A Review of the Potential Impacts of Cannabis Cultivation on Fish and Wildlife Resources

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1. Introduction

"A Review of the Potential Impacts of Cannabis Cultivation on Fish and Wildlife Resources" provides a synthesis of the available scientific literature on potential impacts of cannabis cultivation on fish, wildlife, and associated ecosystems. As defined by the California Department of Food and Agriculture, cannabis (marijuana) cultivation refers to "…any activity involving the planting, growing, harvesting, drying, curing, grading, or trimming of cannabis." The review focuses on outdoor cultivation of cannabis, including greenhouse cultivation.

"The combination of limited water resources, a water-hungry crop, and cultivation in sensitive ecosystems means that marijuana cultivation can have environmental impacts that are disproportionately large given the area under production" (Carah et al. 2015).

2. Pollutants

Cannabis cultivation sites often use substantial quantities of pesticides, including insecticides and rodenticides, to discourage wildlife foraging on cannabis plants and to decrease damage to irrigation lines (NDIC 2007).

2.1 Pesticides (insecticides, herbicides, fungicides)

This section will focus on the effects of pesticides including insecticides, herbicides, and fungicides; effects of rodenticides are addressed in section 2.2.

2.1.1 Direct Effects

The direct effects of pesticides on wildlife include acute poisoning, immunotoxicity, endocrine disruption, reproductive failure, altered morphology and growth rates, and changes in behavior.

Wildlife can be poisoned by pesticides after exposure to a toxic dose through ingestion, inhalation, or dermal contact (Pimentel 2005, Berny 2007). In addition to killing arthropod pests, insecticides are toxic to native insect pollinators, other beneficial arthropods (e.g., spiders, predatory mites, etc.), and beneficial decomposers such as earthworms, fungi, bacteria, and protozoa (Pimentel 2005). Herbicides have also been shown to cause mortality in beneficial arthropods (Freemark and Boutin 1995).

Pesticide poisoning has also been documented in numerous vertebrae taxa, primarily birds (see Appendix A; Nettles 1976, Henny et al. 1987, Littrell and Hunter 1988, Augspurger et al. 1996, Mineau et al. 1999, Fleischli et al. 2004, Pimentel 2005). For example, granivorous birds can die after eating seeds coated in insecticides (Fairbrother 1996, Mineau and Palmer 2013). Mineau and Whiteside (2013) suggested that pesticide use was the most important indicator of grassland bird declines in the

U.S. when they found that the best predictors of such declines were lethal pesticide risk and insecticide use, *not* agriculture intensification. Population declines have also been demonstrated in herbivorous birds due to changes in plant species abundance and composition as a result of herbicide use (Sotherton et al. 1988). Pesticide use has also been shown to decrease species diversity, including loss of sensitive passerine and raptor species (Clark et al. 1986, Smutz 1987, Warner 1994). Additionally, pesticides can cause embryotoxicity to eggs of waterfowl (Hoffman and Albers 1984). Other taxa including fish (Pimentel 2005), amphibians (Relyea and Diecks 2008, Egea-Serrano et al. 2012, Brühl et al. 2013), and reptiles (Mingo et al. 2016) also have documented casualties of pesticide poisoning. Furthermore, pesticide toxicity is increased when combined with environmental stressors (e.g., predators), as has been demonstrated in amphibians (Relyea 2003).

The immune system of wildlife species can be compromised by chronic exposure to low doses of pesticides (Li and Kawada 2006, Zabrodskiĭ et al. 2012). Exposure to pesticides can lower the immune function of anurans leaving them susceptible to death from parasitic infections and pathogens (Christin et al. 2003, Rohr et al. 2008). For example, wood frogs (*Lithobates sylvaticus*) exposed to pesticides are more susceptible to trematode infections (Kiesecker 2002). Additionally, pesticide exposure may decrease an animal's ability to recover from physical injuries (Zabrodskii et al. 2002).

Disruption of the endocrine system is another common consequence of pesticide exposure. In birds, such exposure can cause alterations in the thyroid gland that negatively impacts thyroid homeostasis and thus metabolism (Pandey and Mohanty 2015). Aquatic mammals experience endocrine disruption when pesticides used in cultivation run-off into aquatic systems (Ross 2000). Pesticide run-off can also be problematic for other aquatic species. Male frogs exposed to pesticides have lower testosterone levels which can result in hermaphroditic changes (Hayes 2013). Frogs also experience inhibited growth of the larynx (Hayes 2013), which likely has consequences for mating success if they are unable to participate in mating calls. In fish, pesticides can inhibit important hormones causing delays in growth (Baldwin et al. 2009).

Exposure to pesticides may also result in reproductive failure for many wildlife species. In birds, exposure has been shown to reduce egg production leading to reduced clutch sizes (Grue et al. 1997, Pimentel 2005, Berny 2007). Pesticides can also cause reduced litter sizes in mammals (Grue et al. 1997, Pimentel 2005), and mammalian fertility may also be compromised when pesticides alter ovarian development and function (Tiemann 2008). Similarly, pesticides can lead to chemical castration in frogs (Hayes 2013).

Another direct effect of pesticides on wildlife is their ability to alter morphology and growth rates of certain species; these effects have primarily been documented in amphibians (Relyea 2012). For example, pesticides have been shown to cause limb deformities in wood frogs (Kiesecker 2002). They also can result in a reduction in growth and development leading to death in leopard frogs (*Lithobates pipiens*), a species of special concern in California (Relyea and Diecks 2008).

Pesticides can also cause alterations in wildlife behavior. Arthropods exhibit altered search and attack behaviors after exposure to pesticides (Pimentel 2005). In mammals, pesticides have been shown to decrease coordination and motor skills and slow response rates to noise (Wolansky and Harrill 2008). Exposure has resulted in decreased foraging time in birds (Fairbrother 1996) and change in diet of small mammals (Johnson 1964, Fagerstone et al. 1977). Pesticides also decrease the ability of birds and mammals to thermoregulate (Grue et al. 1997). When fish are exposed to pesticide run-off, they develop swimming abnormalities making them more susceptible to predation (Renick et al. 2015).

2.1.2 Indirect Effects

Pesticides can indirectly impact wildlife through reduction of food resources and refuges, starvation due to decreased prey availability, hypothermia, and secondary poisoning.

Pesticides can decrease habitat availability for wildlife through the elimination of food resources (e.g., plants) as well as refuge sites when plant abundance and diversity is decreased (Pimentel 2005). Small mammals experience decreased survival as a result of diet shifts, greater foraging dispersal, and limited availability of cover (Keith et al. 1959, Tietjen et al. 1967, Johnson and Hansen 1969, Hull, Jr. 1971, Spencer and Barrett 1980). Southern red-backed vole (*Myodes gapperi*) abundance, for example, decreases as primary food sources are reduced and cover is eliminated by herbicides (D’Anieri et al. 1987). Reduced shrub cover from herbicides has also been shown to decrease species diversity of small mammals (Lillywhite 1977). Moreover, small mammals that experience diet shifts have been shown to have lower reproductive success (Spencer and Barrett 1980). Diet shifts and increased foraging dispersal resulting from herbicide use have also been implicated in decreased chick survival of ground-feeding gamebirds (Green 1984, Rands 1986, Warner 1994).

When prey availability is decreased from pesticide use (e.g., arthropod reductions from insecticide exposure), it may contribute to starvation of wildlife species. For example, reduced insect prey populations such as mosquitoes and beetles have been linked to declines in insectivorous bird populations, as insects are vital to birds during the breeding season (Hallmann et al. 2014). Starvation as a result of pesticide use has also been demonstrated in fish (Pimentel 2005), game birds (Pimentel 2005, Berny 2007), and mammals (Grue et al. 1997).

Sublethal levels of pesticide exposure can cause short-term hypothermia in birds and mammals (Grue et al. 1991, Gordon 1994). Mallard ducklings (*Anas platyrhynchos*) exposed to low levels of the insecticide carbofuran experienced hypothermia and increased mortality at temperatures as high as 10º C (50º F) (Martin and Solomon 1991). In mammals, the LD50 (dose at which 50% of test subjects died) dose of pesticides was significantly reduced when temperatures were both higher and lower than average; this suggests that animals were not effectively thermoregulating when exposed to pesticides (Ahdaya et al. 1976).

Lastly, secondary poisoning either through groundwater contamination and run-off or by feeding on exposed animals is a common consequence of pesticide use (Pimentel 2005). There are numerous examples of secondary poisoning of predators and scavengers that fed on incapacitated or dead animals. Gamebirds that fed on insects (that fed on plants treated with herbicides) had decreased chick survival (Berny 2007), and laughing gull (*Leucophaeus atricilla*) adults and chicks have experienced secondary poisoning from insecticides (White et al. 1979). Raptors are also common victims of secondary poisonings. Mendelssohn and Paz (1977) reported a mass mortality of raptors that fed on poisoned voles and birds. Mortality due to secondary poisoning has also been documented in red-shouldered hawks (*Buteo lineatus*; Balcomb 1983), barn owls (*Tyto alba*; Hill and Mendenhall 1980), and bald eagles (*Haliaeetus leucocephalus*; Elliott et al. 1996).

2.2 Rodenticides

Anticoagulant rodenticides (ARs) are toxic pesticides used to decrease the impacts of herbivores (primarily small mammals) on cannabis plants (NDIC 2007). They work by inhibiting blood from clotting and coagulating, ultimately leading to death (Gabriel et al. 2015).

2.2.1 Direct Effects

The direct effects of AR exposure on wildlife are acute poisoning and immunotoxicity. AR use has resulted in the poisoning of numerous non-target species (Eason and Spurr 1995, Erickson and Urban 2004, Brakes and Smith 2005). A likely reason for this is that many manufactures of ARs use “flavorizers” to make them more palatable, including sugar, bacon, cheese, peanut butter, and apple, which makes them attractive to a variety of species (Gabriel et al. 2012). Direct mortality from consumption of ARs has been documented in birds and small mammals (Sánchez-Barbudo et al. 2012).

Exposure to ARs may also compromise the immune system of non-target species making them vulnerable to pathogens and pesticides. Riley (2007) found that AR exposure predisposed wild felids (bobcats (*Lynx rufus*) and mountain lions (*Puma concolor*)) to notoedric mange. Furthermore, voles that were exposed to ARs exhibited higher prevalence of the bacteria that causes tularemia, a zoonotic disease (Vidal et al. 2009).

2.2.2 Indirect Effects

The indirect effects of ARs on wildlife include starvation due to decreased prey availability, secondary poisoning, reduction in clotting mechanisms, and hypothermia.

Similarly to other pesticides, AR exposure may result in predator starvation as prey populations have been shown to be affected by rodenticide use (Wengert 2015). Secondary poisoning from ARs is also common; as the rodenticide accumulates in the prey species, they are easily captured by predators in their weakened state (Berny et al. 1997, Berny 2007). Approximately 70% of animals sampled by CDFW test positive for at least one AR compound (Daniels 2013); they have been found in a variety of taxa including mammals (Littrell and Hunter 1988, Alterio et al. 1997, Stone et al. 1999, Hosea 2000, Fournier-Chambrillon et al. 2004, Riley et al. 2007, McMillin et al. 2008, Proulx and Mackenzie 2012), corvids (Howald et al. 1999, Stone et al. 1999), raptors (Mendenhall and Pank 1980, Hegdal and Colvin 1988, Stone et al. 1999, 2003, Hosea 2000, Franklin et al. 2018, Gabriel et al. 2018), and turkeys (Hosea 2000) (see Appendix B for complete list). Additionally, Burns-Edel (2016) documented secondary poisoning of herbivores through feeding on vegetation which had absorbed rodenticide compounds.

One particular concern from AR use is their impact on rare carnivores of conservation concern. Several studies have found that ARs are a cause of mortality for Pacific fishers (*Pekania pennanti*), a candidate for listing under the ESA and CESA as well as a species of special concern in California (Gabriel et al. 2012, 2015, Thompson et al. 2014). Thompson et al. (2014) found that survival of female fishers was linked to the number of cannabis cultivation sites within their home ranges, and therefore, cultivation sites that utilize ARs may present a similar risk to other carnivores of concern in California including Sierra Nevada red fox (*Vulpes vulpes necator*), Humboldt (coastal) marten (*Martes caurina humboldtensis*), wolverine (*Gulo gulo*), gray wolf (*Canis lupus*), as well as raptors such as northern spotted owl (*Strix occidentalis caurina*), California spotted owl (*S*. *occidentalis occidentalis*), and great gray owls (*S. nebulosa*; Gabriel et al. 2012).

Sub-lethal exposure to ARs may also endanger wildlife by decreasing the ability of animals to clot properly (Valchev et al. 2008). Erickson and Urban (2004) found numerous accounts of predators, particularly raptors, with relatively low concentrations of ARs in their system dying from excessive bleeding as a result of minor wounds from their prey. Examples of this phenomenon have also been documented in screech owls (*Otus asio*; Rattner et al. 2012), barn owls (Webster 2009), and least weasels (*Mustela nivalis*; Townsend et al. 1984). Additionally, similarly to other pesticides, sub-lethal exposure to ARs may cause short-term hypothermia in birds and mammals compromising their ability to thermoregulate (Jaques 1959, Grue et al. 1991, Gordon 1994).

2.3 Fertilizers and Imported Soils

Cultivation of cannabis requires a nitrogen-rich soil environment (O'Hare et al. 2013), and thus, many cultivators use fertilizers and imported soils to increase the nitrogen content of the local soils.

Fertilizers can have a variety of negative impacts on ecosystems. They can decrease species diversity and abundance (Kleijn and Snoeijing 1997), and also decrease activity of aquatic species, including frog tadpoles (Xu and Oldham 1997). Nutrient enrichment will often increase the abundance of pests and pathogens, including those that impact wildlife (Matson et al. 1997, Johnson et al. 2010). For example, fertilizer inputs are often correlated with increases in the occurrence, severity, and distribution of infectious diseases (Johnson et al. 2010). Also, many outdoor cannabis grows include imported soils that may contain invasive plant or animal species that can harm native biodiversity (Butsic and Brenner 2016).

Excess nutrients from fertilizers that wash into watersheds can also have negative consequences for wildlife. They can cause nutrient imbalances in the watersheds (Mallery 2010) and, through pollution of the watershed, can kill fish and other wildlife (NDIC 2007). Fertilizers often cause algae outbreaks in water systems (Mallery 2010), which, when they begin to decay, can deplete the water of oxygen, suffocating fish and other aquatic life (Bland 2014). Algae outbreaks in wetlands have also been shown to increase the abundance of parasites, such as trematodes (*Ribeiroia ondatrae*) that cause limb deformities in amphibians (Johnson et al. 2010). Additionally, fertilizers can enter and contaminate groundwater as well (NDIC 2007).

3. Water Impacts

According to Dudgeon et al. (2006), four of the five greatest threats to freshwater biodiversity today are flow modification, water pollution, habitat degradation, and species invasions. All four of these threats are common consequences of cannabis cultivation. On the west coast, 60% of amphibians, 16% of reptiles, 34% of birds, and 12% of mammals are classified as riparian obligates (Kelsey and West 1998).

3.1 Water Diversion

The primary method by which cannabis cultivation may impact wildlife is through water diversions. California has a Mediterranean climate in which most precipitation occurs during the winter months. Thus, during the growing season for cannabis (May-September), there is very little precipitation. As each cannabis plant requires about 22.7 L (6 gal) of water per day, growers must acquire water through alternate means, most commonly through irrigation by diverting springs and headwater streams. Consequences of water diversion include changes in flow regimes, fish passage barriers, loss of wildlife habitat, changes in water properties, rerouting of streams, and dewatered streams.

3.1.1 Changes in Flow Regimes

Reduced instream flows, prolonged low flows, and loss of seasonal flow peaks can have a number of impacts on wildlife, and changes in flow rates are likely to become even more pronounced as the climate changes (Deitch et al. 2018). High flows remove and transport fine sediment downstream (Poff et al. 1997); without these flows, streams may become graded or buried, decreasing available habitat for aquatic species. Reduction in flow can also cause channels to become disconnected from floodplains resulting in decreased productivity; floodplains are important nursery grounds for some fish species, and they transfer organic matter and organisms into the main channel (Poff et al. 1997). When fish lose access to backwater wetlands, they can experience reduced reproduction and recruitment (Junk et al. 1989, Sparks 1995). These decreases in habitat availability can increase both intra and interspecific competition as well as likelihood of predation (CDFG 2004). Changes in flow rates can also increase the prevalence of invasive species including plants (Horton 1977, Friedman et al. 1998) and fish (Gehrke et al. 1995).

Decreased flows can also increase mortality and negatively impact abundance and diversity of a variety of species. Salmonids, for example, require suitable flow regimes (Moyle 2002). Water diversions have been shown to increase mortality of both juvenile and adult coho salmon (*Oncorhynchus kisutch*; CDFG 2004, CDFW 2015), and Almodovar and Nicola (1999) found that reduced flows can lead to decreased density and biomass of brown trout (*Salmo trutta*). Flow rates can be particularly important for survival of salmonids that live in intermittent streams (Obedzinski et al. 2018). Low flows can result in the loss of sensitive fish species, such as fluvial specialists, leading to decreased diversity (Gehrke et al. 1995, Travnichek et al. 1995, Humphries et al. 2002, Irwin and Freeman 2002, Anderson et al. 2006, Freeman and Marcinek 2006). Reduced flows can also lead to stagnant water conditions, a situation that allows the growth of harmful cyanobacteria resulting in mortality of salmonids and other aquatic animals (Power et al. 2015)

Amphibians can also be sensitive to decreased flows; plethodontid salamanders are intolerant to desiccation and thus vulnerable to headwater stream diversions (Ray 1958). Kupferberg et al. (2012) reported that low flows were strongly correlated with early life stage mortality and decreased adult densities of foothill yellow-legged frogs (*Rana boylii*) and California red-legged frogs (*Rana draytonii*), both species of special concern in California. Plant cover and diversity can also be decreased by reduced flows (Busch and Smith 1995, Stromberg et al. 1996), likely as a result of physiological stress leading to reduced growth rates and recruitment, morphological changes, and mortality (Reily and Johnson 1982, Perkins et al. 1984, Fenner et al. 1985, Kondolf and Curry 1986, Rood and Mahoney 1990). Wash-out and stranding of fish and other aquatic species can also be a consequence of reduced flows (Cushman 1985).

Fish use stream flows (high and low flows) as cues for certain life cycle transitions, and therefore, prolonged low flows can disrupt natural cues and result in changes in timing of life cycle events (Poff et al. 1997). Spawning and egg hatching can be disrupted by sustained low flows (Montgomery et al. 1983, Næsje et al. 1995, Fausch and Bestgen 1997), and migration can be delayed (Jonsson 1991; CDFG 2004).

Reduced seasonal flows can also decrease food supply for aquatic species (CDFG 2004). McKay and King (2006) reported decreased diversity of macroinvertebrates in response to low flows. Such changes can result in a substantial alteration of the aquatic food webs (Power 1992, Wootton et al. 1996). Decreases in prey availability (e.g., macroinvertebrates) can significantly decrease growth rates of salmonids (Harvey et al. 2006).

3.1.2 Changes in Water Properties

Water diversions can alter dissolved oxygen levels, nutrient contents, and pH as well as increase water temperatures (O'Hare et al. 2013). Reduced flow rates are correlated with increases in water temperatures as the volume of water in streams decreases. This presents threats for salmonids as increased temperatures have been shown to reduce growth rates, increase predation risk, and increase susceptibility to disease (Moore and Townsend 1998, Marine and Cech, Jr. 2004). Amphibians that live in headwater streams are also sensitive to changes in water temperature including the southern torrent salamander (*Rhyacotriton variegatus*) a species of special concern in California (Welsh and Lind 1996, Bury 2008). When water temperature increases, it holds less dissolved oxygen, which can be problematic for aquatic animals that are reliant on the oxygen. For example, reductions in dissolved oxygen can decrease survival of juvenile salmonids (Selong et al. 2001, Moyle 2002, Martins et al. 2011). Additionally, warmer water has a lower pH, and the increased acidity of the water may also have negative consequences for aquatic organisms.

3.1.3 Dewatered Streams

In addition to reduced flows, water diversion can also be responsible for dewatering streams completely. A study by Deitch et al. (2009) found that in watersheds in Sonoma County, CA, demand of registered water diversions was greater than stream flows during certain parts of the year. Similarly, Carah et al. (2015) found that estimated water demand for cannabis cultivated along the Eel River was ten times higher than could be sustained by the watershed.

Streams that dry up may be used by a variety of wildlife including aquatics but also numerous non-aquatic species as well. Some salmonids, such as cutthroat trout (*Oncorhynchus clarkii)* and juvenile coho salmon, are known to use small streams that would be at risk of being dewatered by diversions (Richardson et al. 2005). Amphibians such as the California giant salamander (*Dicamptodon ensatus*) and southern torrent salamander are often dependent on small streams, particularly during summer months (Johnston and Frid 2002, Richardson et al. 2005). Also, small streams may provide areas free from predators for Pacific tailed frogs (*Ascaphus truei*; Dupuis and Steventon 1999, Sheridan and Olson 2003). Reptiles including turtles and snakes are also known to use small streams (Meyer et al. 2007), and dippers (*Cinclus mexicanus*) are one of a few bird species known to live in small streams (Richardson et al. 2005).

There are also a variety of species that, while not dependent on streams, use them regularly. Many birds use streams for resources including food, water, and habitat as well as for movement including flycatchers, woodpeckers, jays, warblers, and hummingbirds (Murray and Stauffer 1995, Lock and Naiman 1998, Meyer et al. 2007). Marbled murrelets require access to streams near their nest sites in forests to float fledglings to coastal areas (Sealy 1972). Small mammals like Pacific water shrews (*Sorex bendirii*) also use small streams (Gomez and Anthony 1998), and cervids including mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) often use streams, particularly in summer months, but also intermittently during winter as well (Ager et al. 2003, D'Eon and Serrouya 2005). Streams are also an essential component of fisher habitat, particularly in regards to rest sites; these sites are especially important to fishers in many areas of California that experience hot, dry conditions in the summer—including in the Sierras (Zielinski et al. 2004). Bats, such as the California myotis (*Myotis californicus*) and Townsend's big-eared bat (*Corynorhinus townsendii*), commonly use streams for both traveling and foraging for insects (Seidman and Zabel 2001, Salvarina 2016).

3.1.4 Other Impacts

Water diversions can be barriers to fish passage if they are improperly designed. Additionally, diversions can result in rerouting of streams and channelization, which reduces habitat complexity, can cause terrestrialization of the flora, and reduce species evenness (Deiller et al. 2001). In certain circumstances, groundwater pumping and wells can lead to diversion of surface water and streamflow depletion (Barlow and Leake 2012).

3.2 Dams and Stream Crossings

Construction of dams and stream crossings used for cannabis cultivation can also have negative impacts on ecosystems. These constructions can cause downstream channel erosion and tributary head-cutting, reduced magnitude and frequency of high flows (see section 3.1.1 for impacts of prolonged low flows), channel narrowing, and reduced formation of secondary channels and oxbows (Poff et al. 1997, Asarian and Walker 2016). Additionally, dams and stream crossings can degrade water quality and associated wildlife habitats (Santucci, Jr. et al. 2005). Streams with such constructions can have reduced abundance of anurans due to decreased availability of breeding habitat (Eskew et al. 2012). Breeding populations of foothill yellow-legged frogs, for example, are five times smaller in rivers with dams (Kupferberg et al. 2012). Stream crossings may also act as barriers to salmonids, particularly during migration (Furniss et al. 1991, Rieman et al. 1997). For example, trout biomass has been shown to be negatively correlated to the number of road crossings on a stream (Eaglin and Hubert 1993).

3.3 Delivery of Pollutants

Cultivation of cannabis can also result in delivery of sediment, nutrients, petroleum products, and pesticides into streams, degrading the water quality and increasing turbidity (Reid and Dunne 1984, David A. Alvarez et al. 2008, Carah et al. 2015). Run-off from pesticides and fertilizers has been shown to have a number of negative consequences for aquatic life including external lesions, intersex in fish, and mortality (Alvarez et al. 2008*b*). Sediment that washes into streams can smother gravel beds where salmonids spawn. Moreover, sedimentation can impair growth and survival of juvenile salmonids (Suttle et al. 2004, NDIC 2007). Sediment in streams can also make the water cloudy which decreases the ability of organisms to photosynthesize (Mallery 2010). Vegetation cleared to provide room for cannabis plants is often discarded into stream beds where it can cause barriers to hydrologic flows (Mallery 2010). Amphibians that reside in streams have also been shown to be sensitive to sedimentation and vegetation debris (Welsh and Ollivier 1998, Welsh and Hodgson 2008).

4. Terrestrial Impacts

4.1 Site Development

Even before cultivation begins, development of a cultivation site can have substantial impacts on wildlife. The impacts from site development come from activities that include road construction, fencing, construction of ponds and artificial water sources, greenhouse construction, vegetation clearing, and forest conversion. These activities cause habitat fragmentation that can impact wildlife movement and eliminate corridors.

Often, cannabis sites require the construction of new roads to access cultivation areas. Wildlife mortality can occur as a result of road construction (Trombulak and Frissell 2000), and there is a great deal of research showing that roads can increase the spread of invasive species (Brothers and Spingarn 1992, Greenberg et al. 1997, Gelbard and Belnap 2003, Ansong and Pickering 2013). Additionally, roads can cause soil erosion and surface run-off that can transfer sediment into streams (see section 3.3 for impacts of stream sedimentation) (Beschta 1978, Seyedbagheri 1996, Richardson et al. 2001). Vegetation clearing for road construction can also increase the amount of light that penetrates the forest floor, which may result in changes in species composition (Trombulak and Frissell 2000). Fencing erected around cultivation sites during site construction can also be a hazard to wildlife causing entanglement and mortality (van der Ree 1999, Stuart et al. 2001).

Because of the large water needs of cannabis plants, cultivation sites may construct ponds or other artificial water sources to ensure reliable access to water during the growing season (Bauss 2017). If these ponds are not constructed with proper engineering, they can pose a threat to water quality through delivery of sediment to nearby streams. They also may result in substantial grading and fill in the area. Such water constructions have also been shown to be breeding habitat for invasive species such as the American bullfrog (*Lithobates catesbianus*; Kiesecker et al. 2001, Fuller et al. 2011), which prey on native anurans of special concern including northern red-legged frogs and foothill yellow-legged frogs (Moyle 1973, Kiesecker and Blaustein 1997, 1998, Kupferberg 1997). Also, the presence of artificial water sources can increase the spread of invasive Argentine ants (*Linepithema humile*) which displace native invertebrates (Human and Gordon 1997, Holway et al. 2002).

Some cultivation sites include the construction of greenhouses (Bauss 2017). These greenhouses may require fuel clearance (under fire codes); these areas often become degraded and are prone to establishment by invasive species. Greenhouses are often constructed in 100-year floodplains that require grading and fill; they frequently have concrete floors, which create a permanent construction footprint that cannot be readily converted back to floodplain (Poff et al. 1997). Wang et al. (2017) found that development in such areas can disconnect rivers from their natural floodplains, as well as displace, fragment, and degrade essential riparian habitat. Furthermore, development in floodplains can reduce the benefits of natural flooding regimes including deposition of river silt on valley floor soils and recharging of wetlands. Additional changes such as alterations in channel structure and elimination of backwaters that result in higher velocity flows may negatively impact salmonids which require low flow refugia (Moyle 2002).

Development of a cultivation site can often include clearing of existing vegetation which can have numerous impacts on the local ecosystem (NDIC 2007, Mallery 2010, Milestone et al. 2011, Gabriel et al. 2012). Vegetation removal may result in the loss of special status plant species and the loss of habitat that supports pollinators and birds, particularly habitats necessary during the breeding season. Clearing may also cause fragmentation and loss of sensitive habitats and create edge effects that permeate far beyond the cultivation site (Harris 1988, Murcia 1995). Recent research suggests that cannabis cultivation sites are more likely to be clumped in space, further increasing the effects of fragmentation from vegetation clearing (Butsic et al. 2017).The activities associated with clearing may also disturb associated soil seed banks that sustain local plant populations. Removal of vegetation has also been shown to make communities vulnerable to colonization by invasive plant species and to spread the pathogen responsible for Sudden Oak Death syndrome (*Phytophor ramorum*; Mallery 2010). Additionally, the abundance of dried vegetation remaining after removals may increase risk for fires.

Forest conversion may also be a result of cannabis site development (Burns-Edel 2016, Wang et al. 2017). Forest conversion can lead to loss of nutrient-rich topsoils, disrupted nutrient cycling, and increased erosion (NDIC 2007, Mallery 2010). It may also result in increased exposure of species to predation risk and climate stress. Wang et al. (2017) found that cannabis cultivation sites cause both forest loss and conversion of large habitat patches to small, fragmented patches with greater edge and less interior core areas. They found that the per-unit-area effects of cannabis cultivation were similar or even greater than the effects of timber harvest (Wang et al. 2017). Additionally, areas that have been previously harvested for timber are more likely to be cultivation sites, which could lead to further conversion and degradation of these areas (Butsic and Brenner 2016)

4.2 Site Use and Maintenance

The use and maintenance of cannabis cultivation sites can have a number of impacts on wildlife. The presence of trash and other wastes can be detrimental if consumed by wildlife, and, if the sites are located near streams, they may become pollutants (NDIC 2007, SWB 2013). Also, use of roads, noise from the cultivation site, and the presence of artificial lighting may all have effects on wildlife.

4.2.1 Road Use

Roads and their associated vehicle traffic can have a number of environmental impacts including alteration of the physical and chemical environments, wildlife mortality, altered abundances and diversity of wildlife, and modification of animal behavior.

Road presence and use can alter the physical and chemical environment of the ecosystem in ways that can impact wildlife. Road use results in soil compaction and decreased moisture content under the road, even when the road is not frequently used (Vora 1988, Helvey and Kochenderfer 1990). Temperatures are increased on road surfaces which creates a heat island that may attract animals; for example, birds and snakes congregate on roads which increases their risk of mortality (Whitford 1985). Dust is dispersed from traffic which, when deposited on plants, can hinder physiological process including photosynthesis, respiration, and transpiration as well as cause physical injury to the plants (Farmer 1993). Auerbach et al. (1997) found that dust mobilization can decrease species richness and alter plant community structure. Road traffic can also supply fine sediments and contaminants to aquatic systems, which decreases the clarity (Gjessing et al. 1984, Reid and Dunne 1984); ultimately, this can negatively impact productivity as well as survival and growth of fishes (Newcombe and Jensen 1996). Additionally, roads can disrupt surface flow of water, redirecting it to the roadway (Wemple et al. 1996). This redirection can then result in changes in both timing and the direction of the runoff (King and Tennyson 1984), the effects of which are most evident in smaller streams, such as those commonly near cannabis sites (Wemple et al. 1996). Road diversions of groundwater may also result in high amounts of runoff on hillslopes that can trigger erosion (Seyedbagheri 1996, Wemple et al. 1996, Richardson et al. 2001) which can negatively impact fish and other aquatic organisms downstream for long periods of time (Hicks et al. 1991). Road use may alter the chemical environment through heavy metal contamination which can accumulate in the tissues of plants and animals (Birdsall et al. 1986, Grue et al. 1986).

Traffic on roads can also result in the mortality of wildlife as well as alter the abundance and diversity of species (Trombulak and Frissell 2000). Morality from roads has been documented in raptors (Loos and Kerlinger 1993, Varland et al. 1993, Newton et al. 1997), granivorous birds (Dhindsa et al. 1988), snakes (Rosen and Lowe 1994), amphibians (van Gelder 1973), and mammals (Bashore et al. 1985, Fuller 1989, Bjurlin and Cypher 2003). Furthermore, road presence can also decrease species abundance and diversity. Findlay and Houlahan (1997), for example, found that herptile (reptiles and amphibians) diversity in wetlands declined relative to the density of roads. Even fully aquatic organisms are affected; two studies have reported that the abundance of bull trout, an endangered species in California, was negatively related to road density (Rieman et al. 1997, Baxter et al. 1999).

The presence of roads may also cause changes in the behavior of animals. Road presence has been shown to shift home ranges of a variety of mammals including bears (*Ursus* spp.; McLellan and Shackleton 1988, Brody and Pelton 1989), elk and mule deer (Rost and Bailey 1979, Grover and Thompson 1986), wolves (Thurber et al. 1994, Newcombe and Jensen 1996), and mountain lions (Van Dyke et al. 1986). Roads may also cause alterations in movement at smaller scales as well; a variety of both small and large vertebrates modify their movements in relation to roads (Oxley et al. 1974, Bruns 1977, Swihart and Slade 1984, Van Dyke et al. 1986, Brody and Pelton 1989, Merriam et al. 1989). Roads have also been reported to do decrease the reproductive success of some bird species including bald eagles (Anthony and Isaacs 1989) and sandhill cranes (*Grus canadensis*; Norling et al. 1992), both fully protected species in California. The impacts of roads on wildlife behavior appears to be independent of how frequently they are used. MacArthur et al. (1979) found the energy expenditure, as well as heart and metabolic rates, of female big horn sheep (*Ovis canadensis*) increased near roads regardless of their use. Furthermore, carnivores including gray fox, bobcat, black bear (*Ursus americanus*), badger (*Taxidea taxus*), and ringtail (*Bassariscus astutus*) have also been shown to avoid roads irrespective of their traffic volume (Baker and Leberg 2018).

4.2.2 Noise

Cannabis cultivation sites often have substantial amounts of noise pollution resulting from road use, generators, and other equipment. This is concerning as wildlife responses to noise can occur at exposure levels of only 55-60 dB (Barber et al. 2009). (For reference, normal conversation is approximately 60 dB.) The impacts of noise on wildlife include disrupted communication, changes in predator-prey relationships, effects on foraging efficiency, changes in habitat selection, abundance, density, and diversity, increased stress and decreased immune response, behavioral changes, and effects on reproduction.

Anthropogenic noise can disrupt the communication of many wildlife species (Patricelli and Blickley 2006). Frogs will often decrease their calling activity in response to noise (Sun and Narins 2005, Lengagne 2008, Caorsi et al. 2017). When exposed to noise, birds will sign at a higher pitch to ensure mating calls are heard, which has associated energy costs (Slabbekoorn and Peet 2003, Brumm 2004). If bird songs are not transmitted properly to their intended receivers (e.g., intraspecific males and females), territory occupancy and mate attraction may be negatively affected (Klump 1996). Similar to birds, bats have been shown to alter their echolocation call structure when subjected to anthropogenic noise (Gillam and McCracken 2007), and frogs increase the pitch of their calls (Parris et al. 2009).

Noise exposure can also impact predator-prey relationships. This can occur through changes in the spatial distribution of predator or prey species or through alterations in their movements. Noise may decrease a predator’s ability to hear its prey or vice versa. Noise may be especially impactful on nocturnal animals that primarily use hearing to hunt such as owls and bats. Additionally, prey species have been shown to increase their vigilance rates and anti-predator behavior in response to noise (Francis and Barber 2013). Many prey species increase their vigilance behavior when exposed to noise because they need to rely more on visual detection of predators when auditory cues may be masked by noise (Rabin et al. 2006, Quinn et al. 2017).

Relatedly, foraging efficiency of some wildlife species has been shown to decline in response to anthropogenic noise (Miksis-Olds et al. 2007). Bats have reduced foraging success in areas with chronic noise, and this has been correlated to the decline of 12 bat species in California that are either endangered or of special concern (Schaub et al. 2008, Siemers and Schaub 2011). Chicks of tree swallows (*Tachycineta bicolor*) that are exposed to noise fail to beg when parents return with food (Leonard and Horn 2012). Also, the structure of begging calls from chicks can be affected, and these alterations continue even when the noise is no longer present (Leonard and Horn 2008).

Noise can also impact habitat selection of species as well as abundance, density, and diversity (Francis and Barber 2013). Bats, for example, have been shown to avoid areas with anthropogenic noise (Schaub et al. 2008, Siemers and Schaub 2011). Noise has also been shown to reduce the density of nesting birds (Francis et al. 2009). A study by Bayne et al. (2008) compared areas with natural resource extraction that had low levels of noise to those that had high levels of noise and found that those with high levels of noise had significantly reduced abundance and density of the songbirds.

Exposure to noise can also cause increased stress in wildlife and result in decreased immune responses (Kight and Swaddle 2011). Blickley et al. (2012) reported that noise caused elevated levels of stress hormones in lekking male greater sage grouse (*Centrocercus urophasianus*). Northern spotted owls exposed to vehicle noise also had increased levels of stress hormones; this was particularly evident in males during times when they were exclusively responsible for feeding their mates and nestlings (Hayward et al. 2011). There is also evidence that noise can have an immunosuppressive effect in frogs (Troïanowski et al. 2017).

Reproduction is another aspect that can be impacted by anthropogenic noise. Noise exposure can cause weakened pair preference in birds (Swaddle and Page 2007) as well as reduced pairing success that can lead to a decline in overall reproductive success (Habib et al. 2007). For example, the low frequency songs of great tits (*Parus major*) become ineffective in noisy environments, and these songs are strongly correlated with female fertility and sexual fidelity (Halfwerk et al. 2011). Hebert and Golightly (2006) also suggested that noise may influence the survival and nest success of marbled murrelets (*Brachyramphus marmoratus*), an endangered species in California. In addition, female gray tree frogs (*Hyla versicolor*) cannot successful orient to male calls in the presence of noise, which likely has consequences on their reproductive success (Bee and Swanson 2007)

4.2.3 Artificial Lighting

Cannabis cultivation sites are increasingly using artificial lighting both in greenhouses and for “mixed-light” techniques to increase yields. This lighting can result in substantial light pollution effects on wildlife that include disruption of circadian rhythms and suppressed immune response, changes in foraging behavior, altered navigation and migration patterns, altered predator-prey relationships, impacts on reproduction, and phototaxis. The lighting materials used in cannabis cultivation also have environmental risks if not disposed of properly as they contain mercury and other toxins (O’Hare et al. 2013).

Disruption of circadian rhythms due to light pollution can have both physiological and behavioral consequences for wildlife. Songbirds that live in areas with artificial lights often begin morning choruses during night hours (Derrickson 1988, Miller 2006, Fuller et al. 2007). Artificial lighting can also have negative impacts on bat roosts (Johnston et al. 2004). The lesser horseshoe bat (*Rhinolophus hipposideros*), for example, showed significantly decreased activity and a delay in the start of commuting behavior when exposed to light (Stone et al. 1999). Larval amphibians like American toads (*Bufo americanus*) use photoperiod cues to behaviorally thermoregulate (Beiswenger 1977). Additionally, exposure to artificial light disrupts the production of melatonin in tiger salamanders (*Ambystoma tigrinum*), which ultimately can alter their metabolic rates and requiring them to increase time spent foraging (Perry et al. 2008). Gene expression can also be altered in animals that experience constant illumination (Perry et al. 2008). Finally, exposure to artificial light can suppress the immune response of species resulting in increased pathogen and parasite infections as well as increased tumor growth (Navara and Nelson 2007); this has been demonstrated in a variety of species from birds (Moore and Siopes 2000) and mammals (Bedrosian et al. 2011) to fish (Leonardi and Klempau 2003).

Artificial lighting can also cause changes in foraging behavior. Many animals decrease foraging in high light levels because of the higher risk of predation; this includes rodents (Clarke 1983, Daly et al. 1992), seabirds (Mougeot and Bretagnolle 2000), rabbits (Gilbert and Boutin 1991), bats (Rydell 1992), and fish (Gibson 1978). Beach mice (*Peromyscus polionotus*), for example, decreased foraging in the presence of artificial light (Bird et al. 2004). Light pollution has been shown to disrupt night foraging in birds (CDFG 2007) and affect feeding patterns in juvenile salmon (Valdimarsson et al. 1997). The Pacific tailed frog (*Ascaphus truei*), a species of special concern in California, is normally active at only the darkest times of night (Hailman 1982); thus, they are likely to be influenced when artificial lighting causes them to decrease activity.

Light pollution can also disrupt navigation and migration patterns as changes in ambient light guide migration patterns in a variety of species including salmonids, birds, butterflies, and eels (Rowan 1932, Lowe 1952, Grau et al. 1981, Froy et al. 2003). The migration of Pacific salmon species can be slowed or halted by the presence of artificial lights (Nightingale et al. 2006), as can out-migration of juvenile salmon (Tabor et al. 2004). Also, exposure to light can decrease smoltification and body condition in Chinook salmon (*Oncorhynchus tshawytscha*; Hoffnagle and Fivizzani 1998). Additionally, artificial light can attract and disorient birds, disrupting their migration (Ogden 1996, Longcore and Rich 2016). Similarly, orientation and homing behavior of red-spotted news (*Notophthlamus viridescens*) can be disrupted by artificial light (Phillips and Borland 1992, 1994). The vertical migration of larval salamanders (*Ambystoma* spp.) is also influenced by ambient light levels (Anderson and Graham 1967), and the disruption of their daily vertical movements can reduce growth and survival (Semlitsch 1987).

Predator-prey relationships can also be altered by artificial light. Predators may forage during times they normally would not, thus, overexploiting prey. Conversely, prey activity may decrease, decreasing the availability of prey for predators (Navara and Nelson 2007). For example, heteromyid rodents (pocket mice and kangaroo rats) showed reduced foraging behavior in the presence of artificial lighting as it was correlated with increased predation risk from owls (Brown et al. 1988). Juvenile salmon have also been shown to be more vulnerable to predation with increased light (Ginetz 1972, Tabor et al. 2004).

Artificial lighting may also impact reproduction of wildlife. The nest site choices of black-tailed godwits (*Limos limosa*), for example, are influenced by artificial lighting (Longcore and Rich 2004). In an experiment with juncos (*Junco* sp.), Rowan (1925) discovered that exposure to light can alter timing of breeding; juncos exposed to just a few minutes of artificial light came into reproductive condition despite it still being winter. Light pollution can also decrease night chorusing and mating activity of frogs (Longcore and Rich 2004).

Phototaxis, a phenomenon which results in attraction and movement towards light, can disorient, entrap, and temporarily blind wildlife species that experience it (Longcore and Rich 2004). One well-researched example of this is juvenile sea turtles emerging from nests of sandy beaches often go toward the lights inland instead of toward the sea (Witherington and Bjorndal 1991, Salmon et al. 1995). Anurans, including frogs and toads, have also been shown to congregate at artificial light sources (Buchanan 2006).

5. Direct Ingestion

Wildlife may also directly ingest cannabis plants; the stalks can be enticing to deer, rodents, and potentially other herbivores or omnivores (Mallery 2010). However, the risks of direct ingestion in wildlife have not yet been well studied. Driemeier (1998) found that marijuana consumption can be lethal when consumed by ruminants. Also, evidence from accidental ingestion by canid and felid pets demonstrates that cannabis can cause vomiting, hypothermia, dehydration, changes in heart rate, seizures, and comas (Donaldson 2002, Fitzgerald et al. 2013).

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Appendix A. Birds that have documented pesticide poisonings and their status. (*Sources: Nettles 1976, Henny et al. 1987, Litterell 1988, Augspurger et al. 1996, Mineau et al. 1999, Fleischli et al. 2004, Pimentel 2004*)

Order	Common Name	*Scientific Name*	Special Status*
Accipitriformes	Osprey	*Pandion haliaetus*	CDF-S, CDFW-WL
Anseriformes	American wigeon	*Anas americana*	
Anseriformes	Black-bellied whistling-duck	*Dendrocygna autumnalis*	
Anseriformes	Blue-winged teal	*Anas discors*	
Anseriformes	Brant goose	*Branta bernicla*	CDFW-SSC
Anseriformes	Bufflehead	*Bucephala albeola*	
Anseriformes	Canada goose	*Branta canadensis*	
Anseriformes	Cinnamon teal	*Anas cyanoptera*	
Anseriformes	Fulvous whistling-duck	*Dendrocygna bicolor*	CDFW-SSC
Anseriformes	Gadwall	*Anas stepera*	
Anseriformes	Greater white-fronted goose	*Anser albifrons*	CDFW-SSC
Anseriformes	Green-winged teal	*Anas crecca*	
Anseriformes	Lesser scaup	*Aythya affinis*	
Anseriformes	Mallard	*Anas platyrhynchos*	
Anseriformes	Mottled duck	*Anas fulvigula*	
Anseriformes	Muscovy duck	*Cairina moschata*	
Anseriformes	Northern pintail	*Anas acuta*	
Anseriformes	Northern shoveler	*Anas clypeata*	
Anseriformes	Ring-necked duck	*Aythya collaris*	
Anseriformes	Ross's goose	*Chen rossii*	
Anseriformes	Snow goose	*Chen caerulescens*	
Anseriformes	Wood duck	*Aix sponsa*	
Charadriiformes	Black tern	*Chlidonias niger*	CDFW-SSC
Charadriiformes	Caspian tern	*Sterna caspia*	
Charadriiformes	Common snipe	*Gallinago gallinago*	
Charadriiformes	Dunlin	*Calidris alpina*	
Charadriiformes	Forster's tern	*Sterna forsteri*	

Order	Common Name	*Scientific Name*	Special Status*
Charadriiformes	Herring gull	*Larus argentatus*	
Charadriiformes	Killdeer	*Charadrius vociferus*	
Charadriiformes	Laughing gull	*Larus atricilla*	CDFW-WL
Charadriiformes	Least sandpiper	*Calidris minutilla*	
Charadriiformes	Ring-billed gull	*Larus delawarensis*	
Charadriiformes	Semipalmated plover	*Charadrius semipalmatus*	
Charadriiformes	Semipalmated sandpiper	*Calidris pusilla*	
Ciconiiformes	Black vulture	*Coragyps atratus*	
Ciconiiformes	Cattle egret	*Bubulcus ibis*	
Ciconiiformes	Glossy ibis	*Plegadis falcinellus*	
Ciconiiformes	Great blue heron	*Ardea herodias*	CDF-S
Ciconiiformes	Great egret	*Ardea alba*	CDF-S
Ciconiiformes	Snowy egret	*Egretta thula*	
Ciconiiformes	Turkey vulture	*Cathartes aura*	
Columbiformes	Inca dove	*Columbina inca*	
Columbiformes	Mourning dove	*Zenaida macroura*	
Columbiformes	Rock dove	*Columba livia*	
Falconiformes	American kestrel	*Falco sparverius*	
Falconiformes	Bald eagle	*Haliaeetus leucocephalus*	BLM-S, CDFW-FP, USFS-S, USFWS-BCC
Falconiformes	Cooper's hawk	*Accipiter cooperii*	CDFW-WL
Falconiformes	Ferruginous hawk	*Buteo regalis*	CDFW-WL, USFWS-BCC
Falconiformes	Golden eagle	*Aquila chrysaetos*	BLM-S, CDFW-FP, CDFW-WL, USFWS-BCC
Falconiformes	Merlin	*Falco columbarius*	CDFW-WL
Falconiformes	Mississippi kite	*Ictinia mississippiensis*	
Falconiformes	Northern harrier	*Circus cyaneus*	CDFW-SSC
Falconiformes	Peregrine falcon	*Falco peregrinus*	CDF-S, CDFW-FP, USFWS-BCC
Falconiformes	Prairie falcon	*Falco mexicanus*	CDFW-WL, USFWS-BCC
Falconiformes	Red-shouldered hawk	*Buteo lineatus*	
Falconiformes	Red-tailed hawk	*Buteo jamaicensis*	
Falconiformes	Rough-legged hawk	*Buteo lagopus*	

Order	Common Name	*Scientific Name*	Special Status*
Falconiformes	Sharp-shinned hawk	*Accipiter striatus*	CDFW-WL
Falconiformes	Swainson's hawk	*Buteo swainsoni*	BLM-S, USFWS-BCC
Falconiformes	White-tailed kite	*Elanus leucurus*	BLM-S, CDFW-FP
Galliformes	Greater sage-grouse	*Centrocercus urophasianus*	BLM-S, CDFW-SSC, IUCN-NT, USFS-S
Galliformes	Northern bobwhite	*Colinus virginianus*	
Galliformes	Wild turkey	*Meleagris gallopavo*	
Gruiformes	American coot	Fulica americana	
Gruiformes	Sandhill crane	*Grus canadensis*	CDFW-SSC, BLM-S, CDFW-FP, USFS-S
Passeriformes	American crow	*Corvus brachyrhynchos*	
Passeriformes	American goldfinch	*Carduelis tristis*	
Passeriformes	American robin	*Turdus migratorius*	
Passeriformes	American tree sparrow	*Spizella arborea*	
Passeriformes	Barn swallow	*Hirundo rustica*	
Passeriformes	Black-billed magpie	*Pica hudsonia*	
Passeriformes	Black-capped chickadee	*Poecile atricapilla*	CDFW-WL
Passeriformes	Blue jay	*Cyanocitta cristata*	
Passeriformes	Boat-tailed grackle	*Quiscalus major*	
Passeriformes	Brewer's blackbird	*Euphagus cyanocephalus*	
Passeriformes	Brown thrasher	*Toxostoma rufum*	
Passeriformes	Brown-headed cowbird	*Molothrus ater*	
Passeriformes	Cedar waxwing	*Bombycilla cedrorum*	
Passeriformes	Common grackle	*Quiscalus quiscula*	
Passeriformes	Common raven	*Corvus corax*	
Passeriformes	Common yellowthroat	*Geothlypis trichas*	saltmarsh: CDFW-SSC, USFWS-BCC
Passeriformes	Curve-billed thrasher	*Toxostoma curvirostre*	
Passeriformes	Dark-eyed junco	*Junco hyemalis*	
Passeriformes	Eastern bluebird	*Sialia sialis*	
Passeriformes	Eastern meadowlark	*Sturnella magna*	
Passeriformes	European starling	*Sturnus vulgaris*	
Passeriformes	Field sparrow	*Spizella pusilla*	

Order	Common Name	*Scientific Name*	Special Status*
Passeriformes	Great-tailed grackle	*Quiscalus mexicanus*	
Passeriformes	House finch	*Carpodacus mexicanus*	
Passeriformes	House sparrow	*Passer domesticus*	
Passeriformes	Northern cardinal	*Cardinalis cardinalis*	CDFW-WL
Passeriformes	Pine siskin	*Carduelis pinus*	
Passeriformes	Prothonotary warbler	*Protonotaria citrea*	
Passeriformes	Red-winged blackbird	*Agelaius phoeniceus*	Kern: CDFW-SSC
Passeriformes	Rusty blackbird	*Euphagus carolinus*	
Passeriformes	Song sparrow	*Melospiza melodia*	
Passeriformes	Swamp sparrow	*Melospiza georgiana*	
Passeriformes	Tree swallow	*Tachycineta bicolor*	
Passeriformes	Vesper sparrow	*Pooecetes gramineus*	Oregon: CDFW-SSC, USFWS-BCC
Passeriformes	Western meadowlark	*Sturnella neglecta*	
Passeriformes	White-crowned sparrow	*Zonotrichia leucophrys*	
Passeriformes	White-throated sparrow	*Zonotrichia albicollis*	
Passeriformes	Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	CDFW-SSC
Pelecaniformes	Brown pelican	*Pelecanus occidentalis*	California: BLM-S, CDFW-FP, USFS-S
Piciformes	Northern flicker	*Colaptes auratus*	
Strigiformes	Barn owl	*Tyto alba*	
Strigiformes	Barred owl	*Strix varia*	
Strigiformes	Eastern screech owl	*Megascops asio*	
Strigiformes	Great horned owl	*Bubo virginianus*	
Strigiformes	Short-eared owl	*Asio flammeus*	CDFW-SSC
Strigiformes	Snowy owl	*Bubo scandiacus*	

*BLM-S: Bureau of Land Management- Sensitive; CDF-S: California Department of Forestry & Fire Protection- Sensitive; CDFW-FP: California Department of Fish & Wildlife- Fully Protected; CDFW-SSC: CDFW- Species of Special Concern; CDFW-WL: CDFW- Watch List; IUCN-NT: International Union for Conservation of Nature- Near Threatened; USFS-S: U.S. Forest Service- Sensitive; USFWS- BCC: U.S. Fish & Wildlife Service- Birds of Conservation Concern

Appendix B. Wildlife in which documented secondary poisoning by anticoagulant rodenticides occurred and their status (*see text section 2.2 for sources*).

Common Name	***Scientific Name***	**Special Status***
Bobcat	*Lynx rufus*	
European mink	*Mustela lutreola*	
Coyote	*Canis latrans*	
Red fox	*Vulpes vulpes*	ESA-C, CESA-TH, USFS-S
San Joaquin kit fox	*Vulpes macrotis mutica*	ESA-EN, CESA-TH
Gray fox	*Urocyon cinereoargenteus*	
Northern raccoon	*Procyon lotor*	
Polecat	*Mustela putorius*	
Stoat/ermine	*Mustela erminea*	
America badger	*Taxidea taxus*	CDFW-SSC
Striped skunk	*Mephitis mephitis*	
Moutain lion	*Puma concolor*	
Virginia opossum	*Didelphis virginiana*	
Heermann's kangaroo rat	*Dipodomys heermanni*	ESA-EN, CESA-EN, CDFW-FP
White-tailed deer	*Odocoileus virginianus*	
Common raven	*Corvus corax*	
American crow	*Corvus brachyrhynchos*	
Red-tailed hawk	*Buteo jamaicensis*	
Golden eagle	*Aquila chrysaetos*	BLM-S, CDF-S, CDFW-FP, CDFW-WL, USFWS-BCC
Bald eagle	*Haliaeetus leucocephalus*	CDFW-FP, CDF-S, USFS-S, USFWS-BCC
Red-shouldered hawk	*Buteo lineatus*	
Sharp-shinned hawk	*Accipiter striatus*	CDFW-WL
Cooper's hawk	*Accipiter cooperii*	CDFW-WL
American kestrel	*Faco sparverius*	
Peregrine falcon	*Falco peregrinus*	CDF-S, CDFW-FP, USFWS-BCC
Turkey vulture	*Cathartes aura*	
Barn owl	*Tyto alba*	

Snowy owl	*Bubo scandiacus*	
Screech owl	*Megascops* spp.	
Great-horned owl	*Bubo virginianus*	
Barred owl	*Strix varia*	
Northern spotted owl	*Strix occidentalis caurina*	ESA-TH, CESA-TH, CDF-S, CDFW-SSC, IUCN-NT
Long-eared owl	*Asio otus*	CDFW-SSC
Saw-whet owl	*Aegolius acadicus*	
Turkey	*Meleagris gallopavo*	

*BLM-S: Bureau of Land Management- Sensitive; CDF-S: California Department of Forestry & Fire Protection- Sensitive; CDFW-FP: California Department of Fish & Wildlife- Fully Protected; CDFW-SSC: CDFW- Species of Special Concern; CDFW-WL: CDFW- Watch List; CESA-TH: California Endangered Species Act- Threatened; CESA-EN: CESA Endangered; ESA-C: Endangered Species Act (Federal)- Candidate; ESA-EN: ESA- Endangered; ESA-TH: ESA- Threatened; IUCN-NT: International Union for Conservation of Nature- Near Threatened; USFS-S: U.S. Forest Service- Sensitive; USFWS- BCC: U.S. Fish & Wildlife Service- Birds of Conservation Concern