

Post-release survival of translocated fishers: implications for translocation success

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Abstract

As a vital tool for the conservation of species at risk, translocations are also opportunities to identify factors that influence translocation success. We evaluated factors associated with post-release survival of 90 radio-tracked fishers (*Pekania pennanti*) translocated from central British Columbia, Canada, to the Olympic Peninsula of Washington, USA, from 2008 to 2011. We hypothesized that the survival of translocated fishers would be affected by the same factors that influence the survival of resident, native fishers (i.e., sex, age, season, body condition), and additional factors that were associated with the translocation process (e.g., duration of captivity, release date, yr of release). Fisher survival was most strongly influenced by translocation year (i.e., release-yr cohort), season, sex, and age class of fisher; whereas duration of captivity, standardized body mass, release date, and number of intact canines did not influence survival. Survival was lowest for fishers released in cohort 2 in 2009 and during the breeding season (Mar–Jun), and was greatest for juveniles and males. When combined across release-year cohorts, year 1 survival rates were greatest for juvenile males followed by juvenile females, adult females, and adult males. Sex and age-related differences in survival of translocated fishers were counter to those commonly reported for established fisher populations, where adult females often have the highest survival rates and juveniles the lowest. Predation (40%) and vehicle strikes (20%) were the most common



causes of known mortality among the 24 recovered fishers for which cause of death was determined. We speculate that females face higher risks of mortality in translocated populations because their small size makes them more vulnerable to predation and because adult females in resident populations are less likely than males and juveniles to disperse. Our findings support designing translocations that favor releasing a preponderance of female fishers in recognition of their lower survival rates and to ensure adequate breeders are established in the population, and juvenile and young adult fishers to enhance survival of both sexes. Releases conducted over multiple years will minimize the impact of stochastic annual events that may adversely affect survival in any given year. Persistence, widespread distribution, and documented reproduction of fishers within our study area for ≥ 6 years following the last releases indicate that survival parameters we measured contributed toward successful population establishment over the short term.

KEYWORDS

carnivore, endangered, fisher, forest, *Pekania pennanti*, survival, translocation, Washington

The goal for most conservation translocations is the reestablishment of a self-sustaining population (Seddon 2010, Jachowski et al. 2016). Many translocations have failed to meet this goal and success is far from assured (Griffith et al. 1989, Yalden 1993, Wolf et al. 1998, Miller et al. 1999, Fischer and Lindenmayer 2000). Despite the importance and extensive use of translocations to conserve species at risk (Seddon et al. 2014), little is known about factors that influence translocation success, including the survival of translocated individuals for many species. A greater understanding of these factors is important to managers that must prioritize limited funding for conservation actions with the greatest likelihood of success.

Successful population reestablishment or augmentation through translocation clearly relies on survival of translocated individuals (Armstrong and Seddon 2008). In the absence of information on anticipated survival rates in founding populations, managers may be able to offset the effects of potentially low survival rates by releasing large numbers of individuals (Devineau et al. 2010) or continuing releases until evidence of success is apparent. Although releasing large numbers of individuals is associated with translocation success (Lewis et al. 2012, Powell et al. 2012), many biologists must rely on limited founder populations and best available information to conduct successful translocations.

In the contiguous United States, reintroductions have played a key role in restoring populations of the fisher (*Pekania pennanti*), a mid-sized carnivore of the family Mustelidae, which occurs only in the temperate and boreal forests of North America (Powell 1993, Lewis et al. 2012). The extremely high prices paid for fisher pelts in the late 1800s and early 1900s (up to ~\$350 US/pelt; Seton 1926, Bailey 1936, Grinnell et al. 1937, Dalquest 1948) resulted in overexploitation and the extirpation of fishers from much of the southern portion of their range in the northern United States and southern Canada (Strickland et al. 1982, Powell 1993). Fishers were effectively eliminated throughout Washington, USA, by the late twentieth century (Powell 1993, Lewis et al. 2012), at which time fishers were listed as an endangered species in Washington (Hayes and Lewis 2006). From the mid-1900s to 2012, there were 35 documented translocations of fishers within the species' historical range; 25 (71%) of these



attempts were successful (Lewis et al. 2012). As a result, the fisher is one of the most successfully translocated carnivores (Breitenmoser et al. 2001, Powell et al. 2012).

Improvements in planning, implementation, and monitoring processes have led to greater rates of translocation success in recent decades (Seddon and Armstrong 2016). Despite these improvements and large investments in translocation efforts, past (and recent) failures demonstrate that success should not be taken for granted (Griffith et al. 1989). Because many translocations occurred prior to the availability of radio-telemetry technology and the associated capacity for biologists to monitor survival, few projects have quantified survival of translocated fishers. Thus, few insights have been gained on factors associated with survival that could assist with future translocation programs. Many intrinsic attributes (i.e., sex, age, body mass, canine condition) and extrinsic factors (e.g., season), however, may influence the survival of fishers in resident populations (Douglas and Strickland 1987, Powell 1993, Koen et al. 2007, Buskirk et al. 2012, Sweitzer et al. 2016), providing the basis for evaluating survival in translocated populations. Moreover, survival of translocated fishers may also be affected by the capture and translocation process itself (e.g., yr of release [cohort], date of release, number of broken canine teeth), and the stress associated with captivity (Hartup et al. 2005, Teixeira et al. 2007, Franceschini et al. 2008, Dickens et al. 2010).

As a first step toward restoring a viable population of fishers in Washington (Hayes and Lewis 2006), a partnership of state, provincial, and federal agencies, and private organizations translocated 90 fishers from central British Columbia, Canada, to Olympic National Park in northwestern Washington from 2008 to 2010 (Lewis et al. 2018). We monitored the survival of released fishers to evaluate factors that may influence survival and to provide the first sex- and age-specific survival rates for a translocated fisher population. Our specific objectives were to 1) evaluate intrinsic (e.g., sex, age, body mass, canine condition), extrinsic (e.g., season), and translocation-process factors (e.g., yr of release, release date, duration of captivity) that could influence the survival of fishers released on the Olympic Peninsula; 2) estimate sex- and age-specific survival rates for the founder population; 3) determine the causes of mortality among recovered fishers; and 4) compare the survival rates and causes of mortality between this translocated population and resident fisher populations in other parts of the species' range.

STUDY AREA

We conducted this study from January 2008 to December of 2011 in an area (14,412 km²; Figure 1) that encompassed most of the Olympic Peninsula in northwestern Washington. The Olympic Peninsula is bordered by the Pacific Ocean to the west, the Strait of Juan de Fuca to the north, and Puget Sound to the east (Lewis et al. 2016). The study area was delineated by the areal extent of telemetry locations obtained for all fishers released on the Olympic Peninsula. The center of the Peninsula is dominated by the Olympic Mountains, including glaciated headwaters, steep upper drainages, and lowland valleys that radiate outward in all directions from the mountainous core of the Peninsula. Elevations range from sea level to 2,415 m at the top of Mount Olympus near the center of Olympic National Park. The mountainous center of the Peninsula slopes to a pronounced coastal plain to the west and smaller plains to the north and east.

The Olympic Peninsula has a temperate maritime climate characterized by warm summers (21 Jun–21 Sep) and cool winters (21 Dec–21 Mar; Peel et al. 2007). Annual precipitation ranges from 315–500 cm on the west slope of the Olympic Mountains, whereas annual precipitation is typically <40 cm in the rain shadow of the Olympic Mountains (the NE corner of the Olympic Peninsula; Gavin and Brubaker 2015). Eighty percent of the annual precipitation on the Olympic Peninsula falls from October through March and most winter precipitation falls as rain at elevations <305 m and as snow above 760 m.

The moist climate and broad range of elevations profoundly affect vegetation patterns throughout the Olympic Peninsula, which are dominated by conifer forests. A large portion of the Olympic Peninsula is composed of Olympic National Park, which contains about 2,850 km² of forest and is managed primarily as wilderness. Olympic National Forest bounds much of the Park and encompasses mountainous terrain that is managed for multiple uses and the development of late-seral conifer forests under the Northwest Forest Plan (U.S. Department of Agriculture

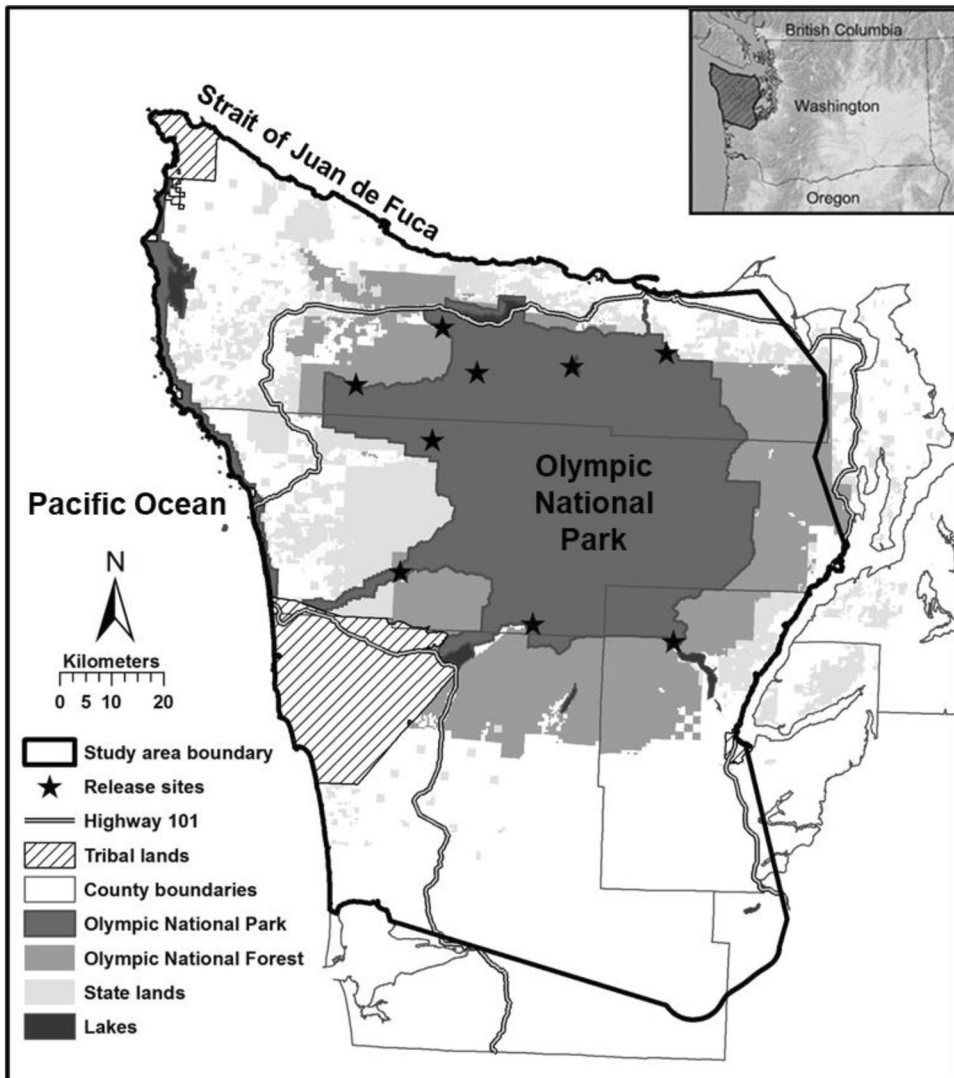


FIGURE 1 Study area and land ownerships on the Olympic Peninsula, Washington, USA, where we released and monitored fishers from 2008 to 2011. Stars indicate general areas where we released fishers from 2008 to 2010

Forest Service and U.S. Department of Interior Bureau of Land Management 1994). Lower elevation lands outside the Olympic National Forest boundary are owned and managed by multiple ownerships and are intensively managed for commercial timber production. United States Route 101 is the largest highway on the Olympic Peninsula, and it encircles much of study area (Figure 1).

In addition to translocated fishers, the Olympic Peninsula study area is occupied by a diverse mammalian carnivore community, including American black bears (*Ursus americanus*), coyotes (*Canis latrans*), mountain lions (*Puma concolor*), bobcats (*Lynx rufus*), river otters (*Lontra canadensis*), Pacific martens (*Martes caurina*), western spotted skunks (*Spilogale gracilis*), long-tailed weasels (*Mustela frenata*), and ermine (*Mustela erminea*; Happe et al. 2020). Primary prey species of fishers on the Olympic Peninsula include snowshoe hares (*Lepus americanus*), mountain beavers (*Aplodontia rufa*), and sciurids (Happe et al. 2021).



METHODS

Capture, translocation, and radio-telemetry

Hired contractors coordinated fisher trapping activities with British Columbia trappers, housed and cared for captive fishers, and assisted with preparations for fisher translocation. Contractors provided trappers with box traps (~25 × 30 × 81 cm; cage-type; made of wire mesh) and specialized transport boxes (40 × 40 × 90 cm; made of plywood) for safely moving fishers from a capture location to the captive facility and subsequently to the Olympic Peninsula (Lewis 2006, 2013). Trapping for fishers coincided with the commercial trapping season in British Columbia, which occurred from 1 November to 15 February (2007–2010).

Contractors retrieved fishers from trappers, transported them to the captive facility in central British Columbia, and provided an individual housing unit for each fisher. Each fisher was provided straw bedding, a litter box, water *ad libitum*, and a diet (raw meat and eggs) that promoted weight gain. While at the captive facility, each fisher was chemically immobilized and examined by a licensed veterinarian. Ninety fishers (50 females, 40 males; Table 1) were certified by the veterinarian as suitable for translocation to Washington. Each certified fisher was vaccinated for rabies (IMRAB-3; Boehringer Ingelheim, Ingelheim, Germany) and canine distemper (Purevax; Boehringer Ingelheim), treated with ivermectin (Merck, Kenilworth, NJ, USA) and Dronsit (Praziquantel; Bayer AG, Leverkusen, Germany) for parasites, equipped with a radio-transmitter and a passive integrated transponder (PIT) tag, measured, weighed, and photographed. We extracted 1 first premolar to age each fisher, and took blood and tissue samples to assess disease exposure and for genetic analyses, respectively. We also assessed the condition of each fisher's teeth; 82 of the 90 certified fishers had 3 or 4 intact canines, whereas 8 (7 females, 1 male) had only 2 canines as a result of tooth loss or breakage prior to project initiation, or as a result of the translocation process. Among the 90 certified fishers, time in captivity ranged from 2 to 55 days ($\bar{x}$ = 21 ± 12 [SD] days). Five fishers (3 females, 2 males) that received special medical treatment were kept in captivity for 32 to 55 days ($\bar{x}$ = 42 ± 9 days).

TABLE 1 Number of fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011, to assess survival by release-year cohort, release dates, sex, and age class

Cohort (release dates)	Age class ^a	Females	Males
Release-year cohort 1 (27 Jan 2008, 2 Mar 2008)	Juveniles	3	5
	Adults	9	1
	Total	12	6
Release-year cohort 2 (21 Dec 2008, 17 Jan 2009, 23 Feb 2009)	Juveniles	6	11
	Adults	14	0
	Total	20	11
Release-year cohort 3 (24 Dec 2009, 21 Jan 2010, 20 Feb 2010)	Juveniles	7	16
	Adults	11	7
	Total	18	23
Total fishers released		50	40

^aFor females, juveniles were <1 year old, and adults were ≥1 year old. For males, juveniles were <2 years old, and adults were ≥2 years old. We used the cementum annuli from an extracted premolar to age each fisher.



To facilitate post-release monitoring, we equipped 82 (50 female, 32 male) fishers with a 40-g very high frequency radio-collar (model MI-2 with mortality sensor, Holohil Systems, Carp, Ontario, Canada). The antenna broke on several of these radio-collars (via normal fisher behaviors), which led to a corresponding loss of signal transmission distance; therefore, we also evaluated satellite collars and implant transmitters on small samples of fishers as alternatives to radio-collars. Specifically, we equipped 5 males (each weighing >4.5 kg) with 120-g Argos satellite-collars (Kiwisat 202, Sirtrack, Havelock North, New Zealand) and 3 males with a 41-g very high frequency transmitter (model IMP/310/L with mortality sensor, Telonics, Mesa, AZ, USA) surgically implanted in the abdominal cavity.

From January 2008 to February 2010, we transported fishers from British Columbia and released them in Olympic National Park. We transported fishers by truck from central British Columbia to Port Angeles, Washington in 1 day, housed them overnight in their transport boxes, and released them the following morning at pre-determined release sites in Olympic National Park. We released fishers at 9 locations within the Park to facilitate fisher occupancy of landscapes dominated by late-successional conifer forest (Figure 1; Lewis 2006, 2014).

The study area was largely unroaded (Figure 1); therefore, we used primarily aerial telemetry to locate and monitor the survival status (alive or dead) of fishers, although fishers occupying areas near roads were occasionally located on the ground by tracking from vehicles and on foot. We attempted to locate each fisher and determine its survival status at least once per week. Inclement weather, the large size and rugged nature of the study area, the relatively short transmission distance of the radio-transmitters, and the extensive movements of many fishers, made it impossible to locate fishers each week. Upon discovering a mortality signal, we recovered carcasses as soon as was possible and sent them to Colorado State University (Veterinary Diagnostics Laboratory) or the University of California at Davis (Veterinary Genetics Laboratory) for an extensive necropsy and cause-of-death determination.

Survival analyses

We examined the survival status of 90 fishers over 48 months (Jan 2008–Dec 2011). For each release-year cohort, we evaluated survival status for 24 months following release. An individual fisher could have 1 of 3 survival-status designations in a specific month: alive, dead, or censored. We censored a fisher from survival analyses for any month that we could not determine its status (i.e., not found); we censored some fishers only temporarily because we subsequently relocated them alive or dead.

We structured known-fate models in Program MARK (White and Burnham 1999) to examine their relationships to intrinsic (i.e., sex, age, body mass, number of intact canines), extrinsic (season), and translocation-process factors (i.e., release-yr cohort, duration of captivity, release date). The known-fate model in MARK allowed for staggered entry (Pollock et al. 1989) of fishers into the population throughout the 3 winters when fishers were released (Table 2). We structured known-fate models that included individual covariates such as sex, age, body mass, number of days in captivity, release date, and the number of intact canines (Table 2). We used a 1-month interval as the time step in survival analyses.

We constructed 2 sets of candidate models based on *a priori* biological hypotheses to investigate the influence of intrinsic, extrinsic, and translocation-process factors on fisher survival. The first set included models that assumed constant survival and models incorporating the main effects of release-year cohort, season, age, and sex (Table 2). In models that included a seasonal influence on survival rate, we grouped the months of the year into 4 biological seasons (Table 2). The age of each fisher was determined by laboratory analysis of cementum annuli (Arthur et al. 1992) of a first premolar. Fisher ages ranged from 0 to 5 years; in candidate models that included a linear effect of age, we incremented each fisher's age by 1 for the second year after its release. We included an interaction term for a subset of models that included both the effects of season and age (season $\times$ age) and for a subset of models that included both the effects of release-year cohort and sex (release-yr cohort $\times$ sex).

We used Akaike's Information Criteria adjusted for small sample size (AIC_c) to assess support among the first set of candidate models. We considered those with ΔAIC_c scores <2 to be highly supported by the data (Burnham and Anderson 2002), and we used these highly supported models as the basis for developing the second set of candidate models.



TABLE 2 Intrinsic, extrinsic, and translocation-process variables incorporated into survival models and grouped as main and additional effects, for translocated fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011. The hypothesized relationship of each of the additional effects is provided in parentheses

Variable	Description
Main effects	
Release-year cohort	Cohort 1 (released from Jan to Mar 2008), cohort 2 (released from Dec 2008 to Feb 2009), or cohort 3 (released from Dec 2009 to Feb 2010)
Season ^a	Orientation season (from release to 28 Feb; year 1 only), breeding season (1 Mar–30 Jun), kit-rearing season (1 Jul–30 Sep), or fall-winter season (1 Oct–28 Feb)
Age	Estimated age in years at the time of release. Age was increased by 1 year at the start of an individual's second year after release.
Sex	Male or female
Additional effects	
Duration of captivity	Number of days from the capture date in British Columbia until the release date for each fisher (negative)
Release date	Timing of release, as indicated by the number of days prior to the beginning of the breeding season (1 Mar) that the release occurred
Standardized body mass	Body mass relative to sex-specific means and standard deviations; computed as (individual body mass–mean sex-specific mass)/standard deviation of sex-specific mass (positive)
Canines	Binary covariate signifying ≥ 3 or ≤ 2 (negative) intact canines

^aDates for kit-rearing and breeding seasons follow those provided by Powell (1993).

In the second set of candidate models, we evaluated improvement in model fit by adding an additional covariate to each of the most highly supported base models from the first candidate set. The added covariates were related to the translocation process or the condition of individual fishers and included duration of captivity, release date proximity to the beginning of the breeding season (release date), standardized body mass, and the number of intact canine teeth (canines; Table 2). Models that included standardized body mass, canines, or release date included a linear effect of that variable in the logit link function, which reflected the assumption that any effect linked to this variable at the time of release persisted throughout the next 2 years.

We used the Kaplan-Meier estimator (Pollock et al. 1989) to estimate empirical survival rates and developed 95% confidence intervals following methods provided by Cox and Oakes (1984). To estimate seasonal and annual survival rates, we calculated the product of monthly survival rates to produce a cumulative survival rate for a season or an entire year.

RESULTS

The first candidate model set comprised 18 models that included only main-effect variables (release-year cohort, season, sex, and age; Table 3). Among these 18 models, 3 were highly supported by the data (i.e., $\Delta AIC_c < 2$) and collectively represented 75% of the total model weights (w_i ; Table 3). Release-year cohort and season were included in all 3 highly supported models, whereas sex and age were each included in 2 (Table 3).



The second candidate model set included these 3 base models (highest ranking models from the first candidate model set; Table 3) and models adding a covariate to each base model. In the second candidate set, no single model stood out as the best model; rather, 8 of the 15 models were highly supported by the data (i.e., $\Delta AIC_c < 2$; Table 4). The 3 base models were among the 8 highly supported models in the second candidate set. Sex was included in 6 of the 8 models, and age was included in 5. When duration of captivity was added to the base models, it resulted in small improvements in model support. In all cases, the model with duration of captivity acquired an 8–25% increase in model weights compared to the corresponding base models (e.g., evidence ratios 1.08–1.25 as per Burnham and Anderson 2002:72), indicating weak support for an effect of the duration of captivity. When we excluded data for 5 fishers that were kept in captivity for a longer period to allow for medical treatment and recovery, there was even less support for models that included duration of captivity. The addition of release date, canines, or standardized body mass to base models resulted in reductions in model support (Table 4).

Survival rates differed among release-year cohorts, seasons, sexes, and age classes of fishers (Table 5; Figure 2). Although confidence intervals overlapped slightly for year 1 estimates, survival rates tended to be highest for fishers released in cohort 1 (0.88; 0.73–1.00 [95% CI]) and lowest for those released in cohort 2 (0.57; 0.38–0.76; Figure 2A). Survival rates were lowest during the breeding season and highest during the orientation period immediately following the releases (1 Dec to 29 Feb; Figure 2B). Survival rates for males (0.79; 0.63–0.94) tended to be greater than those of females (0.62; 0.47–0.76) in year 1 (Figure 2C). Juveniles (0.78; 0.65–0.91) tended to

TABLE 3 Model results for models in candidate set 1 that examined the influence of release-year cohort, season, sex, and age (main effects) on survival of fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011. Results include Akaike's Information Criterion adjusted for small sample size (AIC_c), relative changes in AIC_c (ΔAIC_c), and Akaike's model weights (ω_i)

Model	Number of parameters	AIC_c	ΔAIC_c	ω_i
Cohort + season + sex	7	297.70	0.00	0.34
Cohort + season + sex + age	8	298.34	0.65	0.24
Cohort + season + age	7	299.03	1.33	0.17
Cohort + season	6	300.84	3.15	0.07
Cohort + season + sex + age + season $\times$ age	12	301.02	3.33	0.06
Cohort + sex	4	301.40	3.70	0.05
Cohort + age	4	302.24	4.55	0.03
Cohort	3	305.07	7.37	0.01
Cohort + age + cohort $\times$ age	7	306.13	8.43	0.00
Cohort + sex + cohort $\times$ sex	7	306.15	8.45	0.00
Season + age	5	308.99	11.30	0.00
Season + sex	5	309.26	11.56	0.00
Season + age + sex	6	309.49	11.80	0.00
Age	2	310.02	12.32	0.00
Season	4	310.22	12.52	0.00
Season + age + season $\times$ age	9	310.86	13.16	0.00
Sex	2	311.69	14.00	0.00
Null	1	313.13	15.43	0.00



TABLE 4 Model results for models in candidate set 2 to assess factors influencing the survival of fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011. We used the candidate models in set 2 to examine the influence of adding 4 covariates (duration of captivity, release date, canines, and standardized body mass), individually, to base models from candidate set 1 (indicated in parentheses) to assess improvement in model support. Results include Akaike's Information Criterion adjusted for small sample size (AIC_c), relative change in AIC_c (ΔAIC_c), and Akaike's model weights (ω_i)

Model	Number of parameters	AIC_c	ΔAIC_c	ω_i
Cohort + season + sex + duration of captivity	8	297.48	0.00	0.14
Cohort + season + sex (base model 1)	7	297.70	0.21	0.13
Cohort + season + sex + age + duration of captivity	9	297.96	0.47	0.11
Cohort + season + sex + age (base model 2)	8	298.34	0.86	0.09
Cohort + season + age + duration of captivity	8	298.58	1.10	0.08
Cohort + season + sex + release date	8	298.60	1.11	0.08
Cohort + season + age (base model 3)	7	299.03	1.55	0.06
Cohort + season + sex + age + release date	9	299.48	1.99	0.05
Cohort + season + sex + canines	8	299.49	2.01	0.05
Cohort + season + sex + body mass	8	299.70	2.21	0.05
Cohort + season + age + release date	8	300.07	2.58	0.04
Cohort + season + sex + age + body mass	9	300.34	2.85	0.03
Cohort + season + sex + age + canines	9	300.34	2.86	0.03
Cohort + season + age + canines	8	301.04	3.55	0.02
Cohort + season + age + body mass	8	301.06	3.57	0.02

have higher survival rates than adults (0.59; 0.42–0.76) in year 1 (Figure 2D). When combined across release-year cohorts, year 1 survival rates were greatest for juvenile males (0.83), followed by juvenile females (0.67), adult females (0.60) and adult males (0.33; Table 5). There was no discernable difference in survival rates of fishers between the first and second year following release for any population segment (Table 5).

We recovered carcasses of 35 of 90 fishers released and monitored from January 2008 to December 2011. We determined cause of mortality for 24 of the 35 recovered fishers (68.5%; Table 6); cause of mortality could not be determined for the remaining 11 recovered fishers because of advanced decay or extensive scavenging of the carcass (Table 6). Predation was the leading known cause of mortality, followed by vehicle strikes. Analysis of DNA confirmed that predation by bobcat and mountain lion resulted in 2 fisher mortalities. Fifty-two percent of females died during the study (26 of 50 females) accounting for 74% of 35 documented mortalities, whereas 22.5% of males died (9 of 40), accounting for 26% of the 35 mortalities (Table 6).

DISCUSSION

We monitored fisher survival to determine if survival rates were substantially lower than those reported for established resident populations (Table 7) or if they appeared too low to support a self-sustaining population. Prior to this study, survival estimates for fishers had only been reported for established native populations (Table 7). Survival rates varied among release-year cohorts, seasons, and sex and age classes of fishers released on the Olympic Peninsula resulting in disparate relationships to survival rates reported previously for established populations. The survival rates for fishers in



TABLE 5 Kaplan-Meier survival estimates (*S* with 95% CIs) by release cohort, sex and age class, and years 1 and 2 following release for fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011

Population segment at time of release	Cohort 1		Cohort 2		Cohort 3		All cohorts	
	<i>n</i>	<i>S</i> (95% CI)	<i>n</i>	<i>S</i> (95% CI)	<i>n</i>	<i>S</i> (95% CI)	<i>n</i>	<i>S</i> (95% CI)
Year 1								
Juvenile females	3	1.00 (1.00–1.00)	6	0.56 (0.04–1.00)	7	0.57 (0.21–0.94)	16	0.67 (0.42–0.92)
Adult females	9	0.89 (0.69–1.00)	14	0.43 (0.17–0.69)	11	0.62 (0.29–0.96)	34	0.60 (0.42–0.77)
All females	12	0.92 (0.76–1.00)	20	0.45 (0.22–0.68)	18	0.61 (0.36–0.85)	50	0.62 (0.47–0.76)
Juvenile males	5	1.00 (1.00–1.00)	11	0.78 (0.51–1.00)	16	0.79 (0.56–1.00)	32	0.83 (0.69–0.98)
Adult males	1	0.00 (0.00–0.00)	0		7	0.50 (0.00–1.00)	8	0.33 (0.00–0.86)
All males	6	0.83 (0.50–1.00)	11	0.78 (0.51–1.00)	23	0.77 (0.54–1.00)	40	0.79 (0.63–0.94)
Year 2								
Juvenile females	3	0.67 (0.23–1.00)	2	0.50 (0.00–1.00)	4	0.67 (0.00–1.00)	9	0.61 (0.28–0.94)
Adult females	6	1.00 (1.00–1.00)	6	0.40 (0.00–1.00)	5	0.75 (0.23–1.00)	17	0.71 (0.42–0.92)
All females	9	0.88 (0.66–1.00)	8	0.44 (0.00–0.88)	9	0.71 (0.28–1.00)	26	0.67 (0.46–0.88)
Juvenile males	4	1.00 (1.00–1.00)	7	1.00 (1.00–1.00)	9	0.75 (0.00–1.00)	20	0.90 (0.62–1.00)
Adult males	0		0		1	0.00 (0.00–0.00)	1	0.00 (0.00–0.00)
All males	4	1.00 (1.00–1.00)	7	1.00 (1.00–1.00)	10	0.63 (0.00–1.00)	21	0.84 (0.50–1.00)

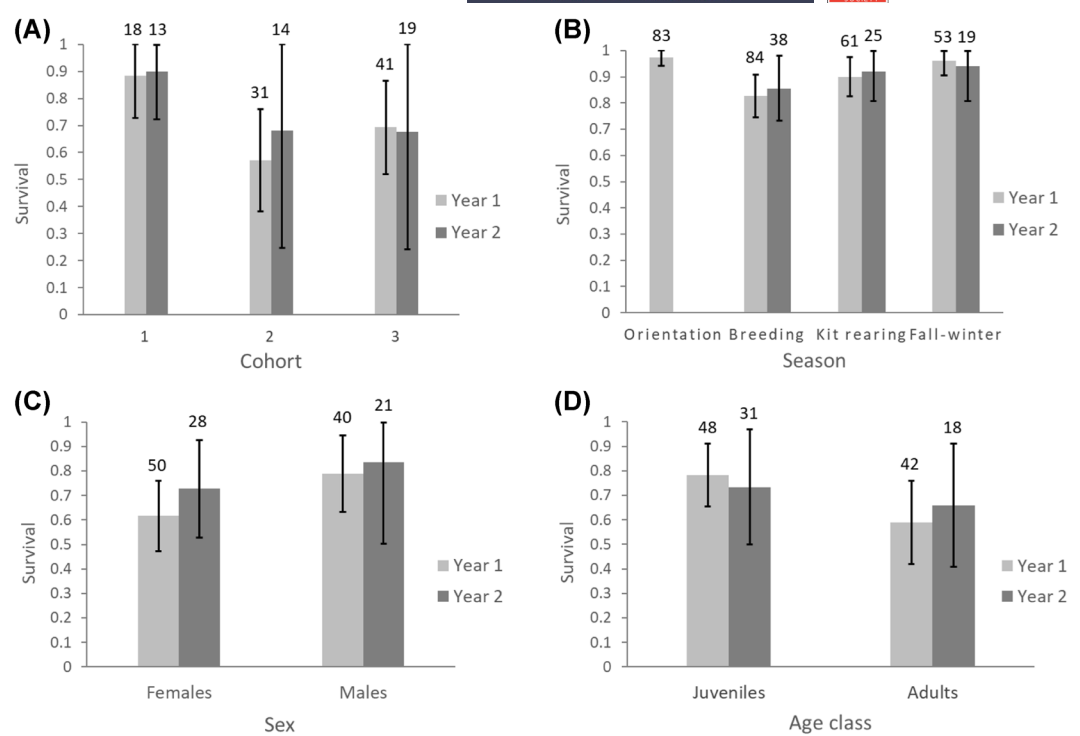


FIGURE 2 Univariate main effects ($\pm 95\%$ CI) of cohort (A), season (B), sex (C), and age class (D) on post-release survival of fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011. Sample sizes are provided above the bars for the estimates in each graphic

TABLE 6 Cause of mortality for fishers released and monitored on the Olympic Peninsula, Washington, USA, 2008–2011. Cause of mortality was determined for 24 of 35 fishers that died and were recovered

Cause of mortality	Females (n)				Males (n)				Total mortalities (%)
	Cohort 1 (12)	Cohort 2 (20)	Cohort 3 (18)	All (50)	Cohort 1 (6)	Cohort 2 (11)	Cohort 3 (23)	All (40)	
Predation	1	3	7	11	1		2	3	14 (40.0)
Unknown	1	6		7		1	3	4	11 (31.4)
Vehicle strike	1	3	1	5		2		2	7 (20.0)
Drowning		1	1	2					2 (5.7)
Trapping related ^a		1		1					1 (2.9)
Cohort totals	3	14	9	26	1	3	5	9	35 (100.0)

^aThis female died from injuries she suffered after being caught in a leg-hold trap ~14 months after release.

release-year cohort 1 were similar to the highest survival rates reported for established resident populations that were either subjected to light trapping pressure (1 fisher harvested over 4 years in a national park in MI [Belant 2007]; mean annual state-wide harvest of 130 fishers in MA, 1992–1994 [York 1996]) or no trapping pressure (CA [Jordan 2007]; Table 7). Survival rates for female fishers in release-year cohorts 2 and 3 were lower than those reported for any established resident population, whereas the survival rates for males in those cohorts were comparable to reported estimates (Table 7). Survival estimates for adult males were based on only 8 individuals (7 of the 8 were from

**TABLE 7** Estimated survival rates for established fisher populations in North America (modified from Buskirk et al. 2012)

Study	Location	Population segment	Survival interval	Survival rate (95% CI)
Sweitzer et al. (2016) ^a	CA	Males	Annual	0.62 (0.54–0.70)
		Females	Annual	0.72 (0.67–0.78)
Belant (2007) ^{b,c}	MI	All fishers	Annual	0.89 (0.50–0.99)
Jordan (2007) ^a	Sierra Nevada, CA	Males	Annual	0.88 (0.54–0.98)
		Females	Annual	0.88 (0.59–0.97)
Koen et al. (2007) ^{b,c}	ON	Males >9 months	Annual	0.33 (0.18–0.60)
		Females >9 months	Annual	0.63 (0.47–0.86)
		Adult males	Annual	0.45 (0.24–0.83)
		Adult females	Annual	0.81 (0.72–0.91)
		Males >9 months	2-year	0.15 (0.04–0.50)
		Females >9 months	2-year	0.51 (0.34–0.78)
York (1996) ^b	MA	Adult males	Annual	0.77 (0.63–0.95)
		Adult females	Annual	0.90 (0.80–1.00)
		Juvenile males	Annual	0.77 (0.46–1.00)
		Juvenile females	Annual	0.84 (0.65–1.00)
Krohn et al. (1994) ^{b,c}	ME	Adult males	Trapping season (39 days)	0.57 (0.42–0.78)
		Adult females	Trapping season (39 days)	0.79 (0.64–0.97)
		Juveniles	Trapping season (39 days)	0.38 (0.25–0.57)
		Adults	Non-trapping season	0.89 (0.81–0.99)
		Juveniles	Non-trapping season	0.72 (0.53–0.99)
Paragi et al. (1994) ^{b,c}	ME	Adult females	Annual	0.65 (0.50–0.86)
		Juveniles	Annual	0.27 (0.14–0.50)

^aFishers were not commercially trapped.^bFishers were recreationally trapped in the study area.^cFishers were translocated previously near this study area.

release-year cohort 3); consequently, there is poor precision in the survival rates estimated for adult males and the estimated survival for both male age classes combined is heavily influenced by the survival of 32 males <2 years old. We determined no clear cause for lower survival rates for cohort 2 fishers; therefore, we attribute it primarily to stochastic variation related to unknown, unmeasured variables or to small sample sizes. Despite the variable relationships of survival between the reintroduced population of fishers and other established populations, subsequent monitoring of population occupancy patterns of fishers on the Olympic Peninsula indicated that the post-release survival of fishers we observed was sufficient for the population to persist and become widely distributed throughout the Peninsula 6 years after the last releases (Happe et al. 2020).

We evaluated factors that managers could control to increase survival rates and translocation success. Of the factors we examined, variation among cohorts in first-year annual survival rates suggests the potential benefits of releasing



fishers over multiple years. There was no meaningful relationship between survival and duration of captivity, even when we included fishers that were kept in captivity for longer periods of time to recover from medical treatments. The high quality of care that fishers received while in captivity likely minimized stress while in captivity and promoted weight gain, which may have minimized negative effects of captivity on post-release survival. The lack of model support for the effect of the number of intact canines on survival may be a function of the small sample of released fishers with only 2 intact canines ($n = 8$) or that the ability to acquire prey is not diminished with a loss of 2 canines.

Survival rates were lower for females during the breeding season and for fishers in release-year cohorts 2 and 3. Females that had not established a home range before or during the first breeding season or that made atypical movements to search for mates, were likely at increased risk (Lewis 2014). The lower survival estimates for males during the breeding season were consistent with their extensive breeding-season movements and fighting among males when competing for mates (Douglas and Strickland 1987, Lewis 2014). Atypical breeding season movements by females, and the typical breeding-season movements of males, likely increased the risk of mortality, especially when those movements occurred in a novel environment (Lewis 2014).

The relative ranking of survival rates among sex and age classes of translocated fishers on the Olympic Peninsula (i.e., survival of juveniles > adults; males > females) was the reverse of that typically observed in established native populations (i.e., adults > juveniles; females > males). Translocated juveniles may be better than adults at exploiting and surviving in novel environments because their habitat selection behaviors may be more substantially influenced by the habitat qualities of the release area than the source area, whereas adults developed habitat selection behaviors that were effective for the source area but may be less effective for the release area (Stamps and Swaisgood 2007). Further, the relative lack of competing males in the reintroduced population may have increased survival, particularly for juvenile males, relative to established resident populations where territorial defense may be more pronounced. Juveniles may also have other age-specific traits (e.g., learning capacity, predator avoidance behaviors, physical condition) that enable them to better exploit novel environments than adults. Although no data are available from previous fisher translocations for comparison, research findings from translocations of the Arabian oryx (*Oryx leucoryx*; Stanley-Price 1989), the golden lion tamarin (*Leontopithecus rosalia*; Kleiman et al. 1991), and the North-Island saddleback (*Philesturnus rufusater*; Armstrong et al. 2005) indicated that juveniles and younger individuals had higher initial survival rates than adults. The smaller size of female fishers (adults and juveniles) makes them more vulnerable than males to predation by mid-sized carnivores (e.g., bobcats and coyotes; Wengert et al. 2014). Translocated adult females may also be at a greater risk to mortality because they have more specific habitat requirements, and they face greater nutritional and energetic demands associated with fetal development and kit rearing than juvenile females or males.

Predation comprised the greatest percentage of known mortalities among translocated fishers on the Olympic Peninsula, whereas predation and trapping were among the most common sources of mortality in other populations both translocated and resident (Table 8). Vehicle strikes were also an important cause of mortality for translocated fishers on the Peninsula, but this was not the case for other reintroduced populations or for established native populations (Table 8). Five of the 7 vehicle strikes occurred on the primary highway in our study area (U.S. Highway 101), which varies in width from 2 to 4 lanes and surrounds the core of the study area in proximity to large expanses of fisher habitat in Olympic National Park and Olympic National Forest. Although secondary paved roads may also present risks to fishers, as indicated by 2 females that were killed on paved roads with reduced speed limits, highways appear to be a significant factor affecting post-release survival. Rodenticide poisoning associated with illegal marijuana plantations has been identified as a substantial source of mortality for fishers in California (Gabriel et al. 2012, 2015); however, it was not (or expected to be) a source of mortality for fishers on the Olympic Peninsula.

MANAGEMENT IMPLICATIONS

The relatively low survival rates we observed for adult females indicates that a founder population biased towards females (55–65%) could compensate for stochastic occurrences of poor survival among years, and also result in greater reproduction and recruitment than a population with an even or male-biased sex ratio. The relatively high



survival rates for juvenile females indicates their value as founder individuals because they are unconstrained by the need to den and produce young the first months after translocation, yet they can breed shortly after being released and give birth to their first litter in year 2. Planning efforts that include multiple years of releases or contingencies for obtaining additional founders can provide management flexibility during a translocation in the event of poor survival or poor capture success in one or more years. To minimize the mortality rate of founders, identification of recovery areas safer from known mortality sources would benefit restoration success. Our results indicate that areas in proximity to major highways and high traffic volumes or those open to trapping may reduce survival and restoration success. Further, ancillary studies that identify predation risk zones for fishers and tradeoffs with prey availability would be very useful for identifying optimal restoration areas.

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ETHICS STATEMENT

Animal handling procedures met or exceeded the guidelines of the American Society of Mammalogists for the use of wild mammals for research (Sikes et al. 2016) and were approved by the British Columbia Ministry of Environment (permit WL07-40389). Each fisher importation from British Columbia to Washington was approved by the United States Fish and Wildlife Service. This approval required the inspection and health certification of each fisher by a licensed veterinarian, an approved capture and transport permit from the British Columbia Ministry of Environment (as noted above), and a pre-approved importation declaration form (U.S. Fish and Wildlife Service form 3-177).

CONFLICT OF INTEREST

The authors declare that there are no conflict of interest.

DATA AVAILABILITY STATEMENT

Data associated with this research has been published in the USGS ScienceBase Catalog (<https://doi.org/10.5066/P9W1P2E0>).



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