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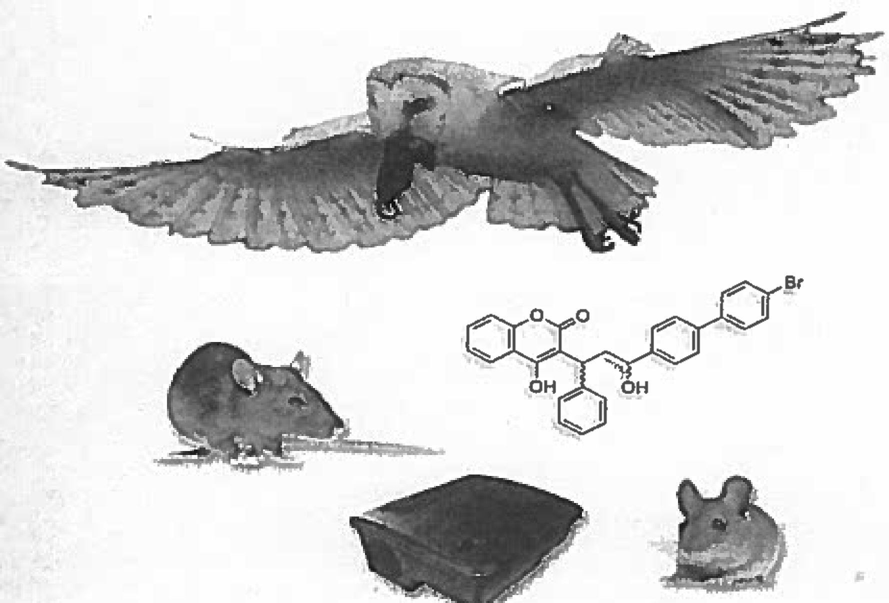
Editors

Anticoagulant Rodenticides and Wildlife



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Chapter 9
Ecological Factors Driving Uptake of Anticoagulant Rodenticides in Predators

Sofi Hindmarch and John E. Elliott

1 Introduction

Ecotoxicological risk is a function of exposure and toxicity (Newman 2014). Hence, when assessing the secondary exposure risk of anticoagulant rodenticides (ARs) to non-target predators and scavengers, we need to consider all potential sources of ARs, the types of ARs consumed, the amount and duration of exposure, and the possible range of residue burdens available in prey. Understanding the foraging behaviour of predators is also one of the key components in assessing the risk of exposure, and many variables come into play when examining such behaviours, including prey available versus prey taken, preferred hunting habitat, and selective versus opportunistic foraging. Prey selection may also differ as a function of landscape, season, prey relative abundance, age, sex, and breeding versus non-breeding status (Barbosa and Castellanos 2005).

Our objective was to conduct a review of the literature as a basis to better understand and synthesize the ecological factors driving AR exposure in predators. Specifically, we examined:

- How do landscapes and their environmental management influence the exposure of predators?
- What are the typical traits of exposed predators?
- What are the most common AR exposure pathways?

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2 Why are ARs a Problem for Predators?

As discussed in chapter 2, ARs are the primary compounds used to control rodent infestations worldwide (Corrigan 2001). First generation anticoagulant rodenticides (FGARs) came onto the market in the 1940s. Subsequently in the 1970s, at least in part due to the development of resistance to FGARs in some populations of Norway rats (*Rattus norvegicus*), second generation anticoagulant rodenticides (SGARs) were developed and registered for use. Today SGARs are the most commonly used ARs in both rural and urban settings (Corrigan 2001; Elliott et al. 2016a). FGARs continue to be widely used, and FGAR residues continue to be documented in predators. However, SGARs are more persistent, bioaccumulative, and toxic (PBT) to non-target species that have either consumed SGAR bait or ingested SGARs secondarily by consuming poisoned prey (Erickson and Urban 2004; US EPA 2011). The highly toxic nature of SGARs reduces the risk of resistance development in target species, and unlike FGARs, where the target species has to feed multiple times, a single feeding of SGARs is generally sufficient to kill the target species (Eason et al. 2002; Fisher et al. 2003). The AR mode of action is to disrupt the vitamin K cycle in the liver, which is necessary to produce blood clotting factors (Whitlon et al. 1978; Mackman et al. 2007). The half-life of blood clotting factors can range from 6–120 h, therefore, following AR exposure, several days can pass before coagulopathy symptoms are apparent (Lee et al. 2006). Consequently, target species can live for 4–13 days after the initial consumption of a lethal dose (Larsen 2003; US EPA 2011). During that time, they may continue to feed on available bait, consuming a total dose that is an order of magnitude above the level needed to cause mortality, and thus potentially transferring a higher and more toxic load to a subsequent predator (US EPA 2011). An analysis conducted as part of a US Environmental Protection Agency (EPA) risk assessment showed that rats could consume doses ranging from nine to 46 times the LD_{50} for the SGARs brodifacoum and difethialone before dying (US EPA 2011).

SGARs are also highly persistent: the half-lives ($t_{½}$) of brodifacoum and difethialone in the liver are 307 days and 29 days respectively, increasing the risk of toxic accumulation in the livers of predators that continuously feed on poisoned prey (Erickson and Urban 2004; Sage et al. 2008; US EPA 2011). Rodents poisoned by ARs do not always die in their burrows out of reach of predators and scavengers. Rather, they have been documented spending more time in open areas, staggering, and sitting motionless before death, all of which increases their susceptibility to predation (Cox and Smith 1992; Howald et al. 1999; Tuyttens and Stuyck 2002; Vyas et al. 2012).

The toxicity and persistence of SGARs combined with their increased usage worldwide is believed to have contributed to the widespread and increasing secondary contamination of non-target predatory birds and mammals (Table 9.1; Merson et al. 1984; Stone et al. 1999; Riley et al. 2007; Walker et al. 2008; Albert et al. 2010; Christensen et al. 2012; Gabriel et al. 2012; Sánchez-Barbudo et al. 2012; Hughes et al. 2013; Jacquot et al. 2013; Ruiz-Suárez et al. 2014; Shore et al. 2015).

Table 9.1 Examples of the influence of land-use on deployment of anticoagulant rodenticide baits, and targeted rodents and documented non-target primary exposure and secondary exposure of predators.

Landscape type	Agriculture		Urban-Wildland Interface	Urban
Bait placement	In & around the perimeter of buildings & fence lines	Mass applied in farm fields Game feeding stations	In & around the perimeter of buildings & fence lines	In & around the perimeter of buildings & fence lines
Target rodents	Rats, house & deer mice	Voles, gopher, prairie dog & ground squirrel	Rats, house, gopher, ground squirrel & deer mice	Rats, house mice, gopher & ground squirrel
Documented non-target exposure through direct consumption of bait	Greater white-toothed shrew (Geduhn et al. 2014) Sorex sp. (Geduhn et al. 2014; Townsend et al. 1995) bank vole (Geduhn et al. 2014; Brakes and Smith 2005; Townsend et al. 1995) Apodemus sp. (Geduhn et al. 2014) harvest mouse wood mouse (Brakes and Smith 2005; Tosh et al. 2012; Townsend et al. 1995) Microtus sp. (Geduhn et al. 2014; Elliott et al. 2014; Brakes and Smith 2005) slug (Elliott et al. 2014)	Iberian (Sánchez-Barbudo et al. 2012) & brown hare (Berny et al. 1997) European hedgehog (Sánchez-Barbudo et al. 2012; Dowding et al. 2010), dove sp. (Sánchez-Barbudo et al. 2012), red-legged partridge (Sánchez-Barbudo et al. 2012), grey heron (Sánchez-Barbudo et al. 2012; Berny et al. 1997), mallard (Sánchez-Barbudo et al. 2012; Berny et al. 1997), less black-backed gull (Sánchez-Barbudo et al. 2012), starling (Sánchez-Barbudo et al. 2012), great spotted cuckoo (Sánchez-Barbudo et al. 2012), Calandra lark (Sánchez-Barbudo et al. 2012), pigeon (Berny et al. 1997), swan (Berny et al. 1997), horseshoe whip snake (Sánchez-Barbudo et al. 2012), wild boar (Berny et al. 1997), rabbit (Berny et al. 1997), roe deer (Berny et al. 1997), badger (Berny et al. 1997), wild turkey (Ruder et al. 2011), American badger (Ruder et al. 2011), raccoon (Ruder et al. 2011)	Raccoon (Erickson and Urban 2004) coyote (Erickson and Urban 2004) Squirrel sp. (Erickson and Urban 2004) woodrat (Moriarty et al. 2012)	Raccoon (Erickson and Urban 2004) coyote (Erickson and Urban 2004; Way et al. 2006) Squirrel sp. (Erickson and Urban 2004)

(continued)

Table 9.1 (continued)

Predator	Agriculture		Urban-Wildland Interface	Urban
Barn owl Nocturnal In most cases vole specialist but does consume other prey when voles are not available	Britain (87% exposed to ARs in 2012) (Walker et al. 2014) (main prey 50% vole sp.) (Glue 1974) Canada (68% exposed to ARs 1988–2003) carcasses sampled from agricultural land and the agricultural – Urban interface (Albert et al. 2010). Main prey ~80% field voles (Hindmarch and Elliott 2015b).	Malaysia 1970–1980 (NA exposed) (main prey 98% rats) (Duckett 1991) Brodifacoum application in fields led to widespread decline and extirpation in certain regions (Duckett 1991).	Spain (68% exposed 2011–2013) (Lopez-Perea et al. 2015) Main prey mice, voles and shrews (Taylor 1994).	
Great-horned owl Usually nocturnal but also diurnal depending on season Generalist and opportunistic hunter, main prey predominantly rodents and lagomorphs	Western Canada (70% exposed to ARs 1988–2003) (Albert et al. 2010) carcasses sampled mainly from agricultural lands and the agricultural – urban interface. Main prey ~77% field voles (Hindmarch and Elliott 2014).		New York, USA (81% exposed to ARs 1998 – 2001) carcasses predominantly found in suburban and urban areas in addition to some rural areas (Stone et al. 2003). Massachusetts, USA (100% exposed to SGARs 2006 – 2012) carcasses found in urban, suburban and semi-rural communities (Murray 2011).	
Red kite Diurnal Mainly scavenger, but may also hunt rodents and birds	Britain (70% exposed to SGARS 1994–1999) (Shore et al. 2000)	France (44% exposed to ARs 1992 – 2002) application of bromadiolone in grass fields to reduce water vole populations (Berny and Gaillet 2008). Main prey 94% water voles (Coeurdassier et al. 2012).		
Polecat Nocturnal Main prey voles, shrew, mice, rats and rabbit. Including fish in central Europe.	Britain (31% exposed to SGARs 1990–1999) (Birks 1998) Main prey 65% rats when hunting farmyards during the winter (Shore et al. 2003).	France (15% exposed to ARs between 1991 – 1994) (Fournier-Chambrillon et al. 2004)		
Stoat, weasel and mink Mainly nocturnal Main prey rodents and rabbits	Denmark (stoats 97% exposed to ARs 1993 – 2007, weasels 95% exposed to ARs 1984–2008) (Elmeros et al. 2011). ARs mainly applied around farm structures but a small amount is also used around game feeding stations and tree plantations (Elmeros et al. 2011). Main prey rodents: Voles and *Apodemus sp.* (Elmeros 2006).	Britain (stoats 23%,weasels 30% exposed to ARs applied at game feeding stations 1996 – 1997) (McDonald et al. 1998) Main prey rabbits and rodents (*Apodemus sp., Clethrionomys glareolus, Microtus agrestis*) (McDonald et al. 1998) France (10% of European & American mink exposed ARs 1990 – 2002) application of bromadiolone in grass fields to reduce water vole populations (Fournier-Chambrillon et al. 2004). Main prey rodents (Fournier-Chambrillon et al. 2004).		
Foxes **Nocturnal** Generalist and opportunistic hunter main prey rodents. Scavenging more common when inhabiting urban environments	Red fox – Germany (60% exposed to ARs 2012–2013) carcasses mainly found in agricultural lands but also sampled suburban and urban areas. AR residue concentrations positively associated with livestock density and urbanization (Geduhn et al. 2015). Main prey water voles (Geduhn et al. 2015).	Red fox – France (100% exposed to ARs 1991–1994) application of bromadiolone in grass fields to reduce water vole populations (Berny et al. 1997). Main prey water voles (Berny et al. 1997).		San Joaquin kit fox - California, USA (74% exposed to ARs 1985–2009) carcasses predominantly found in suburban and urban areas in Bakersfield (Cypher et al. 2014). Consume predominantly gophers and ground squirrels but also insects and birds in urban environments (Cypher 2010).
Bobcat Nocturnal Main prey lagomorphs, rodents, deer and birds			USA, California (92% exposed to ARs 1997–2012) varied diet mostly lagomorphs and non-target rodents (Serieys et al. 2015).	

Non-target rodents along with smaller birds and invertebrate prey are also increasingly being documented as important vectors responsible for secondary AR exposure of predators (Brakes and Smith 2005; Elmeros et al. 2011; Sánchez-Barbudo et al. 2012; Elliott et al. 2014; Ruiz-Suárez et al. 2014; Serieys et al. 2015). However, ARs have become an integral component of rodent control, an industry that has been growing since the inception of SGARs. The lack of cost effective alternatives to suppress rodent populations, strict pest control standards, and in some instances the unaffordability of upgrading structures so that rodents are excluded, means ARs continue to be used extensively and are applied in many different locations ranging from industrial and commercial centres to agricultural landscapes and pristine islands.

3 How Does Landscape and Environmental Management Influence Exposure?

The ecological factors that drive the uptake of ARs in predators and the likelihood of exposure are context specific and depend on the landscape that the predators inhabit, the management of the habitats within the landscape, and prey availability. In general, predators are at greatest risk of secondary AR exposure when their primary prey and their main hunting habitats are being targeted with ARs (Lenton 1984; Hegdal and Colvin 1988; Birks 1998; Berny and Gaillet 2008; Jacquot et al. 2013). The following sections will discuss how the risk of AR exposure and poisoning in predators differs in agricultural versus urban landscapes and how these risks vary depending on the AR used, the amount of AR applied, application techniques, the intensity of application, the spatial scale of application, and timing.

3.1 AR Use in Agricultural Landscapes

Controlling pest populations in agricultural landscapes is a global challenge. Rodents, in particular, rats, cause extensive damage to crops and stored food, and damage estimates range from one fifth to one third of the world's food supply being consumed by rodents annually (Corrigan 2001). In Asia, crops lost to rodents annually could feed an estimated 200 million people per year (Singleton 2003). In Indonesia, rodents are the most destructive pre-harvest pests, responsible for the destruction of at least 15% of the annual production of rice (Geddes 1992). ARs are the primary pesticides used to control rodents in agricultural landscapes. In many countries, AR products are widely distributed and easily purchased, and some government agencies subsidize the purchase of these products by farmers and ranchers (Olea et al. 2009; Ruiz-Suárez et al. 2014). The methods of applying ARs can range from fastening bait stations to the outside perimeter of buildings, to placing AR bait in underground burrows in fields, to mass application of ARs in agricultural fields and orchards (Corrigan 2001; Rattner et al. 2014). The scale of AR field application varies from a couple of hectares to mass application of ARs on a regional scale (3000–10,000 km^2) to control rodent outbreaks (Jacquot et al. 2013; Baldwin et al. 2014).

3.2 AR Use in Agricultural Fields, and the Risk of AR Exposure of Predators

An obvious example of field application of ARs having a lethal impact on a predator comes from Malaysia, where in the early 1980s; brodifacoum was introduced as an alternative to warfarin with the goal of reducing rat damage to oil palm plantations (Lenton 1984; Duckett 1991). Prior to the commencement of AR application, barn owls (*Tyto alba*) were common, and nested in the plantations, where their diet was comprised of 98% Norway rats (Lenton 1984). Brodifacoum application led to a widespread decline and eventual extirpation of barn owls in certain regions of Malaysia (Duckett 1991). Demonstrating that SGAR usage can be particularly hazardous to a predator when, in their primary hunting habitat, ARs are used to target their primary prey (see also Hegdal and Colvin 1988; Birks 1998; Berny and Gaillet 2008; Jacquot et al. 2013). The loss of barn owls on the Malaysian plantations was particularly problematic as they were later shown to be effective at suppressing rat populations as part of Integrated Pest Management (IPM) programs (Duckett 1991).

The water vole (*Arvicola terrestris*) and common vole (*Microtus arvalis*) are widespread across Europe and Asia and mainly inhabit grasslands, marshlands, and crop fields. Both species exhibit large inter-annual fluctuations in population sizes. When their numbers are peaking, they can cause serious damage to agricultural lands by chewing the roots of plants, including young trees (Fichet-Calvet et al. 2001; Stenseth et al. 2003). The scale at which these outbreaks can occur varies widely, and they may even take place on larger regional scales (3000–10,000 km^2) (Singleton et al. 2010; Jacquot et al. 2013). In Central Europe, the SGAR bromadiolone and the FGAR chlorophacinone are typically applied in fields to suppress vole populations during outbreaks (Berny et al. 1997; Olea et al. 2009; Jacquot et al. 2013). Regulations govern the timing and density of the application to reduce the risk of secondary AR poisoning to non-target predators (Berny et al. 1997; Jacquot et al. 2013). In France, a licensed pest control operator can only apply bromadiolone, and the quantity of bromadiolone used and treatment locations must be reported to government agencies (Jacquot et al. 2013). Despite these regulations, AR exposures have been documented regularly in resident and migrant predators consuming voles, such as red kites (*Milvus milvus*), common buzzards (*Buteo buteo*), mustelid species (European and American mink (*Mustela lutreola and Neovison vison*), polecat (*Mustela putorius*), European otter (*Lutra lutra*)) and red foxes (*Vulpes vulpes*) (Berny et al. 1997; Fournier-Chambrillon et al. 2004; Berny and Gaillet 2008; Jacquot et al. 2013).

3.2.1 Mass AR Application and Primary Exposure of Non-Target Prey and their Predators

There is a significant potential for ARs to move into food chains when ARs are mass applied to agricultural fields (Fig. 9.1). In some cases, non-target species feeding on the bait may lead to exposure further up the food chain, both secondary and tertiary. For example, in central Europe one of the methods to control voles is by mass applying AR treated grain to grasslands and crop fields. Despite the seed being red or blue colored to deter birds from consuming the ARs, seed-eating birds, lagomorphs, and reptiles have been found with AR residues in their systems (Borst and Counotte 2002; Olea et al. 2009; Sánchez-Barbudo et al. 2012). While direct consumption of the AR treated grain is the most likely pathway, invertebrates have also been documented feeding on the bait, which opens up a whole new realm of potential exposure pathways, such as secondary exposure in insect eating birds and mammals (Eason and Spurr 1995; Howald et al. 1999; Dowding et al. 2010; Elliott et al. 2014). Tertiary exposure of predators is another potential pathway if predators consume songbirds that have consumed AR exposed insects. ARs are increasingly being documented in predators that primarily consume songbirds such as sparrow hawks (*Accipiter nisus*), merlins (*Falco columbarius*), coopers hawks (*Accipiter cooperii*) and peregrine falcons (*Falco peregrinus*) (Sánchez-Barbudo et al. 2012; Hughes et al. 2013; Ruiz-Suárez et al. 2014; Walker et al. 2014).

Invertebrates are considered to be less sensitive to ARs due to their different circulatory systems. This may allow them to consume larger quantities of ARs without lethal effects. The main exposure route for invertebrates is most likely from feeding on AR bait, including that found inside bait stations (Spurr and Drew 1999; Dunlevy et al. 2000; Johnston et al. 2005). However, carrion insects have also been found with residues of ARs, which would suggest other potential exposure routes to invertebrates such as feeding on dead poisoned rodents, small mammal feces, or even soil-bound AR residues (Rattner et al. 2014; Elliott et al. 2014).

Most of the research and field experiments examining field applications of ARs and their effect on predators come from France. The following section, which examines how different AR application techniques affects non-target predators is therefore based on French studies, where in some areas, voles can cause substantial damage to pasture lands resulting in significant economic losses to regional cheese production.

In France, water vole outbreaks are controlled over large areas (up to 60,000 ha annually) using primarily the SGAR, bromadiolone (Giraudoux et al. 2006; Delattre and Giraudoux 2009). The bait is mixed to a standard bait concentration of 50 mg of bromadiolone per kg of bait (Berny et al. 1997). The bait is either placed above ground or in man-made burrows called galleries which are approximately 15 cm underground, and the amount of bait applied cannot exceed 20 kg/ha (Sage et al. 2008).

Several studies have documented a correlation between bromadiolone field usage, exposure levels, and poisoning of predators (Berny et al. 1997; Berny and Gaillet 2008; Jacquot et al. 2013; Montaz et al. 2014). Berny and Gaillet (2008)

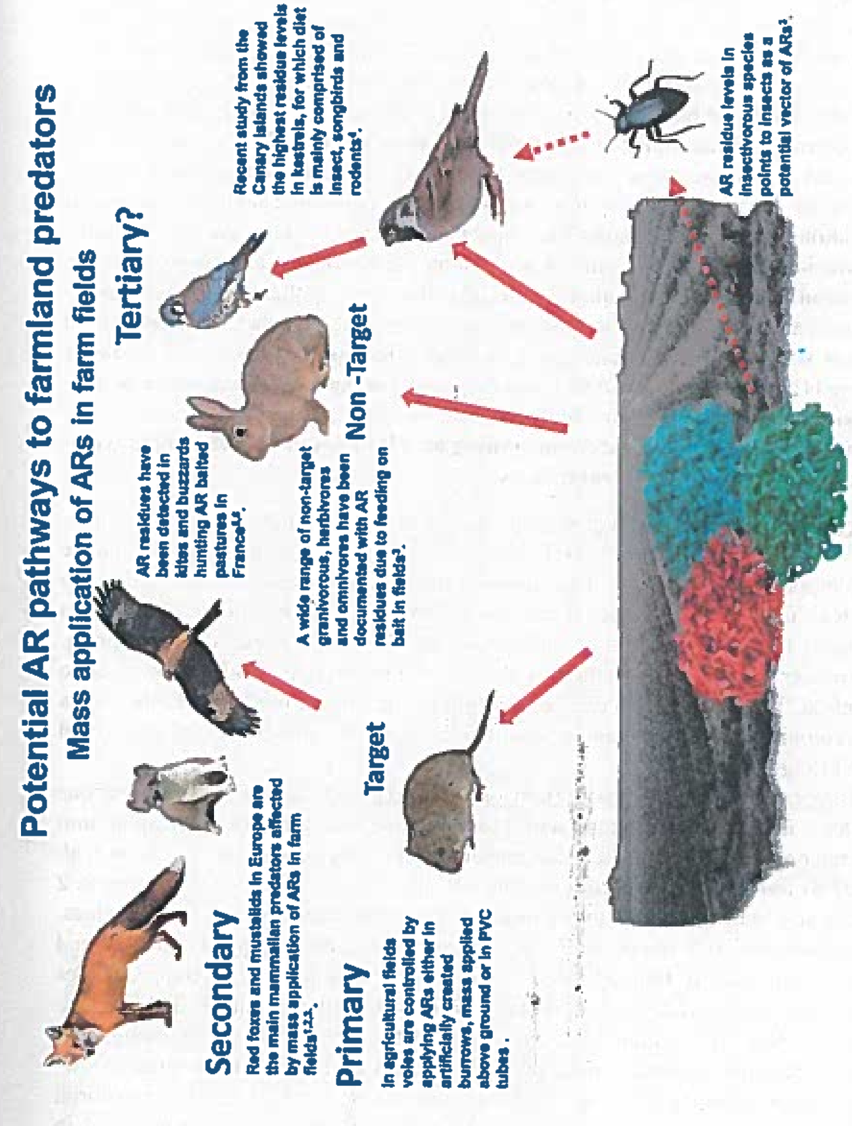

Fig. 9.1 Overview of potential anticoagulant rodenticide (AR) pathways to predators that inhabit and hunt farmland, when ARs are mass applied in agricultural fields. The dotted arrows indicate potential pathways, but more documentation is needed to verify to extent to which these prey species are vectors of ARs in the food-chain

found a positive association between the area of grassland treated with bromadiolone and the proportion of red kites found poisoned. As a result, they proposed that bromadiolone be used more sparsely, targeting only individual fields in order to avoid the risk of entire populations of voles becoming poisoned and in turn increasing the risk to kites and other predators.

The number of AR poisoned predators has been correlated with the seasonal application of ARs in grasslands, which is typically done in the autumn and winter (Berny et al. 1997; Berny and Gaillet 2008). For example, Montaz et al. (2014) found that kites and buzzards were particularly vulnerable to scavenging poisoned voles during their autumnal post-nuptial migration through France. Resident red foxes feed almost exclusively on voles during vole outbreaks. Jacquot et al. (2013) determined that red fox abundance was negatively correlated with the amount of bromadiolone used in grasslands. The impact was greatest in a large area (>1000 km^2) that was intensively treated with bromadiolone, and a partial recovery of the fox population was not observed until 2 years after the initial application. Large scale (> 1000 km) and high density AR application to grasslands and other crops provides a constant source of contaminated prey, directly affecting the number of predators poisoned (Berny and Gaillet 2008); and potentially suppressing local predator populations (Lenton 1984; Jacquot et al. 2013).

The likelihood of a predator consuming an AR exposed vole after field treatment is determined by several factors:

- Decomposition rates of bait in tunnels, which gives an indication of the time-period over which the bait is available to voles. Sage et al. (2007) showed that the decomposition rate varied depending on the soil type, season, and whether the voles changed the bait. They found that the half-life of bromadiolone in galleries ranged from 3.0 to 5.1 days in autumn and from 5.4 to 6.2 days in spring. However, if voles hoarded the bait, the decomposition rate could be slowed up to tenfold. This can lead to a delayed exposure of additional predators if other voles re-colonized parcels of land containing cached ARs and consumed the stored bait (Sage et al. 2007).
- Temporal trends in the proportion of voles exposed, and liver concentrations after a field has been treated with bromadiolone bait can provide insights into when predators are most likely to consume voles with toxic AR levels. Sage et al. (2008) found that all water voles (the AR target rodent) trapped underground 2 days after gallery application of bromadiolone had detectable AR liver residues. Additionally, 41% of common voles (a non-target rodent) trapped above ground were exposed to bromadiolone. Liver concentrations in live-trapped voles increased daily before peaking at day 6–7 for water voles and at day 3–4 for common voles, after which time the concentrations declined. Even though the concentrations dropped considerably 30–40 days post treatment, a repeated low-dose bromadiolone exposure could accumulate to toxic levels in a territorial predator hunting in the treated area. Low-level exposure was documented in voles until the study terminated, 135 days post treatment (Sage et al. 2008).

- Vole density determines the number of AR exposed voles available to predators within an AR treated field. Grolleau et al. (1989) and Sage et al. (2008) documented maximum rodent mortality 5–10 days after field application. Very few voles were trapped on treated plots beyond day 20, consequently reducing the risk of a predator catching an AR exposed vole beyond that time (Sage et al. 2008). However, the risk to predators varies based on vole recolonization rates, and on how much of the cached bait persists and is available to recolonizing animals (Sage et al. 2007, 2008). On a similar note, Olea et al. (2009) argued that the application of ARs during a vole outbreak when the population is peaking would have no effect on their overall abundance, as the population would already be experiencing a natural population collapse. Application of ARs at that time could lead to tons of toxic grain remaining accessible to non-target species over a large area.
- The voles that die above ground after the mass application of ARs on agricultural lands are a source of ARs to scavenging predators. Sage et al. (2008) noted the above ground mortality of voles 4–12 days post treatment, which would then be available to surface scavengers. Montaz et al. (2014) documented high scavenging rates after bromadiolone field application, and dead voles were on average scavenged within 0.5–1.5 days. As a result, they recommended daily collection of vole carcasses, and avoiding AR treatment of fields when high predator densities have been recorded. Predators may feed preferentially on poisoned animals, increasing their risk of AR poisoning as the reduced escape response of AR exposed moribund rodents also increased their vulnerability to predation (Cox and Smith 1992; Brakes and Smith 2005).
- Different foraging tactics influence the overall risk of AR exposure to predators.
- Raptors typically catch voles and other mammals above ground and underground AR exposed and moribund voles are inaccessible. Predatory mammals such as foxes and mustelids have an increased risk of AR exposure as they catch voles above ground as well as underground in burrows, where the majority of AR-exposed rodents die (King and Powell 2007; Sage et al. 2008; Jacquot et al. 2013).

3.2.2 AR use in Agricultural Fields and Risk of Exposure to Predators in North America

In North America, farmers are allowed to apply the FGARs, diphacinone and chlorophacinone, in agricultural fields, vineyards, and orchards to control outbreaks of black rats (*Rattus rattus*), field voles (*Microtus townsendi*), gophers (*Geomyidae sp.*), prairie dogs (*Cynomys sp.*), deer mice (*Peromyscus maniculatus*) and ground squirrels (*Sciuridae sp.*) (Salmon and Dochtermann 2006; Ruder et al. 2011; Baldwin et al. 2014). These species are key prey items for many predators in agricultural landscapes, such as owls, hawks, great blue herons (*Ardea herodias*), coyotes (*Canis latrans*), and fossorial snakes such as gopher snakes (*Pituophis catenifer deserticola*) (Wells and Bekoff 1982; Marti and Kochert 1995; Marks et al. 1994; Houston et al. 1998; Wiggins et al. 2006; Vennesland and Butler 2011; Bishop et al. 2016).

In the prairie region, the colonial black-tailed prairie dog (*Cynomys ludovicianus*) is considered a keystone species due its unique influence on plant and animal communities in grassland ecosystems, despite also being considered a pest by many cattle farmers (Kotliar et al. 1999). In 2009 the US Environmental Protection Agency (US EPA) allowed for the use of the FGAR, Rozol® (0.005% active ingredient chlorophacinone), to control prairie dog colonies in several US states (US EPA 2009). There were concerns about the approval of Rozol® in this instance, as black-tailed prairie dog colonies are a valuable food source for a wide range of avian and mammalian predators, including the endangered black-footed ferret (*Mustela nigripes*) (Plumpton and Andersen 1997; Kotliar et al. 1999; US Fish and Wildlife Service 2016).

Vyas et al. (2012) found 25 dead or moribund mammals, including non-targets, above ground after one targeted Rozol® poisoning of a 17.3 ha black tailed prairie dog colony. Scavenging was documented during AR application, demonstrating the transfer of chlorophacinone to non-target predators under current application practices. In a subsequent study, Vyas et al. (2013) found songbirds to also be at risk from Rozol® applications, and transferring it up the food chain. A recent analysis of AR residues in livers of the American badger (*Taxidea taxus*) across parts of its range in southern British Columbia, Canada, found detectable residues of chlorophacinone in livers, with individual animals having residue concentrations as high as 0.450 µg/g w.w (Elliott et al. 2016b). Rozol® is available for use in Canada to control rodents in orchards, vineyards and on ranchland.

3.2.3 AR Use Around Farm Structures

Baiting farm buildings with ARs is a common practice in industrialized countries, and AR based rodent control programs are often part of government food safety standards, or requirements of the food processing industry (Tosh et al. 2011; Canada GAP 2016; CRRU 2016). SGARs are the most common AR used, and in some cases, baiting is long-term or even permanent inside or around the perimeter of farm buildings (Elmeros et al. 2011; Tosh et al. 2011; Hughes et al. 2013; Canada GAP 2016). Such practices can provide a constant source of ARs to both the target rodents, commonly rats and house mice (*Mus musculus*), and non-target species small enough to access bait stations (Fig. 9.2). Small granivorous birds and non-target rodents such as deer mice (*Peromyscus maniculatus*), wood mice (*Apodemus sylvaticus*), voles and shrews have been documented entering secure bait stations, and feeding on ARs, or have been found with residues of ARs in their systems (Townsend et al. 1995; Brakes and Smith 2005; Tosh et al. 2012; Elliott et al. 2014; Geduhn et al. 2014). The degree to which non-target prey, such as birds and small mammals, act as vectors of ARs depends partly on their mobility and home-range

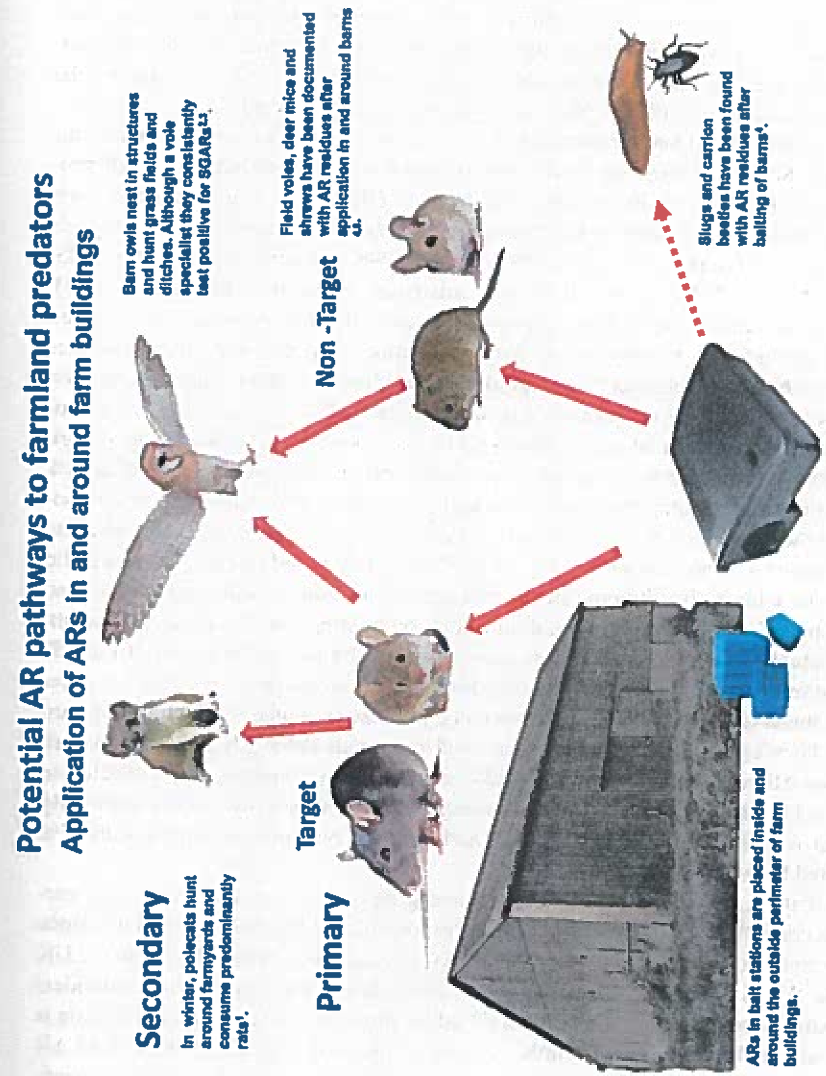

Fig. 9.2 Overview of potential anticoagulant rodenticide (AR) pathways to predators that inhabit and hunt farmland, when ARs are applied in and around farm buildings. The dotted arrows indicate potential pathways, but more documentation is needed to verify to extent to which these prey species are vectors of ARs in the food-chain

size. Wood mice in particular seem to be more mobile than other non-target rodents and may spread AR away from farm buildings (Townsend et al. 1995; Tosh et al. 2012). Despite this, the greatest concentrations of ARs have been measured in mice closest to baited buildings (Geduhn et al. 2014). Temporal trends show that the proportion of non-target small mammals exposed to ARs was significantly higher after baiting than during baiting (Geduhn et al. 2014). Overall concentrations in exposed individuals, however, were lower after baiting than during, which suggests the poisoned non-target small mammals died shortly after baiting ended, similar to the findings of Sage et al. (2008) for water voles in agricultural fields.

Many raptors and mammalian predators hunt both native and commensal rodents on farms. Several of these species have been found with varying degrees of AR residues in their livers, including kites, eagle owls (*Bubo bubo*), great-horned owls (*Bubo virginianus*), barn owls, foxes, stoats (*Mustela erminea*), least weasels (*Mustela nivalis*), polecats, and coyotes (Birks 1998; Ntampakis and Carter 2005; Albert et al. 2010; Elmeros et al. 2011; Christensen et al. 2012; Riley et al. 2007). Barn owls are unique agricultural raptors in the sense that they have not only adapted to hunting on farms, but also nest and roost in barns, silos, and other farm structures (Taylor 1994; Marti et al. 2005). Hegdal and Blaskiewicz (1984) conducted the first large scale field study on barn owls in southwestern New Jersey, USA, and evaluated the potential risk of secondary brodifacoum poisoning to radio-tagged barn owls nesting on farms where this AR had been applied. They concluded that the risk of secondary poisoning was low, as rats and house mice were only a minor component of the barn owls' diet (6.1%) and of six tested barn owl carcasses, only one had trace residues of brodifacoum. More importantly, they found that the barn owls did not spend a lot of time hunting in close proximity to farm buildings where bait had been applied, but instead focused their efforts on hunting voles in grass and marshland habitats. Geduhn et al. (2016) also assessed the risk to barn owls from AR baiting on farms, but they focused on AR exposure in non-target prey such as deer mice, shrews, and voles and the importance of these prey items in the diet of barn owls. They concluded that non-target small mammals drive AR exposure in barn owls as AR residues were found in 13% of non-target mammal prey collected in barn owl nests. These small mammals were key prey for barn owls either seasonally or year-round (Geduhn et al. 2016). In addition, AR residues were present in 55% of tested barn owl carcasses.

In Britain, red kites are often found scavenging on farms and in towns. Prey carcasses contaminated with ARs have been responsible for the death of red kites since their reintroduction, and SGAR poisoned red kites are reported each year in the UK by the Wildlife Incident Investigation Scheme [http://www.hse.gov.uk/pesticides/topics/reducing-environmental-impact/wildlife.htm]. As a result, their population is still vulnerable and small in number (Carter et al. 2003). To assess the risk of AR poisoning in red kites, Ntampakis and Carter (2005) conducted a feeding experiment to determine what sized prey red kites preferred, and how readily they flew down to take carcasses close to buildings, where ARs are typically applied. They found that red kites preferred preying on small to medium sized rat carcasses that could be carried to their nest sites, and that they showed no hesitation in picking up carcasses next to farm buildings.

Predators that depend on farm rats seasonally are also at risk of AR exposure (Galeotti et al. 1991; Birks 1998). This is especially true if the predator consumes more rats in the late fall or early winter, as this is also the time when rats tend to seek shelter indoors due to cooler outdoor temperatures, and are consequently exposed to AR bait placed in and around structures (Huson and Rennison 1981; Feng and Himsworth 2014). In addition, farms and households that only use ARs seasonally will typically engage in rodent control in the fall and winter as this is when they see rats, or evidence of their presence (Feng and Himsworth 2014). In England, the diet of polecats during the winter consists primarily of Norway rats (65%, Birks 1998). Further, farmyards, half of which were baited with SGARs, were the preferred hunting sites for the radio-tagged polecats that were visiting to feed on rat populations (Birks 1998). The risk of exposure in polecats was evident from residue data, as Shore et al. (2003) found that 31% of polecats tested between 1992 and 1999 contained SGAR residues.

The practice of permanently placing SGARs around farm structures, thereby providing a constant source of ARs to rats and house mice has led to AR resistance in rats (Cowan et al. 1995; Endepols et al. 2012; Buckle 2013). Cowan et al. (1995) found that rats resistant to warfarin, difenacoum and bromadiolone consumed significantly more difenacoum and bromadiolone compared to AR-susceptible rats on farms in England. The capacity of resistant rats to consume more ARs without any adverse effects equates to a higher AR load potentially being transferred to predators. 67% of surviving warfarin resistant rats were found with >4 mg/kg body weight of ARs in their systems. However, Atterby et al. (2005) conducted a controlled feeding study comparing AR-susceptible and AR-resistant rats, and found that the highly resistant strain of rats, which had been fed ARs for 20 days, had average residue levels of 0.74 mg/kg body weight, which was unlikely to present a significant risk to predators. Regardless, the current practice of permanently baiting farms with ARs, and increasing the likelihood of resistance in rats needs addressing as it runs contrary to efforts to reduce the risk of secondary poisoning of non-targets via implementation of more sustainable IPM practices (CRRU 2016).

3.3 *Urban AR Use and the Risk of Secondary AR Exposure in Predators*

Cities, due to their dense human populations, attract the archetypical commensal pest species, the Norway rat (Feng and Himsworth 2014; Walsh 2014). Where there is food, shelter, and warmth, there are rodents, and it has even been suggested that rats are obligate pests since they appear to require humans to sustain their populations (Aplin et al. 2003). Big cities like New York City constantly struggle with rat infestations, and there are endless accounts of rats in subways, parks, cars, and homes (Walsh 2014). In 2014, the problem became so severe that the city declared war on the rats, targeting "rat reservoirs" in order to reduce their numbers, along with other more innovative initiatives such as the Rodent Control Academy and mass sterilization of female rats (NY Magazine 2015; Rodent Academy 2016).

Our negative attitudes towards rats and mice results from their status as disease vectors as well as the extensive damage they can do to structures and food (Morzillo and Mertig 2011; Rattner et al. 2014; Himsworth et al. 2015). Significant costs are associated with urban rodent control efforts to reduce the risk of damage caused by rodents (Corrigan 2001; Rattner et al. 2014). AR baits, primarily SGARs, are the main tools used to combat rodents in cities (Corrigan 2001). The AR baiting regime is often permanent, and bait stations are checked and refilled on regular intervals. The density of buildings with permanent AR bait in an urban environment provides a constant source of ARs to target rodents and potentially to non–target small mammals and birds.

Raptors and mammalian predators are increasingly being documented in urban environments and along the urban-wildland interface (Gehrt et al. 2010; Hindmarch and Elliott 2015a, b). This is partially a consequence of urban development encroaching on native habitat, leaving predators with little choice but to adapt to their new suburban environment (McKinney 2002). In addition, reduced human persecution over the last century has meant that predators are able to survive in closer proximity to humans (Kitchen et al. 2000; Bildstein 2008). Predators that successfully inhabit urban landscapes are typically generalists that take advantage of whatever available prey or carrion are present in this environment (Sorace and Gustin 2009). Hence, many urban predators opportunistically consume target and non-target rodents that may have been exposed to ARs (Fig. 9.3; Morey et al. 2007; Hindmarch and Elliott 2014; Hindmarch and Elliott 2015a, b; White et al. 2015). Consequently, predators such as red-tailed hawks (*Buteo jamaicensis*), owls, coyotes, bobcats (*Lynx rufus*), and foxes living in urban environments have been exposed, sometimes fatally, to multiple ARs (Stone et al. 1999; Riley et al. 2003; Stone et al. 2003; Albert et al. 2010; Murray 2011; Lopez-Perea et al. 2015; Nogeire et al. 2015; Poessel et al. 2015; Serieys et al. 2015; Justice-Allen and Loyd 2017).

In Spain, AR exposure in predators was positively associated with human population densities, and this was partly explained by the extensive use of ARs against commensal rodents in urban environments (Lopez-Perea et al. 2015). On the Pacific coast of the USA and Canada, roof rats (*Rattus rattus*) and Norway rats have been shown to be important prey for barred owls, to a lesser degree for great-horned owls, and a subsidiary prey item for barn owls living in urban parks and green-spaces (Lambert 1981; Hindmarch and Elliott 2014, 2015a, b). The consumption of rats by urban owls is likely reflected in the residue data from the region, as 98% (n = 91) of the barred owls, 79% (n = 90) of the great-horned owls and 76.5% (n = 85) of the barn owls tested positive for SGARs in southwestern British Columbia between 1988 and 2010 (Albert et al. 2010; J. Elliott unpub. data, 2017). The barred owls in particular were prone to AR exposure and acute AR poisoning. Based on post-mortem examinations between 1993 and 2003 in

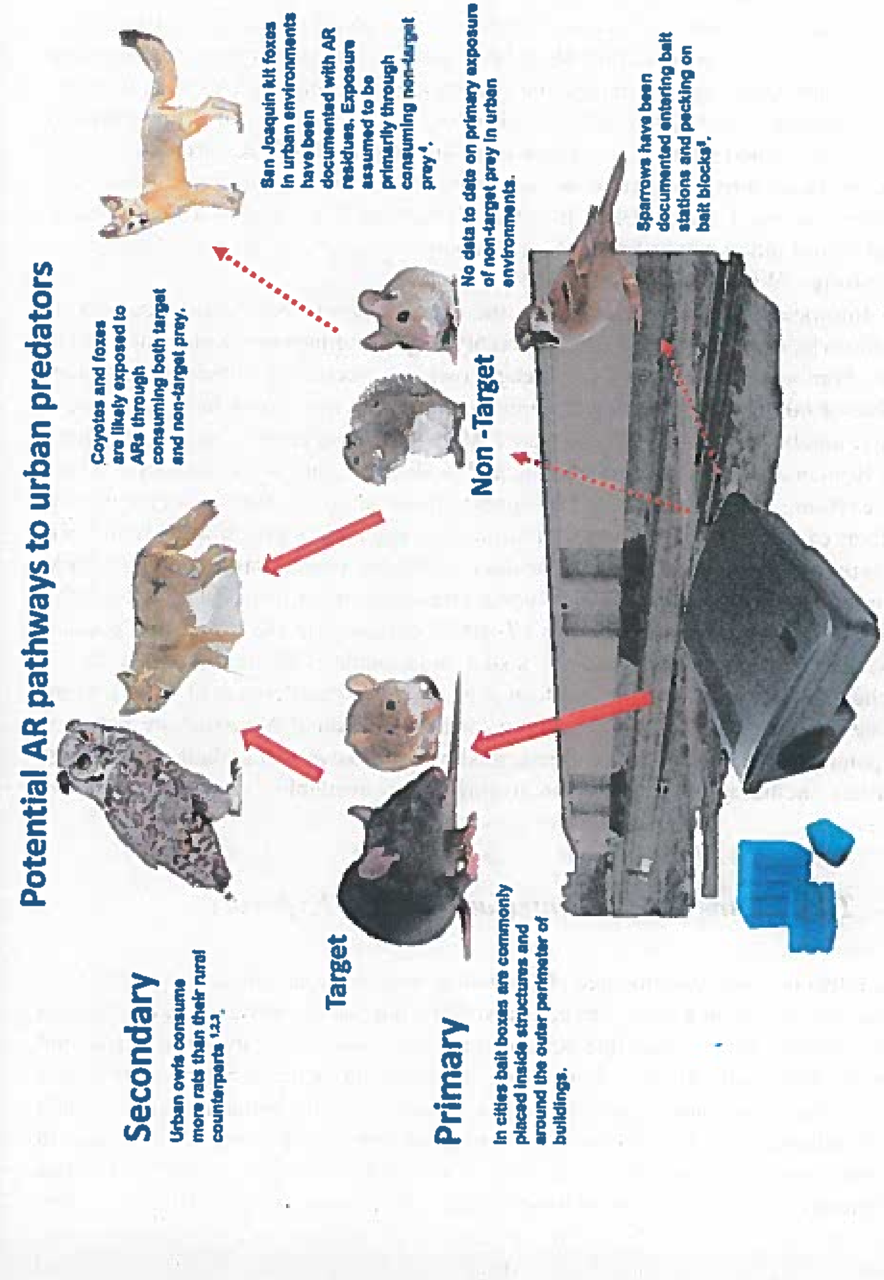

Fig. 9.3 Overview of potential anticoagulant rodenticide (AR) pathways to predators that inhabit and hunt urban areas. The dotted arrows indicate potential

southwestern British Columbia, they had the highest proportion of deaths attributed to AR poisoning (12%, n = 25) (Albert et al. 2010).

Great-horned owls and red-tailed hawks are both cosmopolitan species that have established themselves in urban environments across North America. Due to their similar size, diet, and habitat preferences, red-tailed hawks are commonly referred to as the diurnal counterparts of great-horned owls (Marti and Kochert 1995). The similarities are also reflected in AR residue data which shows that exposure and concentration levels are similar for the two species in eastern USA (Murray 2011: great-horned owl 100%, red-tailed hawk 89%; Stansley et al. 2014: great-horned owl 82%, red-tailed hawk 81%). There is no diet data available for urban red-tailed hawks in the eastern USA, but observations suggest that rats are a part of their diet (J. Morrison Pers. Comm. 2014). In Reno, Nevada, an ongoing diet study of breeding urban red-tailed hawks found that rats comprised ~10% of the prey delivered to the nestlings (White et al. 2015) .

In Europe and Asia, eagle owls are the counterparts to North American great-horned owls, although they are considerably larger and heavier (König and Weick 2008). Similar to great-horned owls, eagle owls are nocturnal, generalist predators capable of taking a large variety of prey, including carrion. They have been documented inhabiting urban and suburban centres in diverse regions such as northern Italy, Romania, South Korea and Spain, and in all cases, rats, were a consistent food source (König and Weick 2008). The opportunistic hunting behaviour of eagle owls and their capacity to inhabit urban environments where they prey upon rats and carrion is reflected in their SGAR residue data, which has been monitored in Denmark, Spain, and Norway. Average SGAR concentrations ranged from 0.1 to 0.2 ug/g in birds with detectable residues, with 57–100% exposure in the monitored populations, which ranged from 7 to 10 tested individuals (Christensen et al. 2012; Sánchez-Barbudo et al. 2012; Langford et al. 2013; Lopez-Perea et al. 2015). More residue data on eagle owls is needed to further understand AR exposure patterns, and potential population levels effects, as they are elusive and endangered in most countries, including those where monitoring data is available.

3.4 *The Urban-Wildland Interface and AR Exposure*

Edge effect is a well-documented phenomenon in biology, and the ecological consequences of an edge in a landscape can present itself in various different ways (Murcia 1995). For example, in the edge between urban and wildland habitats, the flora and fauna is often quite different from either the wildland or the urban areas that lead into it. In some instances, species diversity is higher in this "boundary zone", which can be inhabited by both native and non-native species. The 'peri-urban peak' in species diversity is a pattern that has been documented for birds, lizards, butterflies and plants (Sewell and Catterall 1998; Blair 2001; Germaine and Wakeling 2001; Godefroid and Koedam 2003). In the urban-wildland interface in Buenos Aires, Argentina, higher prey diversity, resulting from the presence of both native and

non-native rodents has been documented for barn owls inhabiting this area, versus for those living in purely agricultural or urban landscapes (Teta et al. 2012).

Morzillo and Schwartz (2011) found that landowners living in southern California inhabiting low-density residential housing in close proximity to wildlands were more likely to use ARs than landowners living in high-density neighborhoods. AR baiting in this boundary zone can cause increased rates of exposure of non-target native rodents and smaller songbirds that inhabit this zone, posing a substantial threat to mammalian and avian predators living or hunting along the urban-wildland boundary (Fig. 9.4). In California, the endangered Californian San Joaquin kit fox (*Vulpes macrotis mutica*) has been exposed to and poisoned by ARs, either through direct consumption or consumption of AR-target and non-target prey (Cypher et al. 2014; Nogeire et al. 2015). Nogeire et al. (2015) modeled the potential population wide effects of AR exposure on the kit fox and found that exposure in developed areas with low housing density accounted for 70% of the AR exposure. They concluded, based on their model, that kit fox exposure in areas of low-density housing near their natural habitat has the most negative effect on the population. Riley et al. (2003, 2007) and Serieys et al. (2015) found AR residues in radio-collared coyotes, bobcats, and mountain lions (*Puma concolor*) hunting within the urban-wildland interface in California. Rabbits (*Sylvilagus audubonii*), woodrats (*Neotoma spp.*), botta's pocket gophers (*Thomomysb ottae*), and California ground squirrels (*Spermo-philusb eecheyi*) were key prey items for these predators in this region (Riley et al. 2010; Gehrt and Riley 2010). Suggesting that AR exposure in the aforementioned smaller prey is responsible for the uptake of ARs in the larger carnivores (Riley et al. 2007, Moriarty et al. 2012). Riley et al. (2007) also discuss the possibility of a tertiary exposure route for mountain lions via the consumption of AR-laden coyotes, as the two AR poisoned mountain lions were documented to have consumed coyotes in the month before they died. AR toxicity has previously been shown to be a cause of death in coyotes in the region (Riley et al. 2003) and in other urban areas of the United States, both through direct consumption and secondary AR exposure (Way et al. 2006; Poessel et al. 2015).

Hindmarch et al. (2017) conducted a radio telemetry study of barn owls that inhabited or hunted the urban – agriculture boundary in southwestern British Columbia, Canada, and found that rats were only a minor component of the overall barn owls' diet (4%, range 0–11%). However, due to the highly fragmented landscape, barn owls often hunted in close proximity to AR-baited buildings where they could potentially predate upon AR exposed rats and non-target prey. In Lake Havasu City, Arizona, four dead burrowing owls were found with residues of brodifacoum in their systems, and poisoning was the cause of death for three of the owls (Justice-Allen and Loyd 2017, range 0.077–0.497 mg/kg). Desert pocket mice (*Chaetodipus penicillatus*) and small rat specimens were likely the vectors for this exposure based on previous research on the burrowing owls' diet, but further verification is needed (Poulin et al. 2011 and references therein). Justice-Allen and Loyd (2017) are continuing their research on suburban burrowing owls, and future research will include a focus on identifying possible AR vectors and the rate of AR exposure.

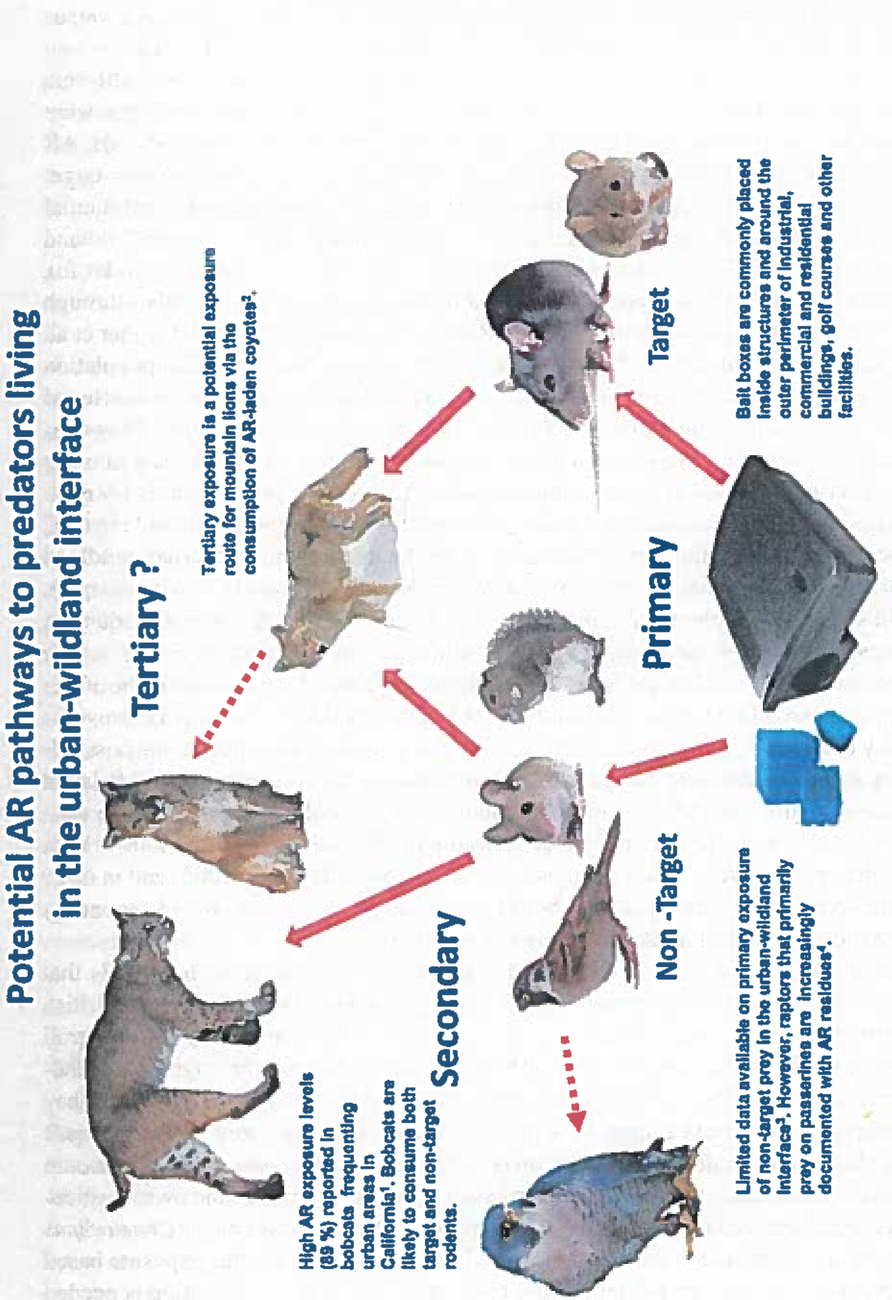

Fig. 9.4 Overview of potential anticoagulant rodenticide (AR) pathways to predators that inhabit and hunt the urban-wildland interface. The dotted arrows indicate potential pathways, but more documentation is needed to verify to extent to which these prey species are vectors of ARs in the food-chain

3.5 *AR Exposure in Wildlands*

In contrast to non-migratory raptors that inhabit human intensive landscapes, migra tory raptors and raptors that inhabit environments with a low human presence, suc as parks and other conservation and protected lands, have typically been found to b less exposed to ARs (Christensen et al. 2012; Langford et al. 2013; Lopez-Pere et al. 2015). In Denmark, of the migratory raptors or raptors inhabiting relativel pristine environments, none had AR levels >150 ng/g in their livers. Monitored spe cies included rough-legged buzzards (*Buteo lagopus*), marsh harriers (*Circus aeru ginosus*), long-eared owls (*Asio otus*) and short-eared owls (*Asio flammeus* (Christensen et al. 2012). Similarly in Spain, AR residues were significantly highe in a resident population of scops owls (*Otus scops*), compared to a population whicl migrated in the winter to areas in Africa with relatively low human population den sities, and where AR use was expected to be minimal (Lopez-Perea et al. 2015).

In Norway, SGAR concentrations in all raptors were found to be lower than ir those in Denmark (Christensen et al. 2012; Langford et al. 2013). This is likely a result of Norway's lower human population density and its fewer urban and agricultural areas compared to Denmark, which thus equates to less AR usage on the landscape. The only exposed species were eagle owls and golden eagles (*Aquila chrysaetos*), and no residues were detected in raptors such as peregrine falcons, osprey (*Pandion haliaetus*), and gyrfalcons (*Falco rusticolus*), whose diets were mainly comprised of birds and fish, and who spend less time in human populated areas in Norway.

An exception to the above trend has been found when AR is used on large game estates to discourage rodents from eating food intended for game birds. AR bait is placed around feeding stations for game birds and these feeding stations are typically located in semi-open wooded areas (McDonald et al. 1998; Elmeros et al. 2011). A wide array of non-target rodents have been observed feeding on bait at these stations (Wood and Phillipson 1977; McDonald et al. 1998; Brakes and Smith 2005). Brakes and Smith (2005) observed the caching of bait by wood mice and bank voles, which could prolong the exposure period in these non-target rodents as well as potentially expose individuals that did not visit the bait stations. In Britain and Denmark, this exposure route has been identified as one of the potential sources of AR exposed prey for stoats and weasels, as their diet is mainly comprised of rabbits, voles, native mice, shrews and birds (McDonald et al. 1998; Elmeros 2006; King and Powell 2007; Elmeros et al. 2011).

Recently in California, a surprising exposure route was uncovered as Gabriel et al. (2012) found that 80% (n = 58) of deceased fishers (*Martes pennanti*), rare forest carnivores, had been exposed to SGARs. The exposure, which in some cases was fatal (n = 4), appears to be linked to illegal marijuana cultivation in forests situated on community and public lands. The growers liberally spread SGARs directly on the ground around young marijuana plants, in order to kill rodents attempting to eat the plants, and thus placing non-target prey at risk of exposure. Unfortunately, the consequence of illegal, non-compliant use of these SGARs raises concerns not only for the survival of the threatened fisher, but also for other at-risk forest species that are highly dependent on these forests, such as the spotted owl (*Strix occidentalis*).

3.6 *Island Pest Eradication Programs and Risk of Secondary AR Exposure*

Programs to eradicate non-native mammals on islands, and the permanent removal of invasive pests within a well-defined geographic boundary are very different and unique scenarios compared to pest control in agricultural and urban landscapes (Howald et al. 2015). The ultimate goal of these initiatives is to eradicate the pest species from a given area to allow the successful re-establishment of native fauna and flora (Russell and Holmes 2015). These initiatives take place on a case-by-case basis and are carefully planned and executed. ARs are typically mass applied in large volumes, usually only once, and often by helicopter, in order to completely eradicate the pest species. These programs are quite successful in re-establishing native fauna and flora (Howald et al. 2007; Le Corre et al. 2015; Russell and Holmes 2015). However, the mass application of ARs often equates to the entire food chain being temporarily exposed to ARs, including groups of species such as marine arthropods and insects and snails (Pain et al. 2000; Masuda et al. 2015; Pitt et al. 2015). There have also been some documented cases of, bald eagles (*Haliaeetus leucocephalus*), Australasian harriers (*Circus approximans*), moreporks (*Ninox novaeseelandiae*), and short-tailed bats (*Mystacina tuberculata*) experiencing temporary population declines due to AR poisoning from efforts to eliminate island pests (Howald et al. 1999; Stephenson et al. 1999, Eason et al. 2002; Dennis and Gartrell 2015). See Howald et al. (2015) for a more detailed discussion on this topic.

4 Conclusions

Worldwide, ARs are the main tool used to control rodents in both urban and agricultural landscapes (Corrigan 2001; Elliott et al. 2016a). The large footprint of land and buildings baited with ARs, and the intensity of baiting has meant that not only are target rodents being exposed to ARs, but also non-target rodents, lagomorphs, songbirds, and insects (Borst and Counotte 2002; Olea et al. 2009; Sánchez-Barbudo et al. 2012; Elliott et al. 2014). Consequently, predators that depend on these prey species are also being exposed to ARs, sometimes fatally (Table 9.1). Given the potential for widespread AR exposure of the food chain, there is a need to understand the extent to which non-target prey are being exposed to ARs, and how this varies depending on AR application technique and landscape type. Even though much can be inferred from diet composition and residue levels found in predators, few field studies have monitored the uptake of ARs in target species and non-target prey, and the subsequent exposure in predators (Geduhn et al. 2016).

In summary, when reviewing AR exposure that has been reported in a wide range of predators, general inferences can be made with regard to the traits of the most affected species. In summary, predators that exhibit both a high incidence of exposure and relatively high liver residue concentrations of multiple ARs typically have similar traits:

They tend to be nocturnal, opportunistic predators for which rodents are a key component of their diet either seasonally or year-round, and they tend to be non-migratory predators living in close proximity to, or within landscapes that are heavily influenced by human activities such as intensive agriculture or urbanization.

However, our understanding of how ARs spread up the food chain is particularly lacking in urban landscapes, where to date, no exposure data has been collected on non-target prey. We also have a very limited understanding of non-target exposure in the urban-wildland interface (Moriarty et al. 2012). Most importantly, we need to decipher whether the mounting evidence of exposure in predators translates into any population level effects.

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