and wild fisher indicate most kit deaths occur early in the denning period, before our monitoring of kit counts began, usually due to exposure and maternal neglect (Frost and Krohn 1994; Matthews et al. 2019).

Because we did not attempt tree cavity checks and instead depended on photo captures for kit counts, it is difficult to capture the differences in survival rates in the early days, even given our probabilistic estimates of initial litter size. Therefore, we interpret our estimated kit survival rates as an average, and potentially slightly higher than true kit survival. Relatedly, our probabilistic estimates of initial litter size are potentially biased low, though the model estimate of 3.40 kits per female is higher than litter sizes

Table 4. Summary of estimated parturition date, initial camera set dates, and kit counts for each monitored female fisher.

Year / Fisher	Study Area	Maternal Age (years)	Estimated parturition date	First cameras set	Dens Identified	# times kits counted[c]	Maximum count	End count
2020								
19F501	ADK	3	28-Mar	23-Mar	3	1	2	2
19F503	ADK	2	14-Mar	20-Mar	4	2	3	3
20F501	ADK	5	21-Mar[a]	23-Mar	1	1	1	NA
20F610	THP	2	26-Mar	26-Mar	5	3	2	2
20F614	THP	4	19-Mar	24-Mar	4	2	3	2
20F615	THP	5	25-Mar	24-Mar	3	3	2	2
2021								
20F612	THP	5	2-Apr	5-Apr	3	2	3	3
20F617	THP	3	25-Mar	5-Apr	4	3	3	3
20F621	THP	3	28-Mar	1-Apr	4	3	1	1
2022								
19F504	ADK	4	26-Mar	28-Mar	5	3	1	1
20F504	ADK	3	9-Apr	12-Apr	4	5	2	2
21F507	ADK	3	28-Mar[a]	7-Apr	1	1	3	0
21F508	ADK	2	28-Mar[a]	28-Mar	4	3	3	3
22F502	ADK	2	28-Mar[a]	30-Mar	5	4	2	1
22F505	ADK	2	26-Mar	31-Mar	3	2	2	1
19F603	THP	4	21-Mar[a]	1-Apr	4	3	4	4
20F607	THP	3	27-Mar	11-Apr	5	2	3	3
21F606	THP	3	21-Mar[a]	25-Apr	1	1	2	NA
21F611	THP	4	15-Mar	6-May[b]	3	2	4	4

observed by Frost and Krohn (1994).

We defined the denning period (during which time we would estimate kit survival) as birth until the day before kits were first observed crawling or climbing outside of the den under their own power. The denning period ultimately extended 77 days (11 weeks), during which time we detected kits with high certainty (probability = 0.94, 95% Credible Interval 0.91-0.98). The use of regular den sites declines after kits are weaned (Aubry and Raley 2006), and along with it, our ability to relocate females and obtain kit counts. During our denning period, we assumed that two females lost their litters, one due to death of the mother and the other given behaviors indicative of litter loss or abandonment (e.g., tree not being used regularly for several weeks).

We estimated the average probability of kit survival for the 77-day denning period to be 0.76 (CRI 0.63-0.86, ranging 0.15 (CRI 0.001-0.56) – 0.99 (CRI 0.94-1.00) among individuals and years (Figure 7, top left panel). We did not detect an effect of maternal age or the number of days below freezing on kit survival. However, kit survival rates decreased and became more variable with increasing stand age (Figure 7, left panels) and alternatively with decreasing amounts of wildland-urban intermix (WUI; Figure 7, right

panels), with WUI determined as a function of the number of buildings embedded within forested-dominated landscapes. Notably, stand age increased and WUI decreased along a west-east gradient. Figure 7 indicates how WUI proved less informative for kit survival than stand age, with Tug Hill fisher exhibiting relatively high levels of survival across the range of WUI values. In the Adirondacks, increasing WUI may be associated with forest clearings that may enrich the local prey base and enhance fisher habitat. Edge habitats, such as those created by development, often have high densities of small mammals (Osbourne et al. 2005), and thus offer ideal foraging areas for predators (Šálek et al. 2010). Indeed, the younger stand ages experiences by Tug Hill fisher corresponded to consistently higher kit

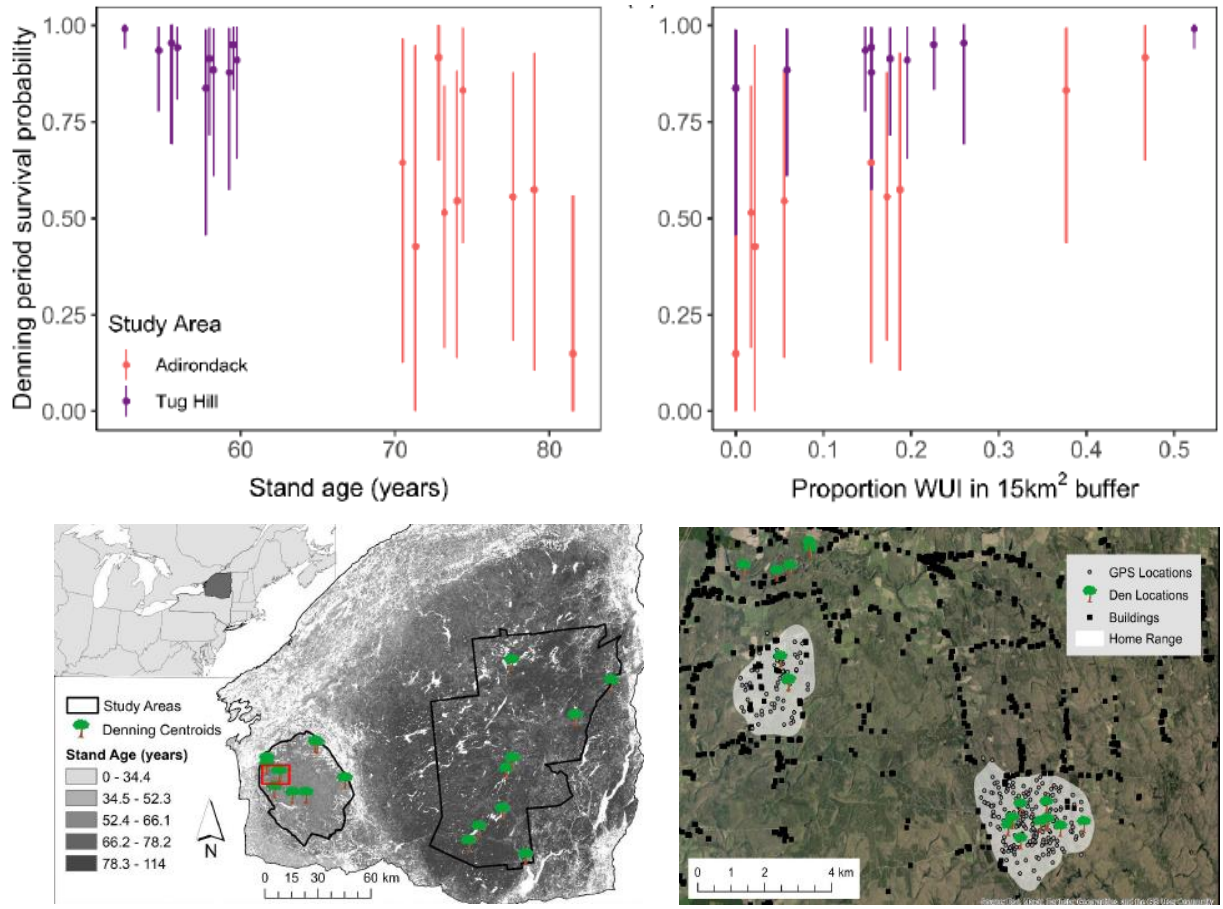

Figure 7. Predicted relationship between fisher kit survival in northern New York state and forest stand age (left) and proportion of wildland-urban intermix (WUI; right). Upper panels show the estimated 11-week (denning period) survival probability for kits reared by each female. Lower panels show the distribution of den locations and fisher home ranges superimposed on forest stand age and the distribution of human dwellings and other structures.

survival than observed for Adirondack fisher.

In comparison to the Tug Hill population, denning females in the central Adirondacks tended to maintain larger home ranges (Figure 8 top), moved further distances away from dens during foraging bouts (Figure 8 bottom), and had longer foraging bouts (42 minutes longer on average), likely reflecting lower prey availability in the Adirondacks compared to outlying regions. For example, snowshoe hare (*Lepus*

americanus), an important prey item for fishers (Powell 1993), often use forests <40 years old (Kawaguchi and Desrochers 2018), and the aging Adirondack forests have exhibited a long-term decline in hare abundance (NYS DEC, unpublished data). In California, Wengert et al. (2014) found that bobcats killed significantly more female than male fishers, and that nearly three quarters of the fishers killed were between March and July when kits were still dependent on the mother (Powell 1993). Thus, given its impact on the amount of time mothers spend away from dens, prey availability may impact recruitment directly through kit survival as well as indirectly via risks to the mother. Our observations indicate that the environmental conditions within the central Adirondack region are more challenging for breeding fishers than the adjacent Tug Hill Plateau, with larger home range sizes and lower kit recruitment largely explaining the perceived differences in population sizes between these two regions.

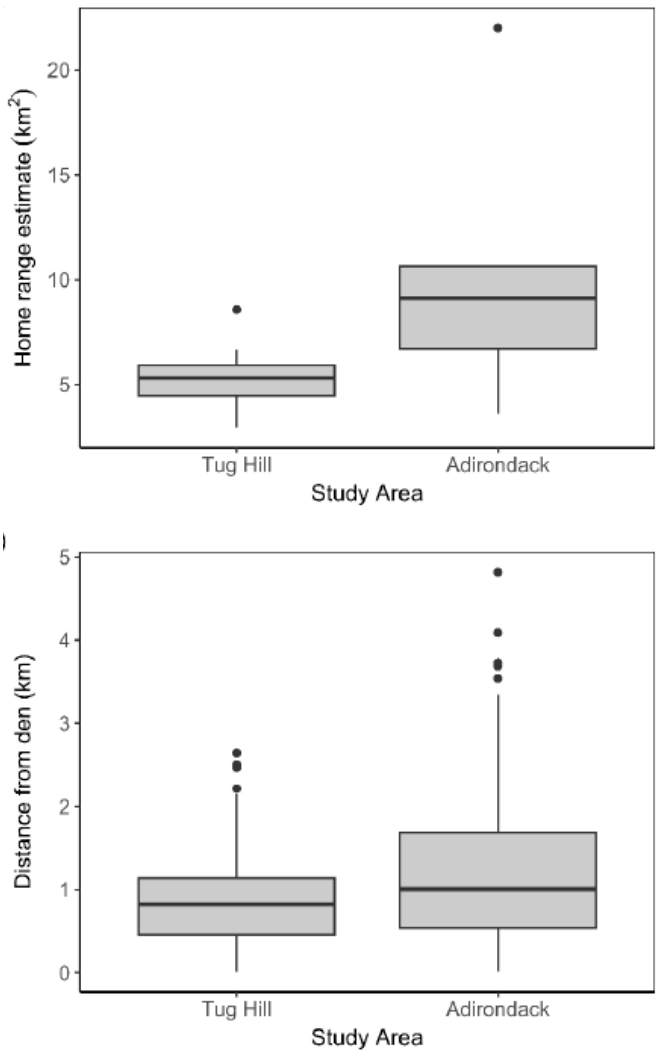

Figure 8. Comparative size of home ranges by breeding female fishers in the two focal areas (top), and the distribution of distance travelled away from den trees during foraging bouts (bottom). The mean is indicated by the thick black horizontal line while the 95% confidence region is shown in gray shading and outliers are indicated by black dots.

Anticoagulant rodenticide exposure in NY fishers

Rodents have been recognized as a threat to human health and livelihoods for millennia (van den Brink et al. 2018). Our defense arsenal includes physical methods (e.g., traps, barriers), biological/cultural remediation (e.g., sanitation, habitat manipulation, resistant plants, house cats), and chemical methods (e.g., toxic baits, fumigants, repellents). In both commercial and private applications, use of toxic compounds is a primary choice because they can remain potent for long deployment periods and a single bait station can dose numerous individuals (compared to a snap trap that can catch only one animal at a time before needing to be rebaited and reset). The most popular toxic compounds are anticoagulant rodenticides (ARs; Eisemann et al., 2018), which interfere with blood clotting ability and render animals that have ingested ARs vulnerable over time to fatal hemorrhage precipitated by minor trauma, exertion, or other factors (Rattner and Mastrota, 2017; Stone et al. 2003).

The various AR compounds on the market include first-generation ARs (FGARs; developed in the 1950s) and, in response to resistance by target pest populations, second-generation ARs (SGARs; developed in the mid-1970s; Jacob and Buckle, 2017; Pelz et al., 2005). Being more acutely toxic than FGARs, SGARs require only a single feeding to deliver a lethal dose (Rattner and Mastrota, 2017; Vandenbroucke et al., 2008) and have longer elimination half-lives (taking up to hundreds of days to metabolize from the liver; Vandenbroucke et al., 2008). Whereas in the US, SGARs are supposed to be restricted to professional or agricultural applicators (US Environmental Protection Agency, 2008), FGARs remain widely available to the public. In terms of deployment, ARs are sold as powders or pellets that may be placed directly in rodent burrows or in durable paste (block) form or soft pouch placed in water-proof bait stations that can be used inside and outside of buildings.

Importantly, ARs (both FGARs and SGARs) act the same whether consumed directly by ingesting bait or indirectly by predating or scavenging AR-laden prey (Orsted et al., 1998). Altered behavioral patterns of AR-laden lab rats, such as shifted diurnal patterns and use of areas away from cover, indicate that ARs may predispose toxic rodents to predation in the wild (Cox and Smjtii 1992). Despite regulations designed to protect children, pets, and non-target wildlife from exposure, detection of ARs in predators and scavengers has proven alarmingly widespread where it has been investigated (Cooke et al., 2022; Badry et al., 2021; Murray, 2020; Wiens et al., 2019; Elmeros et al., 2018; Lohr, 2018; Ruiz-Suárez et al., 2016; Serieys et al., 2015; Gabriel et al., 2012; McMillin et al., 2008). For example, in California 83-96% of tested black bear (*Ursus americanus*), bobcat (*Lynx rufus*) and fisher (*Pekania pennanti*) showed evidence of ≥1 AR in their system, most having been exposed to 2-5 different compounds (McMillin et al. 2018; Serieys et al. 2015; Gabriel et al. 2012; Riley et al. 2007; Riley et al. 2003). As commercial products contain a single AR compound (US Environmental Protection Agency, 2008), residues of ≥1 compound indicate different exposure events and, by extension, repeated or chronic levels of exposure.

All ARs, but especially SGARs, pose substantive risks to non-target wildlife. In addition to directly causing mortality through internal bleeding, sublethal effects of AR exposure include increased susceptibility to parasites and disease, decreased litter size, and increased newborn death (Serieys et al., 2015; Rady et al., 2013; Lemus et al., 2011; Robinson et al., 2005). Although public concern over the high prevalence of ARs in non-target wildlife may be escalating, there remains a lack of comparably effective alternatives for protecting human health and livelihoods from the risks posed by rodents (Quinn et al., 2019; Rattner et al., 2014). As such, greater insight is needed regarding the factors that drive AR exposure in wild carnivores to balance human safety more effectively with healthy wildlife populations.

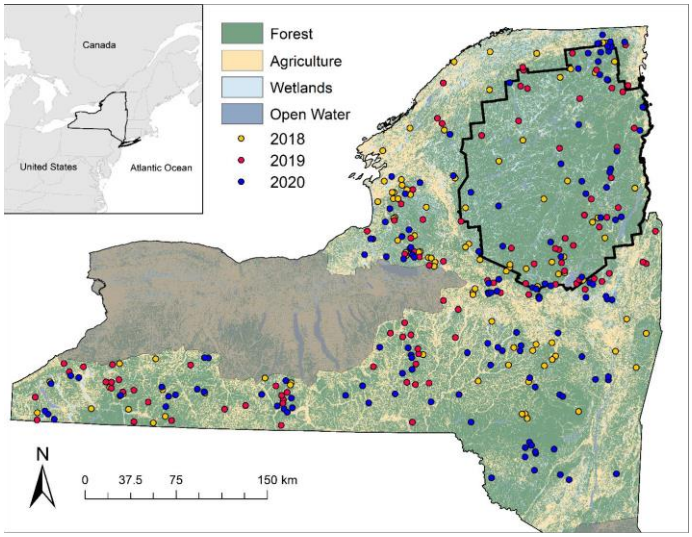

Figure 9. Map of New York showing the location of harvested fisher used to test for exposure to anticoagulant rodenticides. Note that the lake plains region in the northeastern part of the state was closed to harvest during this study.

Our objectives were to document the magnitude of AR exposure in fisher across

NY State and to identify demographic (age, sex) and environmental (agriculture, buildings) factors that may influence fisher exposure. We anticipated that years following a high mast crop would see greater AR exposure in fishers due either to 1) increases in small mammal populations and, by extension, increased application of ARs or 2) fishers ranging further in mast failure years to find food and encountering more AR sources along their travels.

We acquired fisher liver samples from carcasses submitted to the NY DEC as part of the regulated fall harvest, acquiring 100 samples in each of two years (2018 and 2019) and 138 samples in 2020. Fisher age was determined by cementum annuli, and sex was determined by the presence/absence of a baculum. We selected samples to capture variation in forest amount to ensure exposure to forest-dominated as well as agriculture or urban-dominated areas (Figure 9). While we tried to balance age and sex in our sample, the sample was skewed towards adult females as part of the concurrent study on fisher productivity. Ultimately, we sampled 112 males and 226 females aged 0.5-8.5 years old.

Each liver sample was tested for seven FGARs (warfarin, dicoumarol, coumafuryl, chlorophacinone, coumachlor, diphacinone, pindone) and four SGARs (brodifacoum, bromadiolone, difenacoum, difethialone) at the Pennsylvania Animal Diagnostic Laboratory System in New Bolton, PA. We note that dicoumarol can be found naturally in sweet clover hay that has become moldy (Slankard et al. 2019) and its detection may not reflect use of this compound within a commercial product.

Table 5. Number of individual fishers in NY State (out of 238) detected with residues of the indicated anticoagulant rodenticide compound. Type references first- versus the more potent and longer-lasting second-generation compounds.

Compound	Type	Detections
Chlorophacinone	First gen	8 (2%)
Dicoumarol	First gen	23 (7%)
Diphacinone	First gen	249 (74%)
Warfarin	First gen	2 (<1%)
Brodifacoum	Second gen	196 (58%)
Bromadiolone	Second gen	101 (30%)
Difenacoum	Second gen	3 (<1%)
Difethialone	Second gen	12 (3%)

We detected eight different AR compounds in fisher (Table 5), with 83% of tested animals exhibiting residues of at least 1 and at most 5 different ARs. The great majority of animals (79%) tested positive for one (N=78), two (N=120), or three (N=69) different compounds. Diphacinone (FGAR) was the most frequently detected AR type, followed by second-generation brodifacoum and bromadiolone. Fisher age and sex were the strongest predictors of the number of AR compounds detected, with the expected number of compounds being greater in males compared to females and increasing in fishers up to age 3.9 while declining thereafter (Figure 10a). Whereas we expected subadult (i.e., 1.5–2.5 years old) animals, those likely to be dispersing through unfamiliar landscapes, to exhibit the greatest number of compounds and probability of exposure (Matthews et al. 2013), we instead observed a peak in exposure for fishers 3.5–4.5 years old, likely past the point of territory establishment and overlapping prime breeding years. Previous work examining AR exposure in mink (*Neovison vison*) likewise showed an increasing number of compounds in mink from 0 to 5 years of age, which the authors attributed to the long retention of SGAR compounds in liver tissue (Ruiz-Suárez et al. 2016). We expect that this is likely also the case in fishers and speculate that the lower exposure in older age classes that we observed could be due to smaller samples sizes of older animals, or dominant

animals occupying areas with little human encroachment (which may have also contributed to greater age at harvest).

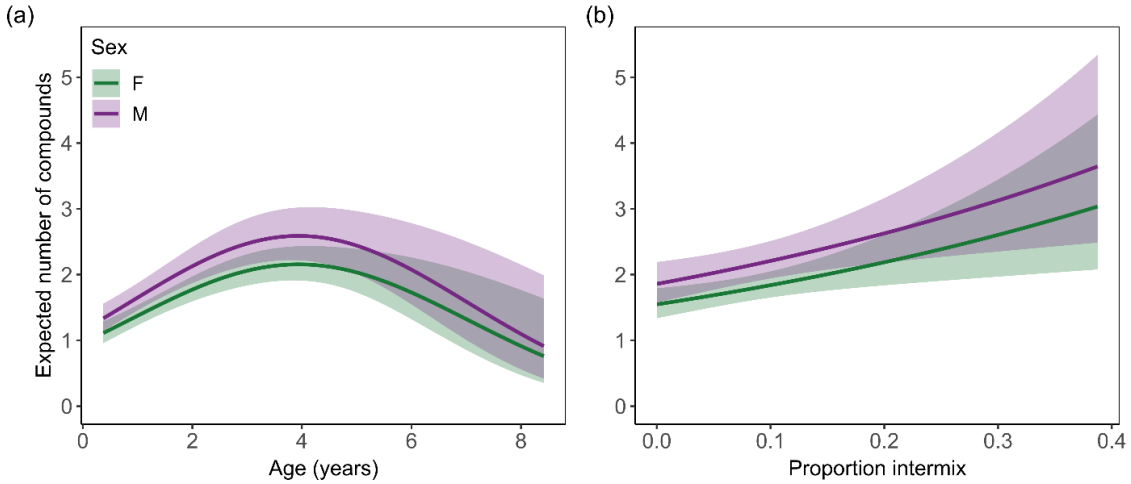

Figure 10. Predicted number of anticoagulant compounds given fisher age (a) and the proportion of the surrounding area (15 km^2) consisting of wildland-urban-intermix, with effects shown separately for male (purple) versus female (green) fisher.

The main landscape feature driving AR exposure in fisher was the amount of wildland-urban-intermix within a 15-km^2 area around the harvest location (Figure 10b). Figure 11 (next page) helps visualize what qualified as wildland-urban-intermix, differentiating between interface (concentrated areas of human development adjacent to forested landscapes) and intermix (low density dwellings and other buildings embedded within forest-dominated areas). Interface was not strongly related to AR exposure, but intermix was indicating rural households, potentially agriculture, and other rural applications of ARs as the main source of exposure for fisher. Notably, a study in California indicated that residents of neighborhoods having relatively lower housing density were more likely to suffer rodent problems and employ AR products compared to more densely settled neighborhoods (Morzillo and Mertig, 2011). This is not to say that fishers are not exposed to ARs when using landscapes closer to urban areas, but rather that we could not detect an effect of interface using our data because fisher are not harvested in and around urban and suburban areas and therefore such conditions did not make it into our sample.

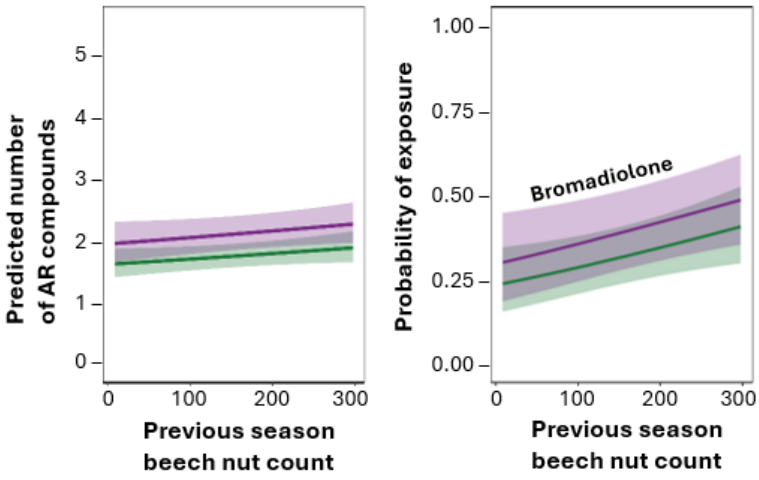

Figure 12. The expected number of compounds fisher were exposed to given lagged effects of beech nuts (left panel). Probability of exposure to a single AR type, bromadiolone, following beech nut masts (right panel).

In terms of the total number of compounds an individual was exposed to, we did not observe any statistically significant effects of percent of agriculture within 15-km^2 or the beech nut masting cycle

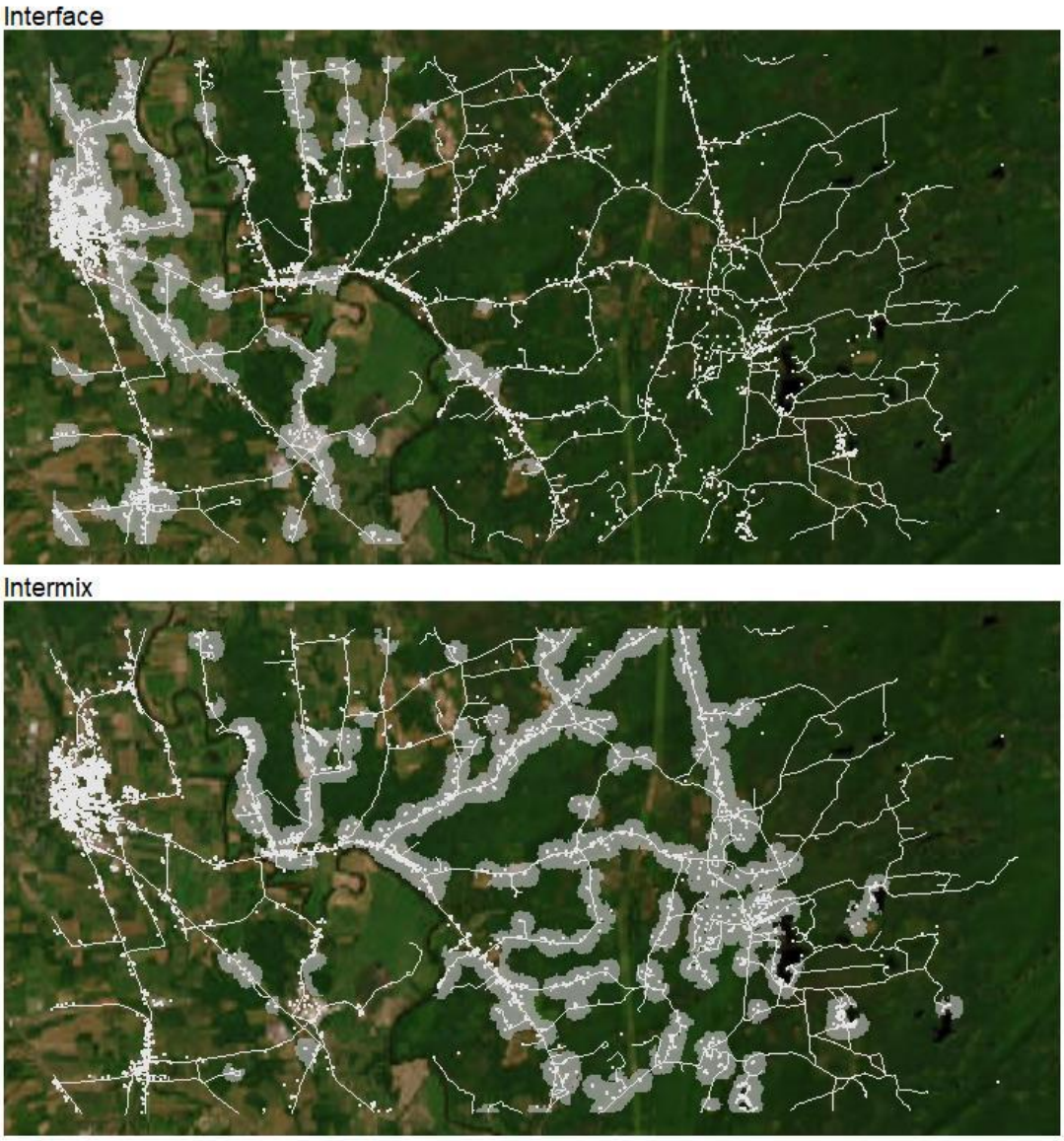

Figure 11. Sample landscape showing wildland-urban interface (top) versus wildland-urban intermix (bottom). Interface delineates zone of influence near concentrated urban centers adjacent to forest while intermix delineates zones of influence around low density dwellings embedded within forest-dominated areas. The latter, interface, is the variable that best explained fisher exposure to anticoagulant rodenticides.

(Figure 12 left panel). However, effects may be masked by having pooled together different compounds, with perhaps some compounds being preferentially used in different landscape contexts (e.g., agricultural versus home use).

When we looked at the three most detected compounds individually, we observed that fisher age and sex, and the amount of wildland-urban-intermix, continued to be strong predictors of exposure (Figure

13). Surprisingly, exposure to second-generation compounds (brodifacoum and bromadiolone) was more strongly driven by wildland-urban-intermix than the first-generation compound diphacinone, despite diphacinone expected to be more generally available to the public. In trying to determine point sources of exposure for fishers, it is important to recognize that EPA regulations cover where SGARs should be sold and used, but not who can purchase them, meaning that even SGARs remain accessible to the public through local hardware stores and online venues. As a result, detection of SGARs does not

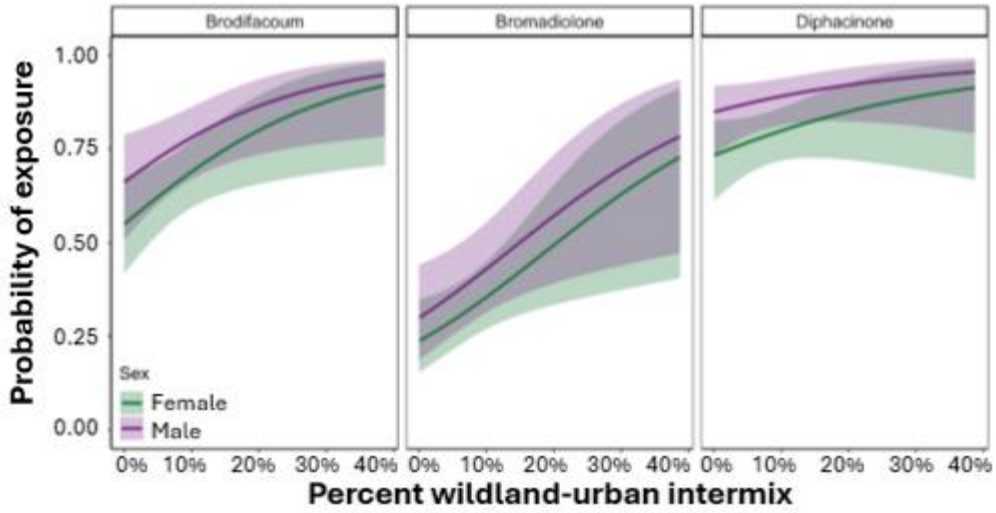

Figure 13. Predicted probability of anticoagulant exposure (with 95% confidence intervals) for female (green) and male (purple) fisher given the percentage of wildland-urban-intermix (top) within 15-km^2 of the fisher's harvest location. Effects are shown for the three most common anticoagulant compounds.

point specifically to a commercial pest control or agricultural application (Bartos et al., 2011). Moreover, given that SGARs are retained longer than FGARs within the animals system, the frequency of detection of each within fisher may not reflect the relative magnitude of their use on the landscape.

While it seems logical that licensed professionals would be more likely to adhere to regulations given that state licensing requires education about safe and effective pesticide use, there are limited data available to test this assumption. There is, however, evidence of widespread non-compliance with regulations in public use of ARs (Bartos et al., 2011; Tosh et al., 2011), i.e., deploying more than is necessary, further from buildings than allowed, using indoor products outdoors, or not placing baits within specialized containers. Moreover, up to 80% of surveyed households in California and farmers in Ireland performed pest control on their own without hiring a contractor (Bartos et al., 2011; Tosh et al., 2011). Given these observations, it seems likely that off label, non-professional use of ARs is driving the high non-target wildlife exposure rates observed in NY. However, potential sources of commercial application within forests also would be consistent with wildland-urban-intermix, such as logging encampments, maple sugaring operations (Houston et al., 1990), and even maintenance buildings associated with protected areas.

In terms of other drivers of exposure to individual compounds, percent agriculture was not a significant predictor while the magnitude of the previous years' beech nut crop predicted exposure solely to bromadiolone (Figure 12 right panel). A complicating factor in determining agriculture effects is that concentrated agricultural land use coincides with areas where fisher populations are considered too

sparse to support a sustainable harvest. As a result, working with samples from the regulated trapping season, we likely lacked sufficient geographic coverage of exposure to agricultural land uses to effectively detect their impacts on fisher.

Lagged effects of beech masting explained some variation in overall number of compounds a fisher was exposed to as well as the probability of exposure to bromadiolone, although the effects were not as strong as we anticipated. Importantly, beech trees are not the only mast-producing species in the northeast and not all tree species have synchronized masting cycles (Schnurr et al. 2002). Consequently, our detection of any effect bears further consideration. The lack of effects on probability of exposure to diphacinone was surprising, as we expected the lagged beechnut count to act as a proxy for increased rodent populations the following year, and for that increase to spur homeowners to more expansive use of rodenticides. However, probability of exposure to bromadiolone, one of the compounds favored by pest management professionals (Memmott et al. 2017, Murray 2017), did increase with the lagged masting index. Pest management professionals may be hired by individuals or businesses to control a range of pests, rodents included, so we speculate that large rodent outbreaks may trigger a call to a professional instead of home or business owners attempting to handle it themselves. However, as noted previously, bromadiolone products are sold off the shelf in farm stores, so also may be accessible for private rather than professional deployment. The effects we detected, while minor, justify further consideration of masting cycles and other ecological drivers of small mammal populations and the use of rodenticides to contain them, especially as climate change is expected to increase small mammal populations in the region (Stenseth et al. 2003, Murray 2017) as well as the number and variability in mast seeding events (Allen et al. 2014).

An important aspect of managing rodenticide exposure is considering the patterns and psychology of AR use by individuals and pest management professionals. A survey of farmers in Northern Ireland revealed that adherence to best practices for reducing secondary exposure is variable and infrequent (Tosh et al. 2011b), and we assume that this is likely the case in North America as well. California pest management professionals expressed concern over the use of rodenticides by the general public (Steinberg et al. 2015), but even professional applicators have shown inattention to best practices (Memmott et al. 2017). In NYS, it is unknown how many or how often homeowners apply rodenticide products to their property, or the triggers that prompt use of rodenticide instead of traps. Proper application of rodenticide compounds could go a long way to reducing the impacts of secondary exposure (Tosh et al. 2011, Memmott et al. 2017), and snap traps could reduce risk even further. Changes in human behavior will likely only be achieved through education on rodent behavior, how rodenticides work, risks of secondary exposure, and the consequences of improper use (U.S. Environmental Protection Agency 2008, Memmott et al. 2017).

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