Issues and Perspectives

Conservation of Kit Foxes in the Great Basin Desert: Review and Recommendations

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Abstract

The kit fox Vulpes macrotis is a species of concern to land managers in the Great Basin Desert of North America. Once common, kit foxes have declined from historical levels. Research on kit foxes in western Utah has spanned nearly 70 y and has potential to inform management and conservation within the Central Basin and Range Ecoregion of the Great Basin Desert. We conducted a systematic literature review on the northern kit fox subspecies V. m. nevadensis. We focused on studies conducted in the Central Basin and Range Ecoregion, which represents the majority of the Great Basin Desert, and provided a comprehensive summary of their ecology and demographics for resource managers. To guide future monitoring, we also reviewed techniques used for kit fox monitoring and research, and evaluated the strengths, limitations, and advances of these techniques. We identified four key factors that deserve consideration when selecting monitoring techniques for kit foxes: estimable parameters, reliability, cost, and rate of data return. Finally, we identify four primary management recommendations. We recommend that managers (1) expand kit fox monitoring and population assessments more broadly across the Great Basin Desert. To ensure future monitoring meets the needs of resource managers, we recommend (2) the application of a structured decision-making process to identify key parameters and approaches. To better understand the factors limiting kit fox populations we recommend (3) population viability and parameter sensitivity analyses to identify drivers of population change. Finally, based on evidence that genetic diversity of kit fox populations has been maintained by undescribed patterns of gene flow, we recommend (4) a broad-scale assessment of population connectivity to identify corridors supporting metapopulation dynamics. These recommendations will facilitate proactive conservation of kit foxes and management practices to reduce future population declines.
Introduction

The kit fox *Vulpes macrotis* (Figure 1) is a nocturnal carnivore native to the lowland basins of arid range-lands in western North America from northern Mexico to southern Idaho, and southern California to western Texas (McGrew 1979; Cypher and List 2014; Figure 2). Kit foxes are among the smallest canids in North America, with a slender build and large ears, which facilitate thermoregulation (McGrew 1979; Figure 1). Kit foxes use dens year-round (McGrew 1979; Arjo et al. 2003; Cypher and List 2014) and prey primarily on smaller mammals and invertebrates (Byerly et al. 2018). The kit fox is classified globally as a species of “Least Concern” by the International Union for Conservation of Nature, but there is evidence that populations are in decline (Cypher and List 2014; Dempsey et al. 2014; Lonsinger et al. 2018b). The San Joaquin kit fox *V. m. mutica* subspecies in central California is listed as federally endangered under the U.S. Endangered Species Act (ESA 1973, as amended; 32 FR 4001, Mar 11 1967). Elsewhere, there is variation in how federal land management agencies and state wildlife agencies classify the kit fox. For example, the Bureau of Land Management (BLM) classifies the kit fox as a “Sensitive Species” in Oregon (BLM 2019), Idaho (BLM 2014), and Utah (BLM 2018), but not in other states (e.g., Nevada; BLM 2017). Similarly, the Department of Defense classifies the kit fox as a “Species of Concern” in some areas (e.g., DPG 2016) but not others. Kit foxes are classified as “Critically Imperiled” in Colorado, Oregon, and Texas; “Imperiled” in Idaho; “Vulnerable” in Utah, Nevada, and Arizona; and “Apparently Secure” in New Mexico (NatureServe 2020; Figure 2). Where kit foxes are considered secure, there are insufficient data to effectively evaluate population status and data-deficient mammals are more often at risk of local extinction than not (Bland et al. 2014).

Among kit fox subspecies, the northernmost subspecies *V. m. nevadensis* was historically one of the most widely distributed in the United States (Figure 2) and was distributed primarily in the Central Basin and Range and Northern Basin and Range ecoregions (Figure 3; McGrew 1979; EPA 2013). Kit foxes in the Colorado Plateau ecoregion were also classified as *V. m. nevadensis* by McGrew (1979), but International Union for Conservation of Nature (IUCN) range maps suggested this ecoregion may represent a northern extent of *V. m. neomexicana* (Figure 2; IUCN 2014). The Central Basin and Range and Northern Basin and Range ecoregions comprise the Great Basin Desert, which is characterized by low-lying basins bounded by north–south running mountain ranges. These basins and range formations resulted from extension of the earth’s crust along parallel faults during the Cenozoic period (Thompson and Burke 1974). The Great Basin Desert is collectively characterized as semiarid, with arid and mesic basins dominated by salt desert shrub (dominated by Chenopodiaceae) and sagebrush steppe (dominated by Artemisia spp.) communities, respectively (Pellant et al. 2004). Ranges in the Great Basin Desert are dominated by pinyon (Pinus spp.) and juniper (Juniperus spp.) woodlands (Pellant et al. 2004). The introduction of nonnative grasses (e.g., cheatgrass Bromus tectorum) and increased frequency and extent of wildfires have modified these systems (Pellant et al. 2004).

Figure 1. Kit fox *Vulpes macrotis nevadensis* in 2014 in western Utah, USA. Photo by Robert N. Knight.
Figure 2. Historical range of kit foxes *Vulpes macrotis* based on McGrew (1979) and the IUCN (2014), including subspecies designations from McGrew (1979). Subspecies include (1) *V. m. arsipus*, (2) *V. m. devia*, (3) *V. m. macrotis* (extinct), (4) *V. m. mutica* (endangered), (5) *V. m. neomexicana*, (6) *V. m. nevadensis*, (7) *V. m. tenuirostris*, and (8) *V. m. zinseri*. Note that there is disagreement in the subspecies designation for kit foxes in eastern Utah, which were classified as *V. m. nevadensis* based on McGrew (1979), but were presumably *V. m. neomexicana* based on the IUCN (2014).
Kit foxes were believed to have been one of the most abundant carnivores in the Great Basin Desert (Egoscue 1956, 1962, 1975; Arjo et al. 2007). From 1951 to 1970, research conducted in western Utah generated seminal information on the natural history of kit foxes (Egoscue 1956, 1962, 1975). Recent research in the same region suggested kit foxes have declined in abundance and distribution and may now be rare (Arjo et al. 2007; Dempsey et al. 2014; Lonsinger et al. 2017; Byerly et al. 2018).

Here, we conducted a systematic literature review of *V. m. nevadensis* within the Central Basin and Range ecoregions (hereafter, Central Basin). The Central Basin represents the core (and most contiguous) portion of the *V. m. nevadensis* distribution. The kit fox populations in western Utah represent the only populations within the *V. m. nevadensis* range for which there is extensive research, providing both historical and contemporary population assessments. We believe it is reasonable to extend information from patterns of kit fox populations in western Utah to other portions of the subspecies range within the same ecoregion, but that extrapolating to different ecoregions (e.g., Northern Basin and Range and Colorado Plateau) or subspecies would be imprudent. We also conducted a literature review on survey techniques for kit foxes. We performed an initial search on Thomson Reuters Web of Science from 1969–2020 for "*Vulpes macrotis*" and either "nevadensis," "Great Basin," "Utah," "Nevada," "Colorado," "Oregon," or "Idaho." To recover historical articles not covered by the Web of Science, we performed the same search on Google Scholar.

**Figure 3.** Boundaries of the Central Basin and Range and Northern Basin and Range ecoregions in western North America, which represent the Great Basin Desert and agreed upon portion of the northern kit fox subspecies *Vulpes macrotis nevadensis* range, and predicted habitat of *V. m. nevadensis* (USGS 2018). Within the states of Nevada and Utah, points represent documented historical and contemporary kit fox locations (Table S1, Supplemental Material), including in eastern Utah, which represents the Colorado Plateau ecoregions and for which subspecies designation is uncertain (see Figure 1). Counties with records but without point locations represent counties with known harvest since 2002. Dugway Proving Ground represents the focal area of the most extensive kit fox research in the Great Basin Desert.
Scholar from 1900 to 1968. We subsequently excluded articles that did not include *V. m. nevadensis* or were outside of the Central Basin. We also excluded theses or dissertations, when the work was also available in a peer-reviewed publication, and book sections that compiled published literature. We also performed a search on Web of Science (1969–2020) for articles including “*Vulpes macrotis*” and keywords of “monitoring,” “survey,” or “detection,” which was not restricted to the Great Basin Desert because we expected methods to be transferable among regions and subspecies. We included additional related literature that was found incidentally. Finally, we contacted state wildlife management agencies and federal land management agencies within the Central Basin to request technical reports (i.e., gray literature) related to kit fox monitoring or management.

Conservation of sensitive species requires reliable and accurate information on population parameters (e.g., abundance, distribution, vital rates, and trends) and community associations (e.g., habitat use and interspecific interactions). Despite these needs, the status of many sensitive species remains ambiguous due to 1) vagueness in the identification of key parameters for assessing status and 2) difficulty in developing effective and cost-efficient monitoring programs. Failure to transform monitoring results into actionable, proactive strategies may further limit effective conservation. Our goal was to compile literature on kit foxes in the Central Basin and on kit fox survey techniques into a succinct and actionable synopsis to 1) guide monitoring and management decisions, 2) aid managers in stemming kit fox declines, and 3) decrease the likelihood of the subspecies being listed under the ESA. To this end, we reviewed the current state of knowledge on kit foxes in the Central Basin, discussed the effectiveness of alternative monitoring strategies, and provided recommendations and research needs for conserving kit foxes.

**Current State of Knowledge**

**Distribution and habitat associations**

Kit foxes have been largely undersampled and consequently the distribution of extant populations is poorly defined in the Central Basin, the Great Basin Desert, and range wide. Large-scale maps of predicted habitat suggested that the environmental conditions across much of the Central Basin were suitable for kit foxes (USGS 2018), and documented detections suggested kit foxes have remained widely distributed across their historical range in the ecoregion (Figure 3). A substantial amount of monitoring and research has been conducted on one kit fox population in the Central Basin, which occurs on the U.S. Dugway Proving Ground and surrounding lands in western Utah (collectively hereafter, Dugway). Otherwise, only limited formal surveys have taken place in other portions of the Central Basin (O’Neal et al. 1987; Richards 2017; Nevada Department of Wildlife, unpublished data).

Local distributions of wildlife populations are increasingly being characterized by occupancy, which represents the proportion of units occupied by a species. When estimated based on a random sample of units, occupancy represents that probability that a randomly selected unit is occupied (MacKenzie et al. 2018). Richards (2017) surveyed five study areas in western Utah for kit foxes (2015–2016) and detected kit foxes within each area. Among these areas, Dugway had the highest occupancy (0.07; Richards 2017). The estimates of occupancy for kit foxes reported by Richards (2017) at Dugway contrasted with occupancy estimates from 2013 to 2014 (0.28–0.49; Lonsinger et al. 2017). Estimates of occupancy depend upon the scale of the unit, which is influenced by sampling methodology and determined as part of the study design (MacKenzie et al. 2018). Generally, as the scale of the unit increases, so does occupancy (toward 1). The scale of the unit used by Lonsinger et al. (2017) was larger than that used by Richards (2017). Consequently, the resulting occupancy estimates are not directly comparable, but this suggests that occupancy estimates may have been higher across areas surveyed by Richards (2017) if the scale of inquiry were increased to align with Lonsinger et al. (2017).

Across their range, kit foxes have historically been associated with relatively flat terrain supporting shrubland habitats or mixed grassland–shrubland (Egosue 1962; McGrew 1979). In the Great Basin Desert, kit foxes were historically found in low-lying basins in shrublands dominated by greasewood *Sarcobatus vermiculatus* (Figure 4a), sagebrush *Artemisia* spp. (Figure 4b), shadscale *Atriplex confertifolia*, four-winged saltbush *Atriplex canescens*, and rabbitbrush *Chrysothamnus* spp. (Egosue 1962; McGrew 1979; O’Neal et al. 1987). Kit foxes tended to avoid woodlands (e.g., pinyon–juniper woodlands) that characterized higher elevations. The introduction of cheatgrass and altered fire regimes have decreased shrubland habitats (Pellant et al. 2004). For example, over ~10 y in western Utah the proportion of grasslands increased while shrublands decreased (Byerly et al. 2018). Where they have been studied, kit foxes have persisted in these primarily grassland habitats (Figure 4c), but perhaps at reduced densities (see Density, Home Range Size and Movements).

Recent findings in the Central Basin indicated kit foxes continued to select low-lying areas (elevations <1,600 m) that were relatively flat (Dempsey et al. 2015; Richards 2017; Table 1). Although some kit foxes have been documented denning in more rugged terrain with greater slopes, these “mountain” foxes moved to less rugged terrain to forage (Kozlowski et al. 2008). In contrast to historical patterns, though, evidence from the Central Basin suggested that kit foxes may be avoiding (or decreasing use of) shrublands (Kozlowski et al. 2012; Lonsinger et al. 2017). Grasslands made up the largest proportion of kit fox home ranges aligning with these conclusions (Kluever and Gese 2017; Table 1). Kit foxes often selected for vegetation heights <1 m tall.
The influence of vegetation height on kit fox habitat use was likely influenced (or confounded) by factors such as land-cover type, density of vegetation, prey availability, and predators (see Interspecific Interactions).

Kit foxes use dens year-round to alleviate thermal stress (behavioral thermoregulation), reduce predation risk, and rear young (Golightly and Ohmart 1983; Arjo et al. 2003). In addition to a natal den, kit foxes maintain auxiliary dens throughout their home range, providing spatially disseminated refugia from predation and alternative dens to mitigate ectoparasite loads (Egoscue 1962; O’Neal et al. 1987; Arjo et al. 2003; Kluveer and Gese 2017; Kluveer et al. 2019a). Kit foxes may dig their own dens, exploit burrows of other species (e.g., badgers *Taxidea taxus* and kangaroo rats *Dipodomys* spp.), or use anthropogenic cavities (e.g., culverts; Arjo et al. 2003; O’Neal et al. 1987). Kit foxes are obligate denners, so they may require specific soil substrates to construct dens. In western Utah, silty soil was the majority soil type within kit fox home ranges (Kluveer and Gese 2017), kit foxes were more likely to colonize sites with silty soil (Lonsinger et al. 2017), and kit foxes avoided areas with gravelly or fine-sand soils (Dempsey et al. 2015; Richards 2017; Table 1). These patterns of soil selection likely relate to ease of excavation and drainage (Egoscue 1956; O’Neal et al. 1987; Arjo et al. 2003) and are not unique to kit foxes in the Central Basin (e.g., Rodrick and Mathews 1999). Historical research suggested that kit foxes commonly denned in shrublands (Egoscue 1962; O’Neal et al. 1987), but recent evidence has suggested they denned primarily in grasslands (Arjo et al. 2003). This shift may have been in response to decreasing shrublands and increasing grasslands (Byerly et al. 2018), increased coyote abundance and activity in shrublands, or both (Arjo et al. 2007; Lonsinger et al. 2017, 2018b).

Water is commonly a limiting resource in desert systems. Kit foxes have evolved behavioral (e.g., nocturnal activity and year-round denning), physical (e.g., large ears and long-slender legs; Figure 1), and physiological (e.g., efficient water economy and high thermal conductance) adaptations allowing them to persist in arid environments (McGrew 1979; Golightly and Ohmart 1983, 1984). Although kit foxes use free-standing water when available (Hall et al. 2013a), they can persist on preformed and metabolic water acquired through prey (Golightly and Ohmart 1984), and they do not appear limited by water availability (Table 1). For example, kit fox detections were similar in areas with and without free-standing water (Hall et al. 2013b), water availability did not influence kit fox occurrence or spatial dynamics (Lonsinger et al. 2017), and the removal of water did not influence kit fox survival or relative abundance (Kluveer and Gese 2017). While drought years have been shown to negatively impact kit fox populations by limiting prey (White and Garrott 1999) and may lead to water stress in lactating females (Randel 2016), annual precipitation in the Great Basin Desert has more than doubled since the mid-1900s and climate projections suggested additional increases in annual precipitation with fewer drought years over the next 60–80 y (Wagner 2003).

**Interspecific interactions**

Primary prey of kit foxes included leporids, small mammals (often kangaroo rats), and insects, but they also consumed birds, reptiles, arachnids, and carrion (e.g., ungulates) to a lesser extent (Egoscue 1956; Kozlowski et al. 2008, 2012; Byerly et al. 2018). Egoscue (1962) identified leporids as the most important prey in western Utah, but more contemporary investigations suggested leporids were less important than kangaroo rats, other rodents, and insects (Kozlowski et al. 2008; Byerly et al. 2018). Dietary changes could be due to declining leporid abundance, increased abundance of

![Figure 4.](image-url)
Table 1. Land-cover and landscape factors influencing kit fox Vulpes macrotis habitat use from 1999–2016; in the Great Basin Desert of western Utah, USA.

| Variable          | Effect | Pattern                        | Sampling method       | Analysis approach      | Source |
|-------------------|--------|--------------------------------|-----------------------|------------------------|--------|
| Elevation         | (–)    | Use highest at elevations <1,600 m | Radiotelemetry        | Resource selection      | 3      |
|                   | (–)    | Occupancy ≥0.2 when elevation <2,000 m | Camera-trapping       | Resource selection      | 6      |
| Slope             | (–)    | Use highest at the lowest slopes | Radiotelemetry        | Resource selection      | 3      |
|                   | (–)    | Occupancy ≥0.2 only when slope <15° | Camera-trapping       | Resource selection      | 6      |
| Shrubland         | (–)    | Avoidance of shrubland habitats | Radiotelemetry        | Distribution modeling   | 1      |
|                   | (–)    | Occupancy ~0 when proportion of shrubland or woodland >40% | Noninvasive genetic sampling | Occupancy modeling      | 5      |
|                   | (–)    | Home ranges consisted of more grassland than water sites | Radiotelemetry        | Compositional analysis  | 4      |
| Silty soils       | (+)    | Selection for silty soils; Avoidance of gravelly or fine-sand soils | Radiotelemetry        | Resource selection      | 3      |
|                   | (+)    | Selection for silty soils | Camera-trapping       | Resource selection      | 6      |
|                   | (+)    | Silty soils: majority soil type in home range | Radiotelemetry        | Compositional analysis  | 4      |
|                   | (+)    | Colonization promoted by silty soils | Noninvasive genetic sampling | Occupancy modeling      | 5      |
| Free water        | (None) | Detection similar in areas with and without water | Camera-trapping       | Proportion tests        | 2      |
|                   | (None) | No influence of water on occurrence | Noninvasive genetic sampling | Occupancy modeling      | 5      |
|                   | (None) | Water removal did not influence survival | Radiotelemetry        | Known-fate modeling     | 4      |

* Sources include (1) Kozlowski et al. 2012, (2) Hall et al. 2013a, (3) Dempsey et al. 2015, (4) Kluever and Gese 2017, (5) Lonsinger et al. 2017, (6) Richards 2017. Direction of effect is either negative (–), positive (+), or no effect (None).

coyotes (for which leporids were the most important prey), or both (Arjo et al. 2007; Kozlowski et al. 2008; Byerly et al. 2018). In the Central Basin, Egoscue (1975) suggested kit fox reproductive rates were driven by leporid abundance and that alternative prey may be insufficient to meet nutritional demands for reproduction. Similarly, White and Garrott (1997) found that decreases in leporids were expected to drive decreases in kit fox reproductive rates and density, and an increase in home range size. The increased importance of kangaroo rats may have also been related to the expansion of invasive grasslands because intermediate levels of cheatgrass supported higher densities of kangaroo rats than some native grasslands and mixed shrubland communities (Kluever et al. 2019b). Generally, recent studies in the Central Basin suggesting rodents were the most important prey of kit foxes aligned with dietary patterns for kit foxes in California (Nelson et al. 2007; Kelly et al. 2019), Arizona (Zoellick et al. 1989) and Mexico (Moehrensclger et al. 2007).

Kit foxes have been killed by a variety of mammalian (e.g., red foxes V. vulpes, badgers, and bobcats Lynx rufus) and avian (e.g., golden eagles Aquila chrysaetos and owls) predators, but coyotes have been the main source of mortality for kit foxes (Arjo et al. 2007; Kozlowski et al. 2008; Kluever and Gese 2017; see Population Dynamics). Although kit foxes and coyotes likely co-occurred naturally in southern portions of their historical ranges (e.g., Mexico and Texas), coyotes were not native to the Great Basin Desert and colonized following European settlement (McGrew 1979; Gompper 2002). Following colonization, coyotes and kit foxes in the Central Basin had significant dietary overlap, which likely related to maximizing the intake of preformed water (via mammalian prey; Kozlowski et al. 2008; Byerly et al. 2018). Coyotes and kit foxes also demonstrated high overlap in temporal activity patterns, but coyotes were slightly more crepuscular (Kozlowski et al. 2008), a pattern consistent with the activity of leporids (Arias-Del Razo et al. 2011).

Land cover in the Central Basin influenced the spatial heterogeneity in prey diversity and abundance (Hall 2012; Kozlowski et al. 2012; Kluever et al. 2016; Smith et al. 2017; Byerly et al. 2018; Kluever et al. 2019b), as well as the potential for interactions between kit foxes and coyotes (Kozlowski et al. 2012; Lonsinger et al. 2017). A comparison of observed versus predicted distributions found that kit foxes avoided prey-rich habitats that they historically used (e.g., greasewood shrublands), suggesting kit foxes may have selected habitats to minimize conflict with coyotes (i.e., safety-matching; Kozlowski et al. 2012). Occupancy models also found a negative relationship between kit fox occupancy and shrubland and woodland cover types; these cover types were positively associated with coyote occurrence (Lonsinger et al. 2017). Similar patterns of kit foxes avoiding coyotes in prey-rich shrublands has been detected in other portions of their range (e.g., New Mexico; Robinson et al. 2014).

Although kit foxes in the Central Basin selected habitats that minimized risk of intraguild predation (Kozlowski et al. 2012), coyotes were widespread and kit foxes often traveled long distances to forage (Kozlowski et al. 2008) or foraged in areas with high levels of coyote activity (Lonsinger et al. 2017), placing them at heightened risk of negative interactions. Lonsinger et al. (2017) found that coyotes were the primary factor influencing local extinction of kit foxes, with kit
Foxes being more likely to be displaced from areas with greater coyote activity. Counterintuitively, kit foxes were more likely to colonize areas with greater coyote activity, likely because these were prey-rich areas (Kozlowski et al. 2012; Lonsinger et al. 2017). These patterns of colonization and extinction in the Central Basin produced kit fox occupancy rates that were lower than occupancy predicted in the absence of coyotes (0.65 [Table 2]; Lonsinger et al. 2018a), suggesting patterns of mesopredator suppression (Soulé et al. 1988).

**Density, home range size, and movements**

In the Central Basin, only the kit fox population around Dugway has been studied intensely enough to estimate density. Kit foxes were reportedly common at Dugway and believed to be the most abundant carnivore in the region through the mid-1900s (Egoscue 1956, 1962; Arjo et al. 2003). These patterns of colonization and extinction in the Central Basin produced kit fox occupancy rates that were lower than occupancy predicted in the absence of coyotes (0.65 [Table 2]; Lonsinger et al. 2017), suggesting patterns of mesopredator suppression (Soulé et al. 1988).

Kit fox densities may change rapidly in response to environmental stochasticity (e.g., precipitation), which is common in deserts (White and Garrott 1999). For example, a closely monitored San Joaquin kit fox population in California fluctuated from 20 to 170 kit foxes/100 km² with large interannual changes (White and Garrott 1999). Historical kit fox density estimates from Arizona and Texas (22–39 kit foxes/100 km²; McLaughlin 1979; Zoellick and Smith 1992) were comparable to historical estimates from Dugway (Table 2). Collectively, these estimates suggested recent kit fox densities in the Central Basin were at least an order of magnitude lower than other reported estimates (Lonsinger et al. 2018b).

Home range sizes of kit foxes vary among and within habitats and are likely influenced by prey availability (Moehrensclager et al. 2007). Home range sizes of kit foxes reported at Dugway (Table 2) were significantly larger than estimates from California (2.5–11.6 km²; List and Cypher 2004), but were comparable to estimates from the Northern Basin and Range ecoregion (20.1–37.4 km²; Eckrich et al. 2018). Arjo et al. (2003) estimated mean annual home range sizes at Dugway from 11.5 ± 2.9 km² to 43.1 ± 20.8 km². Mean home range size of kit foxes varied among seasons, with the smallest and largest home ranges during the pup-rearing (Kluever and Gese 2017: 20.9 km²; McLaughlin 1979; Zoellick and Smith 1992) and breeding (Kluever and Gese 2017: 15.9 ± 11.3 km²; Dempsey et al. 2014: 17.2 ± 9.4 km²) seasons, respectively. Historical home range sizes were not available for Dugway, but estimates from an area ∼130 km south of Dugway were smaller (3.0–3.4 km²; O’Neal et al. 1987) and comparable to estimates in California. It is

### Table 2. Point estimates of kit fox *Vulpes macrotis* population parameters in the Great Basin Desert of western Utah, USA, by sampling period.

| State variables          | 1951–1969 | 1955–1958 | 1966–1969 | 1996 | 1997–2001 | 2009–2012 | 2010–2013 | 2013–2014 | 2015–2016 |
|--------------------------|-----------|-----------|-----------|------|-----------|-----------|-----------|-----------|-----------|
| Density (foxes/100 km²)  | —         | 15–22     | 10–21     | 14   | 2–6       | —         | —         | 2         | —         |
| Home range size (km²)    | —         | —         | —         | —    | —         | —         | —         | —         | —         |
| Annual                   | —         | —         | —         | —    | —         | —         | —         | —         | —         |
| Pup-rearing              | —         | —         | —         | —    | —         | —         | —         | —         | —         |
| Breeding                 | —         | —         | —         | —    | —         | —         | —         | —         | —         |
| Occupancy rate           | —         | —         | —         | —    | —         | —         | —         | 0.28–0.49 | 0.14–0.74 |
| Sex ratios (M:F)         | —         | 1.2:1     | 1.2:1     | —    | —         | 1.7:1     | 1.7:1     | —         | —         |
| Vital rates              | —         | —         | —         | —    | —         | —         | —         | —         | —         |
| Annual adult survival    | —         | —         | —         | 0.65 | 0.52–0.56 | —         | —         | —         | —         |
| Coyote-caused mortality  | —         | —         | —         | 0.67 | 0.19      | —         | —         | —         | —         |
| Reproduction (pups/female)| —         | 0.75–4.2  | 1.4–2.72  | 2    | 1.0–3.8   | —         | —         | —         | —         |
| Immigration              | 0.08      | —         | —         | —    | —         | —         | —         | —         | 0.03–0.07 |

**Genetic metrics**

| Allelic richness         | 5.9       | —         | —         | —    | —         | —         | —         | —         | 6.0       |
| Expected heterozygosity  | 0.62      | —         | —         | —    | —         | —         | —         | —         | 0.67      |
| Inbreeding coefficient   | −0.012    | —         | —         | —    | —         | —         | —         | —         | −0.017    |

**Sources**

1. Egoscue 1962, 2. Egoscue 1975, 3. White and Garrott 1997, 4. Arjo et al. 2003, 5. Arjo et al. 2007, 6. Kozlowski et al. 2008, 7. Kozlowski et al. 2012, 8. Dempsey et al. 2014, 9. Dempsey et al. 2015, 10. Kluever and Gese 2017, 11. Lonsinger et al. 2017, 12. Richards 2017, 13. Lonsinger et al. 2018a, 14. Lonsinger et al. 2018b, 15. Lonsinger et al. 2019.
unclear whether the larger home range sizes at Dugway represented an increase from smaller historical home ranges, perhaps due to behavioral changes to avoid increasing intraguild interactions (i.e., the competition refuges hypothesis; Kamler et al. 2012), secure sufficient prey under increasingly homogenized habitats, or both (Dempsey et al. 2014; Lonsinger et al. 2017; Byerly et al. 2018). Alternatively, the larger home ranges observed at Dugway may be influenced by large areas of resource-poor salt playa, which were adjacent to core kit fox habitats.

Kit fox movements were limited during diurnal periods to around the den, with primary movements occurring at night (Egoscue 1956). In western Utah, kit foxes historically traveled only short distances (<1 km) from their dens to forage (Egoscue 1956; O’Neal et al. 1987). Kozlowski et al. (2008) suggested that increased kit fox home range sizes required longer excursions from dens to optimal (or better) foraging areas. Long nightly movements are not uncommon for kit foxes. For example, average nightly movements ranging from 5.2 to 14.3 km have been documented in the Chihuahuan and Sonoran deserts (Zoellick et al. 1989; Moehrensclager et al. 2007).

Population dynamics

For kit foxes in the Central Basin, annual survival probability (hereafter, survival) has been the most studied vital rate (Table 2). Estimates of adult survival ranged from 0.56 to 0.85 (White and Garrott 1997; Arjo et al. 2007; Kluever and Gese 2017), with earlier estimates having large variances due to smaller sample sizes. Unfortunately, estimates of survival were not available from the earliest kit fox investigations in the Central Basin (Egoscue 1956, 1962). Recent estimates suggested adult survival has declined in the Central Basin (Table 2), although lower estimates have been recorded for endangered San Joaquin kit foxes in California (Cypher and Scrivner 1992). Survival of juvenile kit foxes in the Central Basin has only been estimated once and averaged 0.28 over a 4-y span (Kluever and Gese 2017).

Historically, kit fox mortalities caused by coyotes were considered rare in the Central Basin (Egoscue 1962; O’Neal et al. 1987). More recently, coyotes have routinely been identified as the predominant source of kit fox mortality in the Central Basin (Table 2), with up to 19–67% of kit fox mortalities being caused by coyotes across studies (White and Garrott 1997; Arjo et al. 2007; Kluever and Gese 2017). These levels were generally lower than coyote-caused mortality rates for San Joaquin kit foxes (i.e., 50–87%; White and Garrott 1997). Anthropogenic causes of kit fox mortality in the Central Basin appear to be minimal—Kluever and Gese (2017) recorded only 1 of 50 (2%) kit fox deaths being attributed to human factors (vehicle collision). Though it is legal to trap kit foxes throughout much of the Great Basin Desert, the low value of kit fox hides may contribute to relatively low interest from trappers. For example, from 2006 to 2016, the mean number of kit foxes harvested annually in Utah was 109 individuals (range = 39–315), which was lower than mean annual harvest levels for other canid species (range = 1,208–6,787; Bernales and McFarlane 2016). In Nevada, kit foxes have been harvested from every county from 2002 to 2019, with a statewide annual mean of 383 (range = 76–890) harvested kit foxes (Nevada Department of Wildlife, unpublished data). Golden eagle predation of kit foxes has been reported only once outside of the Great Basin Desert (Cypher and Scrivner 1992), but recently represented >5% of kit fox mortality in western Utah (Kluever and Gese 2017). Golden eagle predation has been reported as the most common source of mortality for swift foxes V. velox (Moehrensclager et al. 2007) and island foxes Urocyon littoralis (Roemer et al. 2002). Kit foxes in western Utah may be foraging longer and more often during crepuscular periods to compensate for limited resources (Kozlowski et al. 2012) or changes in spatial availability of prey (Byerly et al. 2018). More crepuscular activity and land cover changes may be facilitating golden eagle predation of kit foxes because kit foxes may be detected and captured more easily in open grasslands than shrublands.

Kit fox reproductive rate (number of pups produced per adult female per year) has been estimated on several occasions in the Central Basin and ranged from 0.75 to 4.5 (White and Garrott 1997; Arjo et al. 2007; Table 2). The most recent estimate of reproductive rate for kit foxes (1.0 pup/female) was the second lowest estimate in the Central Basin (Arjo et al. 2007) and was comparable to estimates in California (White and Garrott 1997; Randel 2016). Changes in prey abundance may have contributed to variation in kit fox reproductive rates (White and Garrott 1997, 1999; Egoscue 1975; see Interspecific Interactions).

Beyond survival and reproduction, population dynamics are influenced by dispersal among metapopulations. Our knowledge of dispersal tendencies in the Central Basin is limited to research conducted in western Utah, where kit foxes tended to initiate natal dispersal from August to November of their first year (O’Neal et al. 1987). Natal dispersal distances of 20–64 km have been documented in western Utah (Egoscue 1956; O’Neal et al. 1987; Arjo et al. 2007). Interestingly, O’Neal et al. (1987) documented regular use (and crossing) of mountainous terrain during dispersal. Data collected on San Joaquin kit foxes suggested a male-biased pattern of dispersal, though average (7.8 km) and maximum (32.2 km) dispersal distances in California (Koopman et al. 2000) were typically lower than those reported in western Utah. Dispersal places kit foxes at high risk of mortality (O’Neal et al. 1987), and therefore longer dispersal distances in the Central Basin may have implications for effective dispersal (i.e., dispersal resulting in successful reproduction). Still, recent research at Dugway produced genetic evidence of immigration,
indicating the population was connected (at least minimally) by effective dispersal from other populations (Lonsinger et al. 2018a; see Population Genetics).

**Population genetics**

The genetic characteristics of kit fox populations in the Central Basin have received minimal study. The decline in kit fox density at Dugway (see Density, Home Range Size, and Movements) was expected to negatively influence effective population size ($N_e$) and genetic diversity, which could have implications for the adaptive potential and population viability, and these concerns motivated a recent genetic investigation. Comparison of genetic data (i.e., 8–9 nuclear DNA microsatellite loci) collected from kit foxes at Dugway from 2013 to 2014 with historical specimens collected prior to 1970 suggested kit fox $N_e$ had declined ~85% to only 71 (Lonsinger et al. 2018a; Table 2). This recent estimate of $N_e$ suggested the population was approaching levels at which inbreeding depression may have been a concern and the risk of local extinction was heightened (Franklin 1980; Frankham et al. 2014). Still, there was no evidence of declining genetic diversity (i.e., allelic richness and heterozygosity) or inbreeding (i.e., $F_{IS}$ or fixation index; Table 2). Genetic evidence of immigration (though at levels lower than historical levels) likely mitigated the negative consequences typically associated with small $N_e$ (Lonsinger et al. 2018a).

Genetic research has occurred in only three other kit fox populations. Sacks and Milburn (2018) investigated genetic diversity and $N_e$ of kit foxes in the Northern Basin and Range ecoregion of Great Basin Desert (Oregon) and the Mojave–Colorado Desert (California); they also collected six samples from Nevada, but did not include these in estimates of genetic diversity or $N_e$. The genetic diversity of kit foxes in Oregon was lower than genetic diversity in California (Sacks and Milburn 2018), and both appeared to have lower genetic diversity than recent levels of genetic diversity in western Utah (Lonsinger et al. 2018a). The estimate of $N_e$ in Oregon (14.7) was the lowest reported (Sacks and Milburn 2018). Estimates of $N_e$ from the Mojave–Colorado Desert (38.0) and western Utah (54.7–101.9) were not significantly different from one another (Lonsinger et al. 2018a; Sacks and Milburn 2018), nor were they different from the estimate of $N_e$ for a population of San Joaquin kit foxes (40.1; Wilbert et al. 2015). We recommend caution when comparing estimates of genetic diversity and $N_e$ across studies, though, because each study used different genetic markers and thresholds for the inclusion of rare alleles.

**Monitoring Strategies**

**Live-capture–recapture and radiotelemetry**

Live-capture–recapture was first employed as a tool to study kit foxes during the mid-twentieth century. Seminal research using live-capture–recapture increased our understanding of kit fox physiology, den use, food habits, home range size and density, demography, and parasitology (Egoscue 1956, 1962, 1975). Trap types commonly used to capture kit foxes included #7 Havahart (Egoscue 1956, 1962, 1975) and double-door box traps (Arjo et al. 2003; Dempsey et al. 2014). Occasionally, kit foxes exhibited extreme trap shyness at den sites. When this occurred, a box trap enclosure could be utilized to increase trapping efficiency (Kozlowski et al. 2003). Caution is warranted when considering diminutive foot-hold traps for capturing kit foxes on account of greater likelihood of both injury and nontarget captures. The relatively low density of kit foxes in the Central Basin had largely limited the use of live-capture–recapture as a robust and efficient monitoring approach, or for estimation of population state variables (e.g., density). For example, Dempsey et al. (2014) found both scat-deposition and scent-station surveys were more effective at both detecting kit foxes and measuring relative abundance than live capture, which yielded low capture rates (<2%) for kit foxes.

Standard radiotelemetry, where locations of animals were obtained by homing or triangulation, has been employed on several occasions for kit foxes in the Central Basin from 1983–2013 (O’Neal et al. 1987; Arjo et al. 2003; Kozlowski et al. 2008; Kluver and Gese 2017). Radiotelemetry has been pivotal for obtaining vital rate estimates such as survival and evaluating habitat selection of kit foxes at multiple scales (e.g., the geographical range of a population and individual home range scales). Fine-scale (e.g., within home range) habitat selection and movements (i.e., step selection functions) have not been investigated for kit foxes in the Central Basin.

Global positioning system (GPS) collars adequately sized for kit foxes are available and could facilitate studies of fine-scale habitat selection and movement. The small body size of kit foxes restricts current technology to use of GPS collars that store location data on the collar, rather than transmitting it via satellite to the researcher. Although these collars may be equipped with a remote download, this requires the animal to be located and the receiver to be within a short distance and line of sight to the transmitter. To date, the weight limitation on these GPS collars have limited battery size and consequently signal strength. Detecting GPS-collared kit foxes has been challenging because of the weak transmitter signal and kit fox denning behaviors, which has made data retrieval more difficult (Eckrich et al. 2018). For example, Eckrich et al. (2018) recently deployed GPS collars to investigate habitat selection by kit foxes in Oregon, but had low success locating collared animals and recovering data via the remote download option as a result of weak signal strengths; they instead relied on recapturing animals to recover data stored on the collars. Still, leveraging advanced tracking technology for kit foxes has the potential to increase our understanding of dispersal, within home range movements, and antipredator behavior, but also introduces additional logistical and financial challenges.
Scat-deposition surveys

Scat-deposition surveys along two-track and gravel roads provided the highest detection probability for kit foxes when compared with scent (track) stations, spotlighting, and trapping (Dempsey et al. 2014). Scat-deposition surveys produced an index of abundance that best reflected known abundances, making it a potentially useful method for evaluating trends in relative abundance or assessing population response to habitat manipulations (Kluever and Gese 2017). Scat-based indices of relative abundance assume (1) scat detection is similar among survey routes and (2) over time, and (3) that scats are identified correctly. When used to compare relative abundances among species (e.g., kit foxes and coyotes), scat-based relative abundance indices also assume (4) scat detection is similar among species. Spatial (e.g., width and substrate) and temporal (e.g., usage and vehicle traffic) variation in road characteristics can lead to heterogeneity in scat persistence and detection among surveys (Kluever et al. 2015; Lonsinger et al. 2016), limiting the usefulness of scat surveys for long-term population monitoring. In western Utah, scats of kit foxes and sympatric carnivores were commonly misidentified with field-based methods (i.e., based on size, shape, odor, etc.), with misidentifications being biased toward kit foxes (i.e., kit foxes detections were biased high; Lonsinger et al. 2015b). Scat identification can be improved with statistical classification approaches, genetic analyses, or both (Lonsinger et al. 2015b). A significant contribution of scat surveys has included dietary composition for kit foxes and dietary overlap with sympatric carnivores (Kozlowski et al. 2008, 2012; Byerly et al. 2018). As with relative abundance indices, dietary metrics may be biased if scats are misidentified. Byerly et al. (2018) found that bias in diet analyses associated with scat misidentification between kit foxes and coyotes decreased with increasing sample size, but this was likely related to the high dietary overlap between the two species. Dempsey et al. (2015) found that road-based scat-deposition surveys were not appropriate for evaluating resource selection and species distribution, but alternative scat sampling methods that more randomly sample the landscape (e.g., scat detection dogs) could offer a viable scat-based approach to modeling species distributions (Smith et al. 2003).

Scent-station surveys

Scent-station surveys historically involved combining a scent lure (or bait) with track plates or similar track-registering substrates (e.g., sifted dirt or sand), and determining species visitation through track identification (Schauster et al. 2002). Harrison et al. (2001) and Schauster et al. (2002) evaluated the effectiveness of scent-station surveys for swift foxes, whereas Warrick and Harris (2001) and Dempsey et al. (2014) evaluated scent-station surveys for kit foxes. These studies resulted in similar conclusions; scent-station surveys provided reliable detection of kit foxes, but detection rates were lower than those from scat-deposition surveys. Although scent-station surveys require minimal supplies and upfront costs, they require frequent (e.g., daily) visits to document track registers (and reset stations by clearing or removing tracks), which increases field work expenses (e.g., personnel and travel). Scent-station surveys may also be subject to environmental disturbances (e.g., precipitation and wind) that reduce their effectiveness. Scent-station surveys may suffer from misidentification of tracks, particularly when there are multiple sympatric canids (Dempsey et al. 2014). Warrick and Harris (2001) cautioned that scent-station surveys were imprecise and that as an index of abundance, they may only be sufficient for detecting large changes in abundance.

Camera-based surveys

Remote cameras are relatively inexpensive and easy to use for monitoring wildlife populations. Although cameras have known potential setbacks (e.g., mechanical failure, inability to reliably distinguish individuals, misidentification, and small area of detection), this technology has become more durable and reliable. Photos from cameras can provide nearly instantaneous information (subject only to review time of photos) and a source of data for nontarget species that can be archived for future analyses. Machine learning algorithms show potential to automate identification of species, number of individuals, and behaviors from photos, thereby drastically increasing efficiency of camera-based sampling (Norouz-zadeh et al. 2018; Tabak et al. 2019).

Scent lures are commonly used to increase the detection of low-density carnivores by cameras (Rocha et al. 2016; Mills et al. 2019). Lures may not always influence detection positively, particularly for subordinate species or prey species, and this may reduce the effectiveness of lures when using cameras for multispecies or community-level monitoring (Meek et al. 2015; Mills et al. 2019). In a recent study of swift foxes, anecdotal evidence suggested that cameras with a scent lure failed to detect swift foxes that were known to occur in the area when coyotes visited the camera first (J. Jenks, personal communication), presumably because of heightened risk of intraguild interactions. Furthermore, when evaluating fine-scale patterns of habitat use, lures may coerce animals to use less favorable areas or draw animals out of favorable habitats.

Remote cameras have been deployed to target kit foxes in the Great Basin (i.e., western Utah and Oregon), Mojave (Utah, Nevada, and Arizona), and Chihuahuan (i.e., New Mexico and Mexico) deserts, and have provided data on kit fox distribution (Milburn and Hiller 2013), occupancy (Richards 2017), habitat selection (Hall et al. 2013b), reproductive rates (Kluever et al. 2013), interspecific interactions (Robinson et al. 2014), and behavior (e.g., activity patterns, vigilance and investigative behaviors; Hall et al. 2013a; Sergeyev et al. 2020). To date, camera-based monitoring of kit foxes has relied on the application of lures or attractants (e.g., Hall et al. 2013b;
Richards (2017), natural aggregation points (e.g., water sources; Hall et al. 2013a), or known den sites (Kluever et al. 2013). Richards (2017) evaluated the influence of different lures and broadcasting mechanisms (i.e., scented predator survey disks, cotton swabs, and hollowed golf balls) on kit fox detections. Among treatments there was no difference in detection probability of kit foxes, but Richards (2017) did not compare lure treatments with a control without a lure.

Cameras may be better suited to some needs than others. For example, Kluever et al. (2013) found cameras were more reliable than visual observations (e.g., via spotting scope; Cypher et al. 2000) in determining accurate counts of adult and juvenile kit foxes at known dens. Similarly, cameras have proven to be an effective tool in investigating individual behavior of kit foxes, including vigilance (antipredator behavior) at water sources (Hall et al. 2013a) and variation in investigatory tendencies between sites with and without harvest (Sergeyev et al. 2020). Although cameras have been effectively used to estimate kit fox occupancy (Richards 2017) and patterns of co-occurrence with coyotes (Robinson et al. 2014), low detection rates may limit its applicability in some systems. For instance, following initial detection, Hall et al. (2013a) documented too few return visits to cameras with lures to successfully fit two-species occupancy models.

Noninvasive genetic sampling
Noninvasive genetic sampling (NGS) for kit foxes has focused primarily on extending scat-deposition surveys via fecal DNA and has the potential to generate estimates for a wide range of parameters that are important to kit fox conservation. Species identification of scats with mitochondrial DNA has reduced misidentifications (Lonsinger et al. 2015b), informed correction factors for relative abundance (Lonsinger et al. 2016), and improved diet analyses (Byerly et al. 2018). Molecular detection of prey within scats can also improve inferences on diets (Gosselin et al. 2017; Monterroso et al. 2019). Unambiguous molecular species identification permitted scat data to be used in occupancy analyses to evaluate processes influencing detection, occupancy, colonization and local extinction of kit foxes (Lonsinger et al. 2017).

Individual identification of scats with nuclear DNA microsatellite loci has allowed for estimates of kit fox abundance and density (Wilbert et al. 2015; Sacks and Milburn 2018; Lonsinger et al. 2019). When analyzed in a robust design (i.e., multiseason) framework, NGS-based capture-recapture may provide estimates of apparent survival and recruitment, as well as information on immigration and emigration (Lonsinger et al. 2019). Understanding patterns of DNA degradation and rates of sample accumulation can be used to optimize NGS, as was done in western Utah, reducing costs for single- or multispecies monitoring (Lonsinger et al. 2015a).

Additionally, NGS can provide information on sex ratios, population genetic parameters, and movement. Scat-based NGS has been used to quantify genetic diversity and effective population size, assess population genetic structure, and evaluate risk of inbreeding depression (Lonsinger et al. 2018a; Sacks and Milburn 2018). The relatively large sample sizes often generated with NGS can provide information on movement tendencies through genetic assessments of mean dispersal distances (e.g., via spatial autocorrelation; Banks and Peakall 2012) and immigration (Lonsinger et al. 2018b). Noninvasive genetic sampling, however, does not provide information on temporal activity, movement paths, resource selection, or home range characteristics (e.g., location of dens).

Spotlight surveys
Spotlight surveys were tested for monitoring foxes but have shown only minimal success. Warrick and Harris (2001) found that kit fox detections during spotlight surveys were positively related to population estimates, but (like scent-station surveys) were too imprecise to be effective for population monitoring. Schauster et al. (2002) reported spotlight surveys were not as effective as scat-deposition surveys or scent-station surveys for swift foxes. Spotlight surveys failed to detect kit foxes in Utah, even in areas where they were known to occur (Dempsey et al. 2014). Spotlight surveys are labor intensive, but this may be offset if detections of kit foxes are supplemental to other target species (e.g., jackrabbits Lepus spp.) for which spotlight surveys are effective (Dempsey et al. 2014; Kluever et al. 2017). Although data collected through spotlight surveys could be analyzed in a distance sampling framework to estimate density, kit fox detections with this technique have been insufficient for such analyses.

Monitoring Trade-offs
Monitoring strategies and parameters of interest
The appropriate monitoring strategy varies by management need and parameters of interest (Table 3). For example, listing decisions under the federal ESA (and inversely, delisting criteria) are informed by estimates of abundance or density. Consequently, if abundance was identified as the key parameter of interest, only NGS has been effective at estimating abundance of kit foxes (Lonsinger et al. 2018b; Lonsinger et al. 2019). Alternatively, home range estimates from telemetered kit foxes may be extrapolated to provide density estimates and serve as a proxy for abundance (Arjo et al. 2007). When considering trade-offs between alternative monitoring approaches, managers may want to consider value-added metrics that can be acquired with different monitoring strategies (Table 3). In this case, radiotelemetry offers improved understanding of vital rates driving population abundance including direct estimates of survival, cause-specific mortality, and locations of dens
for monitoring reproduction (Arjo et al. 2007; Kluever et al. 2013; Kluever and Gese 2017). In contrast, scat-based NGS provides only estimates of apparent survival, but allows for estimates of population genetic parameters (e.g., genetic diversity, effective population size; Lonsinger et al. 2018a) and dietary tendencies (Gosselin et al. 2017; Byerly et al. 2018).

For territorial species, occupancy may be a suitable proxy for abundance (MacKenzie and Nichols 2004). Both camera-based and NGS-based sampling can provide reliable estimates of kit fox occupancy (Lonsinger et al. 2017; Richards 2017). Camera-based approaches may be favorable for estimating occupancy if managers are interested in the additional behavioral aspects of kit fox ecology (e.g., temporal activity patterns; Hall et al. 2013a) or community-level monitoring. In contrast, NGS-based approaches may be favorable if managers are interested in joint estimation of occupancy, abundance, and genetic parameters.

### Reliability of parameter estimates

Evaluating reliability (i.e., accuracy and precision) of parameter estimates is of critical importance in selecting appropriate long-term monitoring strategies. Effective management requires enough precision to detect meaningful changes in the underlying population (with respect to the parameter of interest). Importantly, reliability of parameter estimates is influenced by the monitoring technique, sampling design, and assumptions of the analytical approaches for generating estimates (Williams et al. 2002; MacKenzie et al. 2018).

Pilot studies can provide data necessary to evaluate the reliability of parameter estimates under competing monitoring methods or sampling designs. For example, Lonsinger et al. (2019) used data collected on kit foxes in western Utah to evaluate NGS-based abundance estimation under competing sampling designs (i.e., single-occasion sampling and multioccasion sampling) and analytical approaches (i.e., capture with replacement, nonspatial robust design, and multisession spatial capture-recapture). Single-occasion methods for estimating abundance from NGS may be appealing to managers because they have reduced effort and costs relative to more traditional capture-recapture methods requiring multiple “capture” events. Indeed, these approaches have been used to estimate abundance of kit foxes in Oregon (Sacks and Milburn 2018) and California (Wilbert et al. 2015). Lonsinger et al. (2019) found that efforts to reduce costs through single-occasion sampling resulted in estimates that were biased low with artificially high precision, indicating this approach may not be appropriate for low-density carnivores such as kit foxes. Noninvasive genetic sampling–based estimates of abundance from multioccasion sampling achieved a coefficient of variation of <10% with four to five sampling occasions, but this level of effort was still insufficient to detect seasonal reproductive pulses (Lonsinger et al. 2018b, 2019).

### Cost comparisons of monitoring strategies

Costs can vary substantially among monitoring strategies and may influence applicability for long-term or large-scale monitoring of kit foxes. Costs may include field personnel, equipment and supplies, data streaming fees (e.g., for GPS collars or cellular cameras), laboratory fees, or data processing and analyses. Cost comparisons among monitoring strategies can be difficult because previous research often integrated multiple objectives,
Table 4. Estimated annual cost, data collection approaches, and focal parameters for two concurrent field efforts that included kit foxes Vulpes macrotis and were performed during 2012–2014 in the Great Basin Desert of western Utah, USA.

| Data collection procedures | Frequency | Estimable parameters |
|---------------------------|-----------|----------------------|
| **Alternative 1**<sup>a</sup> | **Kit foxes and coyotes** | **Kit foxes and coyotes** | **Kit foxes and coyotes** |
| Total annual cost:<sup>b</sup> | $172,291 | 1. Scat deposition surveys | 1. Relative abundance |
| Total area surveyed: | 1,127 km<sup>2</sup> | 2. Scent post surveys | • Home range |
| | | 3. Live-capture | • Survival |
| | | 4. Radiotelemetry | • Cause-specific mortality |
| | | 5. Den observations | • Reproduction |
| Prey | | 1. 2× annually | • Habitat selection |
| | | 2. 3× annually | • Water use |
| | | 1. Small mammal trapping | • Relative abundance |
| | | 2. Lagomorph spotlight surveys | • Habitat influences |
| **Alternative 2**<sup>c</sup> | **Kit foxes and coyotes** | **Kit foxes and coyotes** | **Kit foxes and coyotes** |
| Total annual cost:<sup>b</sup> | $144,652 | 1. Noninvasive genetic sampling via scat surveys | Abundance |
| Total area surveyed: | 3,663 km<sup>2</sup> | 2. Genetic laboratory procedures (DNA extraction, amplification, scoring) | Density |
| | | | Occupancy |
| | | | Colonization |
| | | | Extinction |
| | | | Habitat selection |
| | | | Genetic diversity |
| | | | Effective population size |
| | | | Dietary composition |

<sup>a</sup> Conducted by EMG and BMK.
<sup>b</sup> Costs include actual costs for supplies, equipment, and procedures. Personnel wages varied between alternatives and were standardized at U.S. $12/h for comparative purposes.
<sup>c</sup> Conducted by LPW and RCL.

parameters, and methods simultaneously, some of which may be of greater or lesser importance to managers for long-term monitoring or risk assessment. Furthermore, cost comparisons can be challenging as a result of real or perceived savings resulting from multispecies monitoring efforts and differences in how expenses are incurred (e.g., upfront equipment costs vs. laboratory fees; Lonsinger et al. 2015a; Mumma et al. 2016; Kauth et al. 2020).

For example, we compared annual costs associated with two alternative monitoring strategies performed concurrently during 2012–2014 (Table 4). We acknowledge these alternatives are not exhaustive and do not include evaluation of the reliability of estimates, but they do represent actual concurrent monitoring efforts and, therefore, allowed comparison for illustrative purposes. Coauthors EMG and BMK conducted Alternative 1, in which they included monitoring of kit foxes, coyotes, and prey species, and produced estimates of vital rates (i.e., survival and reproduction), habitat use patterns, and prey associations (Table 4). Coauthors LPW and RCL conducted Alternative 2, in which they included noninvasive genetic sampling of kit foxes and coyotes, and produced estimates of state variables (i.e., abundance and occupancy), habitat use patterns, genetic measures of diversity and effective population size, and dietary composition (Table 4). We standardized all field and laboratory personnel to U.S. $12/h to generate costs for comparison. Estimated annual costs for Alternative 1 was $27,639 higher than Alternative 2, but Alternative 1 included direct sampling and investigation of prey communities, which was not included in Alternative 2 (Table 4). Costs for Alternative 1 were largely related to field data collection. In contrast, <25% of costs for Alternative 2 related to field data collection, with most costs being related to genetic laboratory procedures. The relative cost of each alternative must consider the estimable parameters (Tables 3, 4) and management objectives. Similar comparisons could be made with additional monitoring strategies, including camera-based monitoring, which is increasing in popularity among wildlife practitioners.

**Rate of data return**

Beyond financial costs, rate of data return (time required to collect, process, and analyze data to produce actionable parameter estimates) may be an important consideration for monitoring programs. Rate of data return may be particularly important when managers need to evaluate an acute population response to management actions or disturbances. Assuming multiple monitoring strategies can provide reliable estimates of the parameter of interest with sufficient precision to detect meaningful changes, managers may opt for (potentially more expensive) methods with a more expeditious rate of data return.

As with cost comparisons, rate of data return comparisons can be difficult. Estimates of abundance and vital rates (e.g., survival, reproduction) often require extended sampling efforts, which inherently have a slow rate of data return. In contrast, estimates of occupancy can often be assessed more rapidly and provide a surrogate for abundance (MacKenzie and Nichols 2004). To this end, we briefly compare two sampling approach-
es used to estimate occupancy of kit foxes: 1) camera-trapping and 2) scat-based NGS. For comparison, we assume that sampling units are equal in size and approximate the mean home range size of a kit fox. We also assume that computational times are equitable, with variation in rate of data return being a consequence of disparities in data collection and processing. We focus our comparison on factors influencing rate of data return, including advances in technology or protocols that may further improve rate of data return in the future.

Camera-trapping and scat-based NGS both provide relatively rapid forms of field data collection. Camera-trapping typically requires a minimum of two visits to each camera site (i.e., to set and subsequently check the camera) and the duration between these events (i.e., the sampling occasion) will depend on detection rates of the target species, which may be influenced by the use of lures or attractants (Rocha et al. 2016; Mills et al. 2019). Kit fox detection rates were too low after a 2-wk sampling occasion to produce occupancy estimates (Hall et al. 2013a), but detection rates were sufficient to produce occupancy estimates with 3 wk of sampling (Richards 2017). Additionally, the spatial extent of camera-trapping is limited by the availability of cameras. Rotation of cameras among sites can facilitate sampling at a larger scale, but this naturally increases the duration of field data collection. In contrast, scat-based NGS requires only a single visit to each site for sampling when spatial replication is employed and is not limited by equipment constraints (Lonsinger et al. 2017).

Data processing is a substantial time investment for both camera-trapping and scat-based NGS. Camera-trapping can produce thousands to millions of photographs and processing of these photos can be time-consuming (O’Connell et al. 2011). Machine learning algorithms may greatly reduce this burden, but are most reliable for large distinguishable species and are not able to confidently discriminate among sympatric canids (Norouzzadeh et al. 2018; Tabak et al. 2019). Misclassification rates for carnivores from machine learning are sufficiently high to bias occupancy estimates (Royle and Link 2006; Norouzzadeh et al. 2018; Tabak et al. 2019). Scat-based NGS can produce thousands to tens of thousands of samples and DNA extraction, amplification, and scoring procedures can be time consuming. Scat identification based on appearance or morphometrics (i.e., statistical classification) can expedite rate of data return, but produce misclassification rates for kit foxes that are sufficiently high to bias occupancy estimates (Royle and Link 2006; Lonsinger et al. 2015b). Still, machine learning (for camera data) and statistical classification (for NGS data) may offer tools to focus sample processing on those samples with the greatest uncertainties in identification, accelerating the rate of data return. Machine learning and statistical classification approaches may introduce some misidentifications, but this may be addressed through the application of occupancy models that explicitly account for false positives (Royle and Link 2006).

Conclusions and Recommendations

Our review highlights several knowledge gaps and research needs. Based on the current state of knowledge, we make four primary management recommendations: 1) an expansion of kit fox monitoring and population assessments to other portions of the Central Basin, 2) the application of a structured decision-making process to identify key parameters and approaches for monitoring, 3) population viability and parameter sensitivity analyses to identify drivers of population change, and 4) modeling and assessment of population connectivity to identify areas for land management and conservation to maintain metapopulation dynamics.

Although there has been a considerable amount of research on kit foxes in western Utah, which has provided foundational knowledge, identified potential conservation challenges, and offered insights into population resiliencies, there is a paucity of data from other portions of the kit fox range in the Central Basin. Notably, outside of western Utah the distribution of extant kit fox populations is poorly described, no reliable estimates of population demographic parameters (e.g., survival and reproduction) exist, and population trends are unknown. Despite kit fox population declines documented in western Utah, immigration has allowed the population to retain high levels of genetic diversity (Lonsinger et al. 2018a). Dispersal of kit foxes in the Central Basin has not been adequately studied, but kit foxes have the capacity for long-distance dispersal (O’Neal et al. 1987; Koopman et al. 2000) and metapopulation dynamics may mitigate the negative consequences and risks associated with declining populations.

The putative factors driving kit fox population declines in western Utah (i.e., invasive grasslands, declining prey, and increasing coyotes) likely influence kit foxes across the Great Basin Desert. Despite efforts, effective mitigation of these threats has not been identified and will require a better understanding of the interplay among these factors. For example, earlier research postulated that increasing coyote populations were driven by anthropogenic water developments (Arjo et al. 2007), suggesting that habitat management (i.e., reduction of water) might limit coyotes; more contemporary research refuted this hypothesis (Hall et al. 2013b; Kluever and Gese 2016; Lonsinger et al. 2017). Recent findings that kit foxes were negatively associated with shrublands and coyotes were positively associated with shrublands (Kozlowski et al. 2012) could lead to the misguided interpretation that increasing grasslands would decrease coyotes and benefit kit foxes. In contrast, there was evidence that kit foxes often traveled long distances to forage in shrubland habitats and were more active in areas with greater coyote activity (likely due to fine-scale resource matching; Lonsinger et al. 2017). Further complicating these relationships, Lonsinger et al. (2017) found that kit fox occupancy would not be expected to
increase with reductions in coyote activity, unless coyote activity could be eliminated. Consequently, expansions in kit fox monitoring would benefit from incorporating spatial and temporal sampling designs that facilitate continued research on factors driving population change.

To date, managers have facilitated an impressive amount of research on the kit fox population in western Utah, but it is impractical for managers to maintain monitoring at this same intensity. Decisions about monitoring programs require identification of key parameters (i.e., those most likely to adequately characterize population risk and trajectory) and are commonly driven by desires to 1) minimize cost and effort, 2) maximize accuracy of parameter estimates, and 3) maximize ability to detect meaningful changes in the population of interest. Structured decision-making allows managers to implement an analytical framework to inform complex decisions through formal comparisons of monitoring and management alternatives, while accounting for uncertainty that could ultimately influence the decision processes (Conroy et al. 2008; Robinson et al. 2016). Structured decision-making procedures can vary, but generally includes identification of objectives, alternative actions, models to quantify response to actions, and selection of a preferred method (see Runge 2011 and Brignon et al. 2019 for descriptions of structured decision-making and case studies). Decisions from structured decision-making tend to be more robust, transparent, and defensible than those made based on informal processes (Runge 2011; Brignon et al. 2019). To this end, we recommend land managers engage a team of subject-matter experts and stakeholders to employ structured decision-making and ensure monitoring activities address programmatic needs.

Population viability analysis (PVA) is a modeling framework that uses estimates of population counts or demographic parameters to explore factors driving population dynamics, evaluate the influence of management actions on population dynamics, and assess population risk (Beissinger and Westphal 1998; Ellner et al. 2002). Population viability analyses have been used to inform population and habitat management for endangered San Joaquin kit fox populations (Dennis and Otten 2000; Haight et al. 2002). Count-based PVAs suggested San Joaquin kit fox abundances were best explained by density dependence and precipitation, and incorporating an index of coyote abundance did not improve model fit; the probability of the population dropping below an $N_r = 50$ was ≤52% within 20 y (Dennis and Otten 2000). Haight et al. (2002) extended PVAs to evaluate the influence of habitat protection strategies on probability of population extinction, but this required detailed demographic data on kit fox populations. An expansion of kit fox monitoring across the Central Basin can provide information necessary to initiate count-based PVAs and better understand risk of population declines. In western Utah, research has provided estimates of demographic parameters for kit foxes that can be used to parameterize demographic PVAs and model the influence of management alternatives on population dynamics and risk.

Sensitivity analyses of PVAs can provide valuable insights into which population parameters most strongly influence population dynamics (McCarthy et al. 1995), an important consideration when trying to disentangle the impacts of changing habitats, prey abundances, coyote abundances.

Among kit fox population parameters investigated, dispersal and connectivity (i.e., metapopulation dynamics) are the most poorly understood aspects, but may be critical to long-term population persistence and maintenance of genetic diversity (Frankham 1996, 2005; Lonsinger et al. 2018a). Understanding how landscape features facilitate or impede gene flow among metapopulations and identifying corridors for targeted conservation may help sustain kit fox populations. Noninvasive genetic sampling and landscape genetic techniques can be combined to efficiently assess patterns of connectivity and barriers (i.e., landscape and habitat features) that restrict gene flow. Ultimately, landscape genetic modeling can inform land management decisions and direct limited funding to optimal areas for maintaining and improving critical linkages for kit foxes.

**Supplemental Material**

Please note: The *Journal of Fish and Wildlife Management* is not responsible for the content or functionality of any supplemental material. Queries should be directed to the corresponding author for the article.

**Table S1.** Historical (1915–1970) and contemporary (1971–2020) kit fox *Vulpes macrotis* locations obtained from Global Biodiversity Information Facility, iNaturalist, Utah Division of Wildlife Resources, Nevada Department of Wildlife, and Sacks and Milburn (2018).

Found at DOI: https://doi.org/10.3996/JFWM-20-025.S1 (1.15 MB TXT); also available at https://openprairie.sdstate.edu/nrm_datasets/1/

**Reference S1.** Bernales HH, McFarlane L. 2016. Utah furrowy annual report. Publication No. 16–17 Annual Performance Report for Federal Aid Project W-65-M, segment 64. Salt Lake City: Utah Division of Wildlife Resources.

Found at DOI: https://doi.org/10.3996/JFWM-20-025.S2 (1.6 MB PDF)

**Reference S2.** [BLM] U.S. Bureau of Land Management. 2017. BLM Nevada sensitive and status species list. Found at DOI: https://doi.org/10.3996/JFWM-20-025.S3 (607 KB PDF).

**Reference S3.** Cypher B, List R. 2014. *Vulpes macrotis*. The IUCN Red List of threatened species 2014. e.T41587A62259374. Found at DOI: https://doi.org/10.3996/JFWM-20-025.S4 (3 MB PDF); also available at http://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T41587A62259374.en.

**Reference S4.** [DPG] Dugway Proving Ground. 2016.
Integrated natural resources management plan. Dugway, Utah: U.S. Army Directorate of Environmental Programs. Found at DOI: https://doi.org/10.3996/JFWM-20-025.S5 (4.59 MB PDF).

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