2016

Paying the Pipers: Mitigating the Impact of Anticoagulant Rodenticides on Predators and Scavengers

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Anticoagulant rodenticides, mainly second-generation forms, or SGARs, dominate the global market for rodent control. Introduced in the 1970s to counter genetic resistance in rodent populations to first-generation compounds such as warfarin, SGARs are extremely toxic and highly effective killers. However, their tendency to persist and accumulate in the body has led to the widespread contamination of terrestrial predators and scavengers. Commercial chemicals that are classified by regulators as persistent, bio-accumulative, and toxic (PBT) chemicals and that are widely used with potential environmental release, such as dichloro-diphenyl-trichloroethane (DDT) or polychlorinated biphenyls (PCBs), have been removed from commerce. However, despite consistently failing ecological risk assessments, SGARs remain in use because of the demand for effective rodent-control options and the lack of safe and humane alternatives. Although new risk-mitigation measures for rodenticides are now in effect in some countries, the contamination and poisoning of nontarget wildlife are expected to continue. Here, we suggest options to further attenuate this problem.

Keywords: rodenticide, nontarget wildlife, risk mitigation, anticoagulants, polluter-pays principle

Humans have occupied a large proportion of the globe's biodiversity hotspots, and in the process, many native species have been displaced and replaced with those that can tolerate or adapt to urban or agricultural landscapes (McKinney 2002). Among the most human-adapted species are rodents, particularly rat (*Rattus*) and mouse (*Mus*) species, which have been cohabiting with humans since Neolithic times (Reperant et al. 2013). There is a long history of humans attempting to control commensal rodents and contain the associated risks to human health from rodent-borne diseases, the destruction of food stores, and damage to infrastructure and other property. Recent estimates of the global impact of rodent pests are as high as $50 billion annually (Eason et al. 2010). Although many creative techniques have been devised to suppress rodent populations, for the past 50 years, as with most pest control, chemical biocides, primarily anticoagulant compounds, have been the dominant option worldwide. Once typified by the "blood-thinning" drug and rat poison warfarin, this prototypic first-generation anticoagulant (FGAR) compound has increasingly been replaced by more toxic and persistent analogues, or *second-generation anticoagulant rodenticides* (SGARs). Although highly effective, these chemicals are not specific to rodent pest species. Each year, US poison centers receive reports of rodenticide exposure by humans, mainly children, and ingestion by companion pets numbering in the tens of thousands (EPA 2011), and human exposures have been documented in Europe (Berny et al. 2010). SGAR contamination and poisoning of nontarget wildlife, particularly scavenging and predatory species such as raptorial birds, foxes, and weasels, which also provide important ecosystem services—including the control of rodent populations—are increasing in degree and scale (Rattner et al. 2014). As the extent of the environmental impact of anticoagulant usage became increasingly apparent over the past decade, agencies in North America, Europe, and elsewhere have wrestled with the regulatory challenge of balancing the demand for pest-control products with mitigating the impacts on nontarget organisms.

Widespread use, widespread contamination

Food production, storage, or transport facilities almost anywhere in the world may be commonly ringed with bait stations containing primarily SGARs. Less obvious are those

BioScience 66: 401–407.
doi:10.1093/biosci/biw028

Advance Access publication 18 February 2016

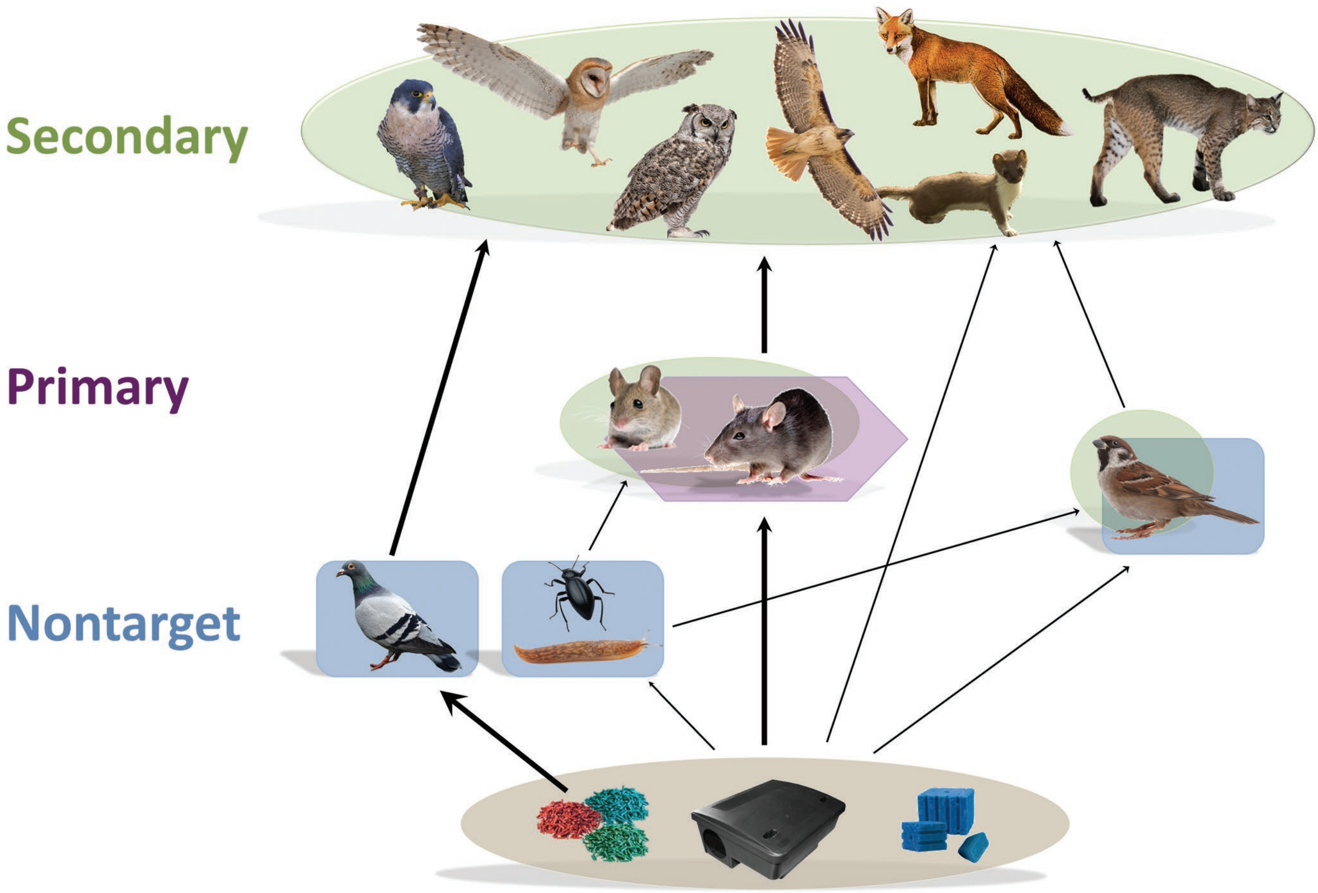

Figure 1. Rodenticide pathways to wildlife: Tamper-resistant bait stations are required in North America (but not in the EU) for the outdoor application of rodenticides, although other small organisms can also enter and feed. When used as a crop-protection product or for conservation use in some jurisdictions to eradicate pest mammals, loose pellet or bait blocks may be used without bait stations. Exposure patterns are complex, with many species potentially encountering a mixture of primary, secondary, or even tertiary exposure. The bold arrows indicate the most likely routes of transfer.

placed into sewers, waste disposal, and transport operations anywhere with human food or wastes. Many homeowners and apartment building managers regularly deploy rodenticide baits in a prophylactic manner (EPA 2011). Sales and use data are difficult to obtain because they are considered confidential business information, but estimates are in the hundreds of millions of dollars annually in the United States and European countries, for example (Rattner et al. 2014).

Compared with major plant-protection products that are commonly applied by tractor or aircraft over large areas in attempts to locate and kill pests, the actual quantity of rodenticide active ingredient used is minor because of the extreme acute toxicity, particularly of the SGARs, and the targeted nature of their deployment. Although there are some exceptions—such as the field application of loose baits into "artificial plowed galleries" in France to control water voles (*Arvicola terrestris*; Courdassier et al. 2012) and broadcast usage in New Zealand for invasive mammals (Blackie et al. 2014)—the major use of rodenticides is via bait stations. These are deployed to attract target species that then disperse after consuming the poison and can become the food of many avian and mammalian predators and scavengers (figure 1). Ironically, those predators are also the primary natural agents of control. Many predators will switch their diets and prey on rats and commensal birds, which often are the most common prey available in human-dominated landscapes (Shore et al. 2003, Riley et al. 2007, Hindmarch and Elliott 2014, 2015a, 2015b).

Since the first reports of anticoagulant residues in British raptors (Newton et al. 1990), SGARs have become contaminants of avian and mammalian predators and scavengers in jurisdictions worldwide (table 1), including national parks remote from intensive human activities (Gabriel et al. 2012). Many questions still remain, and further research is needed to quantify what proportion of exposed animals are acutely poisoned, the importance of sublethal effects such as increased clotting times, and whether there are any population-level impacts (Thomas et al. 2011, Coeurdassier et al. 2012, Jacquot et al. 2013, Rattner et al. 2014, Hindmarch and Elliott 2015a). The fact remains, however, that there are now

Table 1. Select examples of the bioaccumulation of anticoagulant rodenticide residues in the livers of diurnal and nocturnal birds of prey from locations worldwide.

Species	Sample Size[a]	Location	Percentage incidence	Reference
Various raptors	265	New York, United States	49	Stone et al. 2003
Various raptors	30	France	73	Lambert et al. 2007
Tawny owl	172	United Kingdom	19	Walker et al. 2008a
Red kite	23	United Kingdom	74	Walker et al. 2008b
Various owl species	164	Western Canada	70	Albert et al. 2010
Various raptor species	161	Massachusetts, United States	86	Murray et al. 2011
Various raptors	96	California, United States	92	Lima and Salmon 2010
Great horned owl	125	Canada	65	Thomas et al. 2011
Various raptors	430	Denmark	84–100	Christensen et al. 2012
Various species	129	Spain	28	Sánchez-Barbudo et al. 2012
Various species	773	Scotland	47	Hughes et al. 2013
Various species	30	Norway	53	Langford et al. 2013
Barn owl	63	United Kingdom	87	Walker et al. 2014
Various raptors	104	Canary Islands	61	Ruiz-Suárez et al. 2014

[a]Percentage of samples with liver residues of at least 1 SGAR.

relatively few anthropogenic chemicals, other than SGARs, that are widespread contaminants of top predators and are lethal toxicants. It is important to recognize that chemicals that are lethally toxic to breeding adult birds at ambient environmental exposure have had some of the greatest impacts on populations of long-lived "k-selected" top predators, moreso in many instances than more subtle reproductive toxicants. Classic examples include the cyclodiene insecticide dieldrin in British raptors (Newton 1990), lead from hunters' projectiles in California condors (Finkelstein et al. 2012), and most spectacularly, the veterinary drug diclofenac in Asian vultures (Oaks et al. 2004). By comparison, persistent organic pollutants (POPS), such as brominated flame retardants and perfluorinated surfactants, have received much more attention from scientists and regulators, and some are now scheduled for listing under the Stockholm Convention of Persistent Organic Pollutants, primarily on the basis of their persistence and bioaccumulative traits and long-range transport in the environment (*http://chm.pops.int/TheConvention/ThePOPs/TheNewPOPs/tabid/2511/Default.aspx*). However, in contrast to SGARs, currently, there is sparse evidence for significant effects of environmentally relevant concentrations of those compounds on wildlife populations, including the top predators that accumulate the greatest concentrations (e.g., Henny et al. 2009, Cesh et al. 2010, Harris and Elliott 2011, Fair et al. 2013). SGARs are lethal toxicants that are regularly deployed in a manner to provide a direct pathway and impact on rare and valuable top predators (Thomas et al. 2011, Gabriel et al. 2012, Rattner et al. 2014).

Although wildlife managers are concerned about the impact of SGARs on nontarget wildlife, somewhat ironically, these chemicals, particularly brodifacoum, have been widely used across the globe in conservation efforts to remove introduced rodents from previously predator-free islands. Entrenched populations of invasive rodents, principally *R. norvegicus*, have eliminated endemic bird and mammal species from some islands and severely affected breeding seabirds on many others (Howald et al. 2007). On some islands, populations of other predators are limited, but on islands along the Pacific coast of Canada and the United States, bald eagles (*Haliaeetus leucocephalus*), for example, have been poisoned during rat-eradication efforts (Howald et al. 1999). Like pesticide regulators, wildlife managers have opted to accept the risks of local contamination and impact on nontarget wildlife because of the effectiveness of anticoagulants and their cost efficiency over other options. Until alternative and safer control rodenticides are developed, it seems likely that such conservation use of SGARs will continue (e.g., Blackie et al. 2014).

Pathways forward

New risk-mitigation measures for anticoagulant use are now in effect in Canada (*www.hc-sc.gc.ca/cps-spc/pubs/pest/_fact-fiche/restriction-rodenticides/index-eng.php*) and more recently in the United States after a lengthy litigation process with one manufacturer (*www2.epa.gov/rodenticides/canceling-some-d-con-mouse-and-rat-control-products*). Point-of-sale measures restrict household users to first-generation anticoagulants or other rodenticides with alternate modes of action, such as the neurotoxin bromethalin. Packages are now limited in size and bait formulated into rigid blocks and sold with or in a tamper-resistant bait station. SGARs will, however, continue to be registered federally in the United States and Canada for use in and near buildings, waste receptacles and fence lines in agricultural settings, by licensed applicators. Again, data are limited on commercial sales, but one Canadian jurisdiction reported steady or increasing sales of commercial SGAR products over the period 1995 to 2009 (Elliott et al. 2014). The more toxic compounds, brodifacoum and difethialone, are now confined to indoor usage, with only the less toxic and persistent SGAR, bromadiolone, permitted for outdoor application. Those measures should reduce exposure of nontargets to the highly toxic SGARs. However, although technically indoor use, the potential continues for the movement of brodifacoum, for example, consumed by rodents to the exterior of unsealed buildings in these exposed rodents, putting predators at risk (Elliott et al. 2014). There is, therefore, a need to continue to monitor AR exposure and risk in nontarget populations.

The US state of California has gone further than the federal initiative. In California, the SGARs brodifacoum, bromadiolone, difenacoum and difethialone have been designated as "restricted materials" and can only be obtained and applied by a certified pesticide applicator under permit from a county commissioner. Aboveground bait may be placed no more than 50 feet from a manmade structure unless there is a feature that harbors or attracts targeted pests (*www.cdpr.ca.gov/docs/legbills/rulepkgs/13-002/13-002.htm*). In addition, the California Food and Agriculture Code (Section 12978.7) now prohibits the use of these SGARs in state parks, wildlife refuges, and conservancies.

In the European Union (EU), SGARs are recognized as posing significant risk to birds and nontarget mammals but continue to be authorized for use as biocides to protect public health and, in some member states, as plant-protection products. Several risk-mitigation measures (RMMs) have been suggested and applied in some member states by their authorities that deliver marketing authorizations (Berny et al. 2014). Because RMMs are set by each individual member state, a single commercial product may have more than one set of RMMs attached to its marketing authorizations across Europe.

The step taken in North America to remove SGARs from the domestic retail market should primarily reduce risk to humans, particularly children, and companion pets (*www2.epa.gov/rodenticides/canceling-some-d-con-mouse-and-rat-control-products*). Cross-border e-commerce may provide a loophole to gain access to restricted pesticides, including rodenticides in some jurisdictions. However, in the United States, for example, online sales of pesticides have been subject to the same controls as purchases from traditional stores for more than a decade (EPA 2004). The exposure of nontarget wildlife to SGAR products should also decrease in suburban and urban areas, where domestic use is a major contributor. However, nontargets, particularly predators and scavengers, may continue to encounter substantial residues certainly of bromadiolone and potentially of the more toxic SGARs in their diet from continuing use in structural and food production and transport facilities.

The development of safe and effective rodenticides is a complex research and development challenge, although there are some promising new advances (Blackie et al. 2014). Until such time, we suggest a three-pronged approach that could further mitigate adverse nontarget effects.

Rationalize usage and deployment strategies. The first of these, which is already being implemented by some corporations and jurisdictions, is to rationalize usage and deployment strategies. For decades, structural rodent management relied on the regular, prophylactic use of rodenticides to prevent infestations and meet health and safety standards. Bait stations were required to be placed at specified intervals and were subject to audit. The focus was on the placement of bait rather than on testing efficacy in rodent control. Recently, in the United States, however, under the EPA's Pesticide Environmental Stewardship Program (PESP), some major food and "big-box" retailers have moved to greatly reduce rodenticide usage in their food-supply chains (*www.epa.gov/pestwise/pesp/members/strategies/walmart.pdf*). That approach essentially employs the long-established principles of integrated pest management (IPM) to monitor pest presence and apply pesticides only as needed. It also takes the concept further to develop, for example, "Go Green" programs which have used data on the ecology and behavior of rodents to develop more effective control programs.

A cautionary note, however: Although there are data on cost savings to corporate and other end users from such IPM-based reductions in usage (Arjo et al. 2009), it is much less clear whether changing from prophylactic to evidence-driven bait deployment has resulted in significant reductions in the availability of poisoned rodents to predators and scavengers. There is some evidence that restrictions on the field use of anticoagulants in France resulted in both decreased amounts of products applied and increased population densities of the red fox (*Vulpes vulpes)* following periods of reduced rodenticide usage (Jacquot et al. 2013). We are not aware of other studies that quantified the mitigating efficiency on actual risks. For other types of pesticide application, such quantification was essential to ensure the implementation of mitigating measures, such as the effectiveness of buffer zones and the use of specific spray nozzles to minimize the spray drift of pesticides into adjacent waterbodies (e.g., de Snoo and de Wit 1998). That has resulted in sophisticated models to assess spray drifts and is implemented in the guidance of pesticide use and its further regulation and labeling (Hewitt 2000). Without such quantitative evidence, the justification for specific IPM measures may encounter skepticism and opposition from some stakeholders.

Develop and implement outreach and educational stewardship programs. The second measure would be consideration for the further development and implementation of outreach and educational stewardship programs by industry and government. Such programs are already in effect in areas of Europe (*www.cefic.org/Documents/About-Us/Industry%20sectors/EBPF/Guideline-on-Best-Practice-in-the-Use-of-Rodenticides-in-the-EU.pdf*), and arguably the most developed is the stewardship scheme commencing in the United Kingdom in 2016 (*www.thinkwildlife.org/stewardship-regime*). That has been developed and led by an industry consortium (*www.thinkwildlife.org/about-crru*) working with the relevant Competent Authority and has the overall aim of reducing exposure in nontarget wildlife while ensuring efficacious rodent control, including areas where there is resistance to some SGARs. The program, underpinned by the development and dissemination of a code of best practice (*www.thinkwildlife.org/crru-downloads/crru-uk-code-of-best-practice*), involves multiple activities, including approval and certification of training courses and a requirement of proof of competence at the point of sale of professional products. A further major component is the monitoring of outcomes, with data assessed by the Competent Authority. Such monitoring includes the periodic survey of the knowledge, attitudes, and practices of all professional rodenticide users; the independent monitoring of changes in exposure (as measured from tissue residues) in a sentinel nontarget species, the barn owl *Tyto alba* (Shore et al. 2014); and the evaluation of the breeding success of selected barn owl populations in relation to rodenticide use. Top predators, such as the barn owl, provide broad ecosystem services, including the regulation of rodent populations.

Implementing a paying-the-piper strategy. A third measure might entail compensation for the collateral damage of predatory birds and mammals and could be considered, although the analogy is not perfect, as a *paying-the-piper* approach. The cost of impact on rodent-regulating allies, including raptors, weasels, canines, and felids, could be borne generally by users of the products, not the commons (viz. imposition of a form of the polluter-pays principle). The concept is widely recognized and is simply that those who damage or deplete the environment should bear the costs. Applications of the concept include having resource extractors pay for not only the costs of waste disposal, cleanup, and restoration but also the costs of enforcing the regulations. That is effectively a form of paying for ecosystem services (Engel et al. 2008). Other examples include the payment of deposit fees on beverage containers, as well as ecofees on car batteries, tires, and other products (Driedger 2001). Many agree that the principle is inherently sound and logical, both "legally and economically" (OECD 2008); differences surround defining who or what is affected by the pollution and, therefore, who should be compensated (Driesen 1997). Some of the arguments about the principle are fundamentally rooted in differences in political philosophy, related to views on private property rights and the contention that owners of private property and therefore resources make better stewards (and therefore conservationists) than the commons or public (Cordato 2001). In the majority of political jurisdictions, however, the reality is a mix of public and private ownership of land and resources, and wild plants and animals are considered to be public resources and the property of the state or commons (Geist et al. 2001).

There are already farsighted examples in which the polluter-pays approach in the form of fees, levies, or responsibility for education and monitoring of impacts have been applied to management and regulation of rodenticides. California set a precedent by implementing an eco-fee system at point of sale (*www.vpcrac.org/about/surcharge-legislation*), whereby a fee of $ 0.50 per pound (227.5 grams) is added to the cost of vertebrate-pest-control products (e.g., anticoagulant rodenticides). Fees are used mainly for research on the development of alternative products, improvements in the safe use of existing products, and investigation of toxicity and environmental effects. It also should be recognized that, as we discussed above, in the United Kingdom, SGAR manufacturers and suppliers aim to pay what can be considered effectively a fixed eco-fee by developing, leading, and funding a comprehensive SGAR stewardship program.

We suggest that the broader application of such a paying-the-piper approach, in concert with the rationalized deployment and educational outreach, could help offset the impact of the ongoing global use of SGAR compounds. Fees might be used more broadly, such as for compensation and mitigation programs for the affected predators, in the form of active management of both populations and habitat. There are precedents for the use of money in this way obtained in the United States from Natural Resource Damage Assessments of oils spills and contaminated sites (*www.epa.gov/superfund/programs/nrd/primer.htm*). The most recent and well-publicized example is the settlement between the US federal government and BP to compensate for injury and damage to resources resulting from the Deep Water Horizon oil spill (*www.doi.gov/deepwaterhorizon*).

Conclusions

Given the likelihood that anticoagulant rodenticides will continue to be deployed widely across the globe to suppress pest rodent populations, then some ongoing impacts on nontarget wildlife seem inevitable. Here, we suggest that in addition to recent risk-mitigation measures that have been imposed in some jurisdictions, other activities might be implemented. Namely, we suggest that (a) industry consider the implementation of validated IPM procedures to reduce and optimize use of products, (b) user groups adopt effective education and outreach programs for applicators and the public, and (c) the consideration of eco-fees on rodenticide sales, similar to those in effect in California (*www.vpcrac.org/about/surcharge-legislation*). Such fees could be used to raise funds for research into developing new products, investigating and monitoring select nontarget species, and providing compensation for habitat or mitigation measures for affected nontarget populations. Given that governments elsewhere in the world rely heavily on the United States and Europe for leadership in chemical regulations, the adoption of these proposed measures could have broader implications.

Acknowledgments

S. Lee assisted with the artwork. C. Bishop reviewed earlier versions of the manuscript. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the US Government.

References cited

Albert C, Mineau P, Wilson LK, Trudeau S, Elliott JE. 2010. Rodenticide residues and autopsy data for three owl species from British Columbia, Canada. Archives of Environmental Contamination and Toxicology 58: 451459.

Arjo WM, Shwiff S, Kirkpatrick K. 2009. Short-term evaluation of two integrated pest management programs for mountain beaver (*Aplodontia rufa*) control. Crop Protection 28: 703–709.

Berny P, Esther A, Jacob J, Prescott C. 2014. Risk Mitigation Measures for Anticoagulant Rodenticides as Biocidal Products: Report to the European Union. Publications Office of the European Union. doi:10.2779/241180

Blackie HM, et al. 2014. Innovative developments for long-term mammalian pest control. Pest Management Science 70: 345–351.

Cesh LS, et al. 2010. Polyhalogenated aromatic hydrocarbons and metabolites: Relation to circulating thyroid hormone and retinol in nestling bald eagles (*Haliaeetus leucocephalus*). Environmental Toxicology and Chemistry 29: 1301–1310.

Christensen TK, Lassen P, Elmeros M. 2012. High exposure rates of anticoagulant rodenticides in predatory bird species in intensively managed landscapes in Denmark. Archives of Environmental Contamination and Toxicology 63*:* 437–444.

Coeurdassier M, et al. 2012. The diet of migrant Red Kites *Milvus milvus* during a Water Vole *Arvicola terrestris* outbreak in eastern France and the associated risk of secondary poisoning by the rodenticide bromadiolone. Ibis 154: 136–146.

Cordato RE. 2001. The Polluter Pays Principle: A Proper Guide for Environmental Policy. Institute for Research on the Economics of Taxation.

De Snoo GR, de Wit PJ. 1998. Buffer zones for reducing pesticide drift to ditches and risks to aquatic organisms. Ecotoxicology and Environmental Safety 41: 112–118.

Driedger R J. 2001. From cradle to grave: Extended producer responsibility for household hazardous wastes in British Columbia. Journal of Industrial Ecology *5*: 89–102.

Driesen DM. 1997. Societal cost of environmental regulation: Beyond administrative cost–benefit analysis. Ecology LQ 24: 545–617.

Eason CT, Fagerstone KA, Eisemann JD, Humphrys S, O'Hare JR, Lapidge SJ. 2010. A review of existing and potential New World and Australasian vertebrate pesticides with a rationale for linking use patterns to registration requirements. International Journal of Pest Management 56: 109–125.

Elliott JE, Hindmarch S, Albert CA, Emery J, Mineau P, Maisonneuve F. 2014. Exposure pathways of anticoagulant rodenticides to nontarget wildlife. Environmental Monitoring and Assessment 186: 895–906.

Engel S, Pagiola S, Wunder S. 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65: 663–674.

[EPA] Environmental Protection Agency. 2004. Fact Sheet on Pesticides Sales in E-Commerce. EPA Office of Environmental Protection and Compliance Assurance.

———. 2011. A Set of Scientific Issues Being Considered by the Environmental Protection Agency Regarding: Scientific Conclusions Supporting EPA's FIFRA Section 6(B) Notice of Intent to Cancel Twenty Homeowner Rodenticide Bait Products. EPA Office of Chemical Safety and Pollution Prevention.

Fair PA, et al. 2013. Associations between perfluoroalkyl compounds and immune and clinical chemistry parameters in highly exposed bottlenose dolphins (*Tursiops truncatus*). Environmental Toxicology and Chemistry 32: 736–746.

Finkelstein ME, et al. 2012. Lead poisoning and the deceptive recovery of the critically endangered California condor. Proceedings of the National Academy of Sciences 109: 11449–11454.

Gabriel MW, Woods LW, Poppenga R, Sweitzer RA, Thompson C, Matthews SM, Higley JM. 2012. Anticoagulant Rodenticides on our public and community lands: Spatial distribution of exposure and poisoning of a rare forest carnivore. PLOS ONE 7 (art. e40163).

Geist V, Mahoney SP, Organ JF. 2001. Why hunting has defined the North American model of wildlife conservation. Transactions of the North American Wildlife and Natural Resources Conference 66: 175–185.

Harris ML, Elliott JE. 2011. Polychlorinated biphenyls, dibenzo-*p*-dioxins and dibenzofurans and polybrominated diphenyl ethers in birds. Pages 471–522 in Beyer WN, Meador J, eds. Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. CRC Press.

Henny CJ, Kaiser JL, Grove RA, Johnson BL, Letcher RJ. 2009. Polybrominated diphenyl ether flame retardants in eggs may reduce reproductive success of ospreys in Oregon and Washington, USA. Ecotoxicology 18: 802–813.

Hewitt AJ. 2000. Spray drift: Impact of requirements to protect the environment. Crop Protection 19: 623–627.

Hindmarch S, Elliott JE. 2014. A specialist in the city: The diet of barn owls along a rural to urban gradient. Urban Ecosystems 18: 477–488.

———. 2015a. When owls go to town: The diet of urban barred owls. Journal of Raptor Research 49: 66–74.

———. 2015b. Comparing the diet of Great Horned Owls (Bubo virginianus) in rural and urban areas of southwestern British Columbia. Canadian Field-Naturalist 128: 393–399.

Howald GR, Mineau P, Elliott JE, Cheng KM. 1999. Brodifacoum poisoning of avian scavengers during rat control on a seabird colony. Ecotoxicology 8: 431–447.

Howald GR, Donlan C, Galván JP, Russell JC, Parkes J, Samaniego A, Tershy B. 2007. Invasive rodent eradication on islands. Conservation Biology 21: 1258–1268.

Hughes J, Sharp E, Taylor MJ, Melton L, Hartley G. 2013. Monitoring agricultural rodenticide use and secondary exposure of raptors in Scotland. Ecotoxicology 22: 974–984.

Jacquot M, et al. 2013. Using long-term monitoring of red fox populations to assess changes in rodent control practices. Journal of Applied Ecology 50: 1406–1414.

Lambert O, Pouliquen H, Larhantec M, Thorin C, L'Hostis M. 2007. Exposure of raptors and waterbirds to anticoagulant rodenticides (difenacoum, bromadiolone, coumatetralyl, coumafen, brodifacoum): Epidemiological survey in Loire Atlantique (France). Bulletin of Environmental Contamination and Toxicology 79: 91–94.

Langford KH, Reid M, Thomas KV, 2013. The occurrence of second generation anticoagulant rodenticides in non-target raptor species in Norway. Science of the Total Environment 450–451: 205–208.

McKinney ML. 2002. Urbanization, biodiversity, and conservation: The impacts of urbanization on native species are poorly studied, but educating a highly urbanized human population about these impacts can greatly improve species conservation in all ecosystems. BioScience 52: 883–890.

Murray M. 2011. Anticoagulant rodenticide exposure and toxicosis in four species of birds of prey presented to a wildlife clinic in Massachusetts, 2006–2010. Journal of Zoo and Wildlife Medicine 42: 88–97.

Newton I, Wyllie I, Freestone P. 1990. Rodenticides in British barn owls. Environmental Pollution 68: 101–117.

Oaks JL, Gilbert M, Virani MZ, Watson RT, Meteyer CU, Rideout BA, Khan AA. 2004. Diclofenac residues as the cause of vulture population decline in Pakistan. Nature 427: 630–633.

[OECD] Organisation for Economic Co-operation and Development. 2008. Modernizing Canada's agricultural policies. Pages– in OECD. OECD Economic Surveys: Canada 2008. OECD. (2 March 2016; *http://dx.doi.org/10.1787/eco_surveys-can-2008-7-en*)

Rattner BA, Lazarus RS, Elliott JE, Shore RF, van den Brink N. 2014. Adverse outcome pathway and risks of anticoagulant rodenticides to predatory wildlife. Environmental Science and Technology 48: 8433–8445.

Reperant LA, Cornaglia G, Osterhaus AD. 2013. The Importance of understanding the human–animal interface. Pages– in Mackenzie JS, Jeggo M, Daszak P, Richt JA, eds. One Health: The Human–Animal–Environment Interfaces in Emerging Infectious Diseases. Springer.

Riley SPD, Bromley C, Poppenga RH, Uzai FA,Whited L, Sauvajot RM. 2007. Anticoagulant exposure and notoedric mange in bobcats and mountain lions in urban southern California. Journal of Wildlife Management 7: 1874–1884.

Ruiz-Suárez N, et al. 2014. Assessment of anticoagulant rodenticide exposure in six raptor species from the Canary Islands (Spain). Science of the Total Environment 485: 371–376.

Sánchez-Barbudo IS, Camarero PR, Mateo R. 2012. Primary and secondary poisoning by anticoagulant rodenticides of non-target animals in Spain Science of the Total Environment 420: 280–288.

Shore RF, Birks JDS, Afsar A, Wienburg CL, Kitchener AC. 2003. Spatial and temporal analysis of second-generation anticoagulant rodenticide residues in polecats (*Mustela putorius*) from throughout their range in Britain, 1992–1999. Environmental Pollution 122: 183–193.

Shore RF, Henrys PA, Walker LA. 2014. Power Analysis of Liver Second Generation Anticoagulant Rodenticide (SGAR) Residue Data in Barn Owls from Britain: A Predatory Bird Monitoring Scheme (PBMS) Report. Contract Report to the Health and Safety Executive. Centre for Ecology and Hydrology. (2 March 2016; *https://wiki.ceh.ac.uk/download/attachments/134414860/HSE%20Power%20analysis%20report%202014-final%20version.pdf?version=1&modificationDate=1410527042831&api=v2*).

Stone WB, Okoniewski JC, Stedelin JR. 2003. Anticoagulant rodenticides and raptors: Recent findings from New York, 1998–2001. Bulletin of Environmental Contamination and Toxicology 70: 34–40.

Thomas PH, Mineau P, Shore RF, Champoux L, Martin PA, Wilson LK, Fitzgerald G, Elliott JE. 2011. Second generation anticoagulant rodenticides in predatory birds: Probabilistic characterization of toxic liver concentrations and implications for predatory bird populations in Canada. Environment International 37: 914–920.

Walker LA, Turk A, Long SM, Wienburg CL, Best J, Shore RF. 2008a. Second generation anticoagulant rodenticides in tawny owls (*Strix aluco*) from Great Britain. Science of the Total Environment 392: 93–98.

Walker LA, Shore RF, Turk A, Pereira MG, Best J. 2008b. The Predatory Bird Monitoring Scheme (PBMS): Identifying chemical risks to top predators in Britain. Ambio 37: 466–471.

Walker LA, Chaplow JS, Moeckel C, Pereira MG, Potter ED, Shore RF. 2014. Anticoagulant Rodenticides in Predatory Birds 2012: A Predatory Bird Monitoring Scheme (PBMS) Report. Centre for Ecology and Hydrology. (2 March 2016; *https://wiki.ceh.ac.uk/download/attachments/134414860/PBMS%20Report%20Rodentocide%202012.pdf?version=1&modificationDate=1402491816000&api=v2*).

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