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Office of Pesticide Programs (OPP) Docket
Environmental Protection Agency Docket Center (EPA/DC), (28221T)
1200 Pennsylvania Ave. NW
Washington, D.C. 20460-0001

RE: [EPA–HQ–OPP–2010–0752] Public Comments on Registration Review Proposed Interim Decision for Sodium Cyanide, Case No. 3073 (September 2018)

To Whom It May Concern:

Thank you for the opportunity to comment on the U.S. Environmental Protection Agency's (EPA's) Proposed Interim Registration Review Decision for Sodium Cyanide, Case Number 3073 (hereinafter "Proposed Interim Decision").[1] Sodium cyanide is a highly toxic pesticide registered for restricted use under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. §§ 136 et seq.[2]

Sodium cyanide is used in M-44 ejector devices — also known as "cyanide bombs" — to kill coyotes (*Canis latrans*), red fox (*Vulpes vulpes*), gray fox (*Urocyon cinereoagenteus*), and wild dogs suspected of preying on livestock. Because of the dangers posed by sodium cyanide to wildlife and people, we respectfully request that the EPA not reregister sodium cyanide for this use in the lower 48 states and, instead, commence cancellation proceedings during the pending registration review process. In the alternative, we request that the EPA impose additional, stricter use restriction modifications than those put forward in the Proposed Interim Decision.

I. Introduction

FIFRA, 7 U.S.C. § 136 et seq., provides the framework for federal registration of pesticide use, sale, and distribution. The law is intended to prohibit the use of pesticides that cause unreasonable adverse effects on the environment.[3] The Administrator of the EPA is responsible for carrying out

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[1] U.S. Envtl. Protection Agency, Sodium Cyanide Proposed Interim Registration Review Decision Case Number 3073 (Sept. 2018) [hereinafter "PID"].

[2] Our comments pertain to all active registrations for M-44 cyanide capsules (sodium cyanide) in the lower 48 states and hereinafter reference all active registrations collectively when using the term "sodium cyanide" or "M-44 devices," including EPA Registration No. 56228-15 (APHIS), EPA Registration No. 35978-1 (Wyoming), EPA Registration No. 35975-2 (Montana), EPA Registration No. 39508-1 (New Mexico), EPA Registration No. 33858-2 (Texas), EPA Registration No. 13808-8 (South Dakota), and EPA Registration No. CA840006 (Sodium Cyanide).

[3] 7 U.S.C. § 136a(a); *see id.* at § 136(bb)(defining, in relevant part, "[t]he term 'unreasonable adverse effects on the environment' means (1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide. . . .").

the mandates of the Act.[4] Pursuant to this obligation, the Administrator may limit the use of certain pesticides to prevent unreasonable adverse effects on the environment.[5] FIFRA requires the Administrator to review registered pesticides periodically to ensure that continuing use will not have unreasonable adverse effects on the environment.[6]

M-44 cyanide capsules (containing the pesticide sodium cyanide) are registered for restricted use under FIFRA (EPA registration No's. 56228-15, 35978-1, 35975-2, 39508-1, 33858-2, 13808-8, and CA840006). Wildlife Services, a program of the U.S. Department of Agriculture, Animal and Plant Health Inspection Service (APHIS), is a registered user of sodium cyanide (EPA Registrant No. 56228-15). Other registered users include Wyoming Dep't of Agriculture (No. 35978-1), Montana Dep't of Agriculture (No. 35975-2), New Mexico Dep't of Agriculture (No. 39508-1), Texas Dep't of Agriculture (No. 33858-2), and South Dakota Dep't of Agriculture (No. 13808-8).

The Administrator commenced the FIFRA registration review process for sodium cyanide in 2010, and released the Proposed Interim Decision for sodium cyanide in September 2018.[7] These comments respond to the associated 60-day comment period announced concurrent with the publication of the Proposed Interim Decision in the Federal Register in December 2018.[8]

M-44 Devices and Overview of Use

Sodium cyanide is the pesticide active ingredient used in M-44 devices, which are also known as "cyanide bombs." Unlike bombs, no explosives are used.[9] Instead, an M-44 uses a spring-loaded device that is screwed or pushed into the ground. The device is topped with scented bait to lure animals (such as coyotes, foxes, and other canids) to bite. Once the animal's teeth clench on the bait, a spring shoots a pellet of sodium cyanide into the animal's mouth.

The sodium cyanide combines with available moisture including saliva to make hydrogen cyanide gas, which is readily absorbed by the lungs and poisons the animal by inactivating an enzyme essential to mammalian cellular respiration.[10] That leads to central nervous system depression, cardiac arrest, and respiratory failure.[11] While death often comes quickly, sometimes an animal receives a sublethal dose that results in agonizing symptoms such as partial paralysis, labored

4 7 U.S.C. § 136(b).
5 7 U.S.C. § 136a(c)(5)-(6).
6 7 U.S.C. § 136a-1; 40 C.F.R. § 155.40.
7 PID at 6.
8 U.S. Envtl. Protection Agency, Registration Review Proposed Interim Decisions for Several Pesticides; Notice of Availability, 83 Fed. Reg. 62571 (Dec. 4, 2018).
9 While APHIS objects to the use of the word "bomb" in reference to M-44s, some members of the public have adopted the term because M-44 "cyanide bombs" act as such per common dictionary definitions, which generally define "bombs" as containers filled with a destructive substance designed to explode on impact or when detonated. M-44s are filled with powdery sodium cyanide poison; their spring-activated ejectors spew the poison into the air in a cloud; the ejectors' force can spray the cyanide up to five-feet. M-44s are deadly devices and to some members of the public, the definition of "bomb" is appropriate.
10 U.S. Fish & Wildlife Service, *Biological Opinion: Effects of 16 Vertebrate Control Agents on Endangered and Threatened Species* (1993) at II-73 [hereinafter "1993 BiOp"].
11 1993 BiOp at II-73.

breathing or blindness.[12] Animals receiving a sublethal dose might recover from the toxicity but could die from predators or exposure during the recovery period.[13]

Sodium cyanide is a Category 1 acute toxicant according to the EPA: the most hazardous, due to its high level of toxicity and the imminent harm it poses to the environment and to humans.[14] Sodium cyanide is highly soluble in water and highly toxic to most aquatic organisms, and as a result, M-44 capsules may not be used within 200 feet of water.[15]

Wildlife Services and state agencies use M-44s in locales across the country to kill so-called "nuisance" wildlife, including coyotes, gray foxes and red foxes, and free-roaming dogs.[16] M-44s containing sodium cyanide are deployed primarily by Wildlife Services; however, the following states also have authority for their use: South Dakota, Montana, Wyoming, New Mexico, and Texas.[17] According to its 2015 and 2016 data, Wildlife Services uses M-44s in the following states: Colorado, Idaho, Montana, North Dakota, Nebraska, New Mexico, Nevada, Oklahoma, Oregon, Texas, Utah, Virginia, West Virginia and Wyoming.[18]

In 2017, the most recent data available, Wildlife Services reports that it killed at least 13,232 animals with M-44s, including: 21 dogs, 12,119 coyotes, 1,013 foxes, 48 raccoons, 21 opossums, 5 skunks, 2 swine, 2 ravens and one gray wolf.[19] Of these 2017 M-44 deaths from Wildlife Services, over 200 were nontarget animals, including: 110 foxes, a gray wolf, 48 raccoons, 21 opossums, and more.[20]

Impacts of M-44s on Endangered Wildlife

In a 1993 Biological Opinion that analyzed the impacts of sodium cyanide on endangered wildlife, the U.S. Fish and Wildlife Service (FWS) found that any carrion-feeding animal able to activate the M-44 device is at risk. For that reason, FWS placed additional restrictions on use of M-44s to try to reduce the risk to wildlife protected under the Endangered Species Act.

In its 1994 Reregistration Eligibility Decision (RED) pertaining to the use of sodium cyanide capsules in M-44 units, EPA concluded that the M-44 device did not pose unreasonable risks to humans or the environment if used in accordance with the 26 use restrictions listed on the label, plus language determined by FWS to be needed to protect endangered species likely to be jeopardized by use of M-44s.

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[12] U.S. Dep't of Agriculture, APHIS, Wildlife Services, The Use of Sodium Cyanide in Wildlife Damage Management (May 2017) at 17, 20.
[13] *Id.* at 20.
[14] U.S. Environmental Protection Agency, Reregistration Eligibility Decision (R.E.D.) Facts: Sodium Cyanide (1994) *available at* https://archive.epa.gov/pesticides/reregistration/web/pdf/3086fact.pdf.
[15] 1993 BiOp at II-73.
[16] *Id.*
[17] *Id.*
[18] U.S. Dep't of Agriculture, Wildlife Services, *2016 Program Data Reports, available at* http://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/sa_reports/sa_pdrs/ct_pdr_home_2016; U.S. Dep't of Agriculture, Wildlife Services, *2015 Program Data Reports, available at* http://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/sa_reports/sa_pdrs/ct_pdr_home_2015.
[19] U.S. Dep't of Agriculture, Wildlife Services, *2017 Program Data Reports, available at* https://www.aphis.usda.gov/wildlife_damage/pdr/PDR-G_Report.php?fy=2017&fld=KILLED_EUTH&fld_val=0.
[20] *Id.*

As the EPA is aware, that analysis is now decades old. Since then, M-44s have killed numerous non-target, federally protected endangered animals.

Even when M-44s are used as intended to kill coyotes and other canids, harm to the environment can occur because of the important ecosystem roles played by these animals. Numerous studies analyze how carnivore removal, in particular, can cause a wide range of unanticipated impacts that are often profound, including on native plant communities, wildfire and biogeochemical cycles, the spread of disease or invasive species, and more (e.g. Beschta and Ripple 2009; Levi et al. 2012; Bergstrom et al. 2013; Bergstrom 2017) (cited in and attached to the 2017 petition).

We understand that the EPA has initiated formal consultation with the FWS pursuant to Section 7 of the Endangered Species Act (ESA) regarding the impacts of sodium cyanide and M-44 use on species listed under the ESA. We look forward to reviewing the agencies' renewed analysis in the required, revised Biological Opinion anticipated for completion by December 31, 2021. EPA cannot finalize its decision on reregistration prior to completion of this analysis, and we request that a subsequent public comment opportunity be afforded upon publication of the new Biological Opinion so that the public may incorporate the findings of the revised analysis into their comments on the pending registration review.

<u>Availability of Viable Alternatives</u>

The balance of interests clearly weighs in favor of prohibiting M-44s given the numerous viable alternatives to protect livestock from predation. For example, guard animals (including dogs, llamas, and donkeys) can be deployed, herders and range riders can be employed, and livestock operators can change animal husbandry practices to lessen the risk of predation. Deterrents, such as sound- and light-emitting frightening devices can also be used to scare away potential predators. Indeed, numerous studies have demonstrated the effectiveness of nonlethal methods to protect livestock from predators (e.g. Shivik et al. 2003; Lance et al. 2010) (cited in and attached to 2017 petition).

Moreover, numerous scientific studies seriously call into question the efficacy of lethal predator control (e.g., Berger 2006, Harper et al. 2008; Musiani et al. 2003, Wielgus and Peebles 2014, Treves et al. 2016) (cited in and attached to 2017 petition) (see also Bergstrom 2017; Ekland et al. 2017; Lennox et al. 2018; Santiago-Avila et al. 2018 (attached herewith)). For example, in a study based upon a review of 25 years of livestock depredation data, Wielgus and Peebles (2014) found that with increased predator persecution, livestock losses increased in the following year.

In short, several viable and more effective alternative tools to address livestock conflicts exist, eliminating the need for M-44 sodium cyanide capsules altogether.

II. 2017 Petition to Cancel Registration for Sodium Cyanide

In August 2017, WildEarth Guardians and the Center for Biological Diversity, along with a number of co-petitioners, petitioned the EPA to cancel, suspend, issue a stop order, and initiate a Special Review for all sodium cyanide registrations in the lower 48 pursuant to FIFRA and its implementing regulations. Petitioners remain convinced that the cancellation of sodium cyanide's registrations is proper: cancellation of a pesticide's registration is warranted where the pesticide, "when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment."[21] As documented in the petition, the registration for sodium cyanide should be cancelled because its continued use is causing unreasonable adverse effects on the environment, members of the public, and non-targeted wildlife and companion animals. As requested in the EPA's November 2018 response to the petition, Petitioners hereby resubmit and fully incorporate the contents of their August 2017 petition – and the studies upon which it relies – into the current comment record (attached).

III. Proposed Interim Decision Comments

The Administrator has published the Proposed Interim Decision to move forward with the completed registration review components and implement interim risk mitigation measures via label changes to try to alleviate some of the risk petitioners outlined in the August 2017 petition.[22] While we appreciate that the Administrator is taking some affirmative action to ensure the safety of the public and non-target wildlife that may be exposed to lethal M-44s, we caution that the proposed label changes do not go far enough, and respectfully request that the Administrator commence cancellation proceedings for sodium cyanide altogether. Detailed comments regarding the various components of the Proposed Interim Decision are included below.

A. Use/Usage

The Proposed Interim Decision acknowledges the intended use of products containing sodium cyanide, primarily — and as relevant to these comments — as a predacide in lethal M-44 devices.[23] However, the "Use and Usage" section of the Proposed Interim Decision contains no information or data regarding the frequency or location of use. There is no information regarding the number of M-44s currently or historically placed on the landscape, nor is there information regarding the effectiveness of use. We request that the Administrator conduct a more thorough review of M-44 use nationwide and include such data in any interim or final registration review decision so that the public may be properly informed of the full extent of M-44 use and the potential level of exposure to members of the public, their companion animals, and non-target wildlife. That analysis must incorporate data from the registrants, including Wildlife Services and the state agencies with registrations. A 2017 report from Wildlife Services explains that between FY11 and FY15, Wildlife Services used an average of 27,629 sodium cyanide capsules annually in 17 states.[24] The label provides for a maximum of 10 capsules per 100 acres.[25]

B. Scientific Assessments

The "Scientific Assessments" section of the Proposed Interim Decision is severely lacking in multiple respects. A key concern — that of the high potential for unintended death of members of the public, companion animals, and non-target wildlife — is inadequately analyzed in the documents

[21] 7 U.S.C. § 136d(b); *see also id.* §§ 136(bb)(providing that "[t]he term 'unreasonable adverse effects on the environment' means (1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide ….").
[22] PID at 4.
[23] PID at 7.
[24] U.S. Dep't of Agriculture, APHIS, Wildlife Services, The Use of Sodium Cyanide in Wildlife Damage Management (May 2017) at 22.
[25] *Id.*

made available for public review and comment. Comments regarding each component of the scientific assessment are included below.

1. Human Health Risks

At the outset, it is inappropriate that the Proposed Interim Decision and associated *Sodium Cyanide Human Health Assessment in Support of Registration Review*[26] (hereinafter "Human Health Assessment") largely ignore the predacide impacts of sodium cyanide on humans, and instead focus almost solely on the limited insecticide uses of the pesticide. This is in error. Humans have been — and absent cancellation, will continue to — be exposed to the lethal impacts of sodium cyanide use in M-44s. The extremely toxic impacts of this form of exposure — albeit unintended — must be fully analyzed and documented for public review.

The Human Health Assessment acknowledges that "[m]ost forms of cyanide are extremely toxic to mammals following acute exposure by the oral or inhalation routes."[27] Signs of cyanide poisoning include: "weakness and confusion; headache; nausea; metabolic acidosis; gasping for air in a manner similar to asphyxiation, but with a more abrupt onset; difficulty breathing; respiratory arrest; loss of consciousness; seizures prior to death; cardiac arrest and death."[28] A victim exposed to sodium cyanide via an M-44 in 2003 recently died as the result of his exposure.[29] Accidents and unintended exposure have occurred in the past, and absent cancellation, will likely occur again.[30]

The risk to human health and safety is real and apparent. The Administrator relies on the fact that only two human incidents were reported in the most recent five-year review (January 1, 2013 to April 16, 2018) to justify its conclusion that the low frequency of reported incidents does not suggest concern at this time.[31] As the agency itself admits, "severity" upon exposure is "extreme,"[32] and as such, even one incident of human death or injury alone should be considered frequent enough to justify concern for "unreasonable adverse effects." Even so, dozens of human exposures have occurred, and the Proposed Interim Decision must analyze the risks based on that data. Specifically, Wildlife Services mentions 42 human exposures from FY84 to FY15.[33]

Additionally, the Administrator should consider the efficacy of its reporting requirements and assess whether changes are necessary to ensure that all incidents are, in-fact, being properly reported. As explained in the "Ecological Incidents" section of the Proposed Interim Decision (and discussed further in our comments in the context of ecological impacts below), EPA's reporting requirements may not be sufficient to ensure all incidents are actually being reported. The risk of death of even one human should not be brushed off so lightly.

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[26] U.S. Envtl. Protection Agency, Memo: Sodium Cyanide, Human Health Risk Assessment in Support of Registration Review (Sept. 18, 2018) (Docket No. EPA-HQ-OPP-2010-0752-0092) [hereinafter "HHA"].
[27] HHA at 6.
[28] *Id.*
[29] *See* State of Utah, Certificate of Death, Dennis Ray Slaugh (State File No. 2018002960)(Feb. 24, 2018)(stating cause of death as "Acute Myocardial Infarction[;] Due to (or as a consequence of): Coronary Artery Disease[;] Other significant conditions: Cyanide Poisoning/ Exposure From M44 Device 2003")(attached).
[30] *See* Predator Defense, Featured Incidents of Pet Killings and Human Poisonings Caused by M-44s (Sept. 2018)(documenting human and pet M-44 incidents from April 1990 to February 2018) (attached).
[31] HHA at 9.
[32] *Id.* at 6.
[33] U.S. Dep't of Agriculture, APHIS, Wildlife Services, The Use of Sodium Cyanide in Wildlife Damage Management (May 2017) at 23.

Accordingly, we respectfully request that the Administrator include additional data regarding the human health impacts from accidental exposure to sodium cyanide via M-44s to complete its analysis in the Human Health Assessment prepared as part of the pending registration review. Further, we request that the Administrator elect not to reregister, and instead commence cancellation proceedings for, the registrations of sodium cyanide because the pesticide presents unreasonable adverse effects to humans and the environment.

2. Ecological Risks

The Administrator's "Ecological Risks" section of the Proposed Interim Decision and associated *Sodium Cyanide: Preliminary Ecological Risk Assessment to Support the Registration Review of Sodium Cyanide*[34] (hereinafter "Draft Risk Assessment") are also lacking. We respectfully refer the Administrator to the contents of our August 2017 petition (pages 12–19) which outlines ample factual support for the conclusion that M-44 use has unreasonable adverse effects on the environment by harming non-target wildlife, federally protected threatened and endangered species, and people and companion animals, and fully incorporate that information herein.

In addition, we express our sincere concern that the Administrator seems to be drawing faulty conclusions regarding the number and type of ecological incidents based on inaccurate data as the result of flawed registrant reporting requirements. The EPA itself acknowledges the deficiency at issue:

> Incident reports for nontarget organisms typically provide information only on mortality events and plant damage incidents. EPA's changes in the registrant reporting requirements for incidents in 1998 may account for a reduced number of reported incidents . . . Since 1998, registrants are only required to submit detailed information on 'major' fish, wildlife, and plant incidents. Sodium cyanide incidents generally involve unintended single animal deaths and the intended target animals are terrestrial wildlife. *Single animal deaths are not considered "major" incidents and may not be reported under current reporting requirements*.[35]

The fact that single animal deaths are not considered "major" incidents worthy of reporting is entirely inappropriate, especially considering the purpose and manner in which the M-44 device is used: the M-44 device is designed to trigger and kill a single animal at a time. Based upon current reporting requirements, therefore, the registrant is not required to report most ecological incidents involving unintended sodium cyanide exposure from M-44s. This is wholly inadequate and renders any conclusions drawn from this flawed data as entirely failing and unsupportable. We request the Administrator use the pending registration review process to remedy this clear deficiency by implementing a risk mitigation provision requiring registrants to fully account for all M-44s placed in the field by documenting the total number of incidents (intended and unintended, and including single and multiple animals) of sodium cyanide injury or death resulting from M-44 use.

[34] U.S. Envtl. Protection Agency, Memo: Sodium Cyanide, Draft Risk Assessment to Support the Registration Review (Sept. 12, 2018) (Docket No. EPA-HQ-OPP-2010-0752-0094) [hereinafter "DRA"].
[35] PID at 11–12 (emphasis added).

Even given the weak reporting requirements, the Incident Data System still catalogs 114 reported ecological incidents between 1978 and 2017.[36] These include numerous protected species, including wolves and eagles. That is unacceptable.

The Proposed Interim Decision discusses a 2017 report from APHIS and summarizes the nontargets killed by M-44s. It states that the devices killed "10 endangered kit foxes."[37] However, the 2017 report provides that the kit foxes are "Unlisted subspecies or DPSs."[38] Please resolve this inconsistency.

The risk of secondary toxicity provides additional justification for cancelling the registration. The Proposed Interim Decision explains that a raven died after feeding on an opossum poisoned by an M-44.[39] Undoubtedly, additional incidents occur but are not discovered or reported. Nevertheless, applicators check the devices just once per week, which means that poisoned carcasses remain in the environment and available for scavengers to be poisoned through secondary toxicity.

Further, we express our concern that the conclusions drawn in the Draft Risk Assessment were largely ignored in the Proposed Interim Decision. For example, the EPA states:

> Once the [M-44] device is activated and the animal exposed, likelihood of mortality is high. This is confirmed by numerous incident reports with wildlife and domestic canids which have encountered the devices, *despite the use restrictions designed to prevent these non-target exposures*. In addition, USDA records regarding actual results of the registered use of these M-44 devices from 2011 to 2015 *do* indicate that non-target birds and mammals were found dead near triggered devices.[40]

The Draft Risk Assessment further explains that the devices kill the targeted wild canid species only about half of the time (53 percent).[41] In fact, the science shows that 18 nontarget species visited the devices as often as coyotes, which were targeted.[42] This shows that M-44s are indiscriminate killers that pose too high a risk to nontarget wildlife.

Moreover, about one-third of the time the device fires, no dead bodies are recovered (9,759 out of 24,059 total firings in a five-year period).[43] The science shows that "[o]nce the device is activated and the animal exposed, the likelihood of mortality is high."[44] So for the remaining firings, it is likely that the animals wandered offsite and died, or died and were moved offsite by scavengers[45] These data further support that M-44s pose an unreasonable risk to nontarget wildlife.

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[36] DRA at 11.
[37] PID at 12.
[38] U.S. Dep't of Agriculture, APHIS, Wildlife Services, The Use of Sodium Cyanide in Wildlife Damage Management (May 2017).
[39] PID at 12.
[40] DRA at 15 (emphases added).
[41] *Id.* at 4.
[42] *Id.* at 12.
[43] *Id.* at 4.
[44] *Id.* at 15.
[45] DRA at 12.

Yet, in the Proposed Interim Decision, the Administrator — rather than acknowledge that the use restrictions do not prevent non-target exposures — proposes only to modify some of the use restrictions as a means to address some of the concerns presented by continued M-44 use.

While we appreciate the Administrator's efforts to implement at least minimal additional restrictions on M-44 use (additional comments on the proposed modifications and additions below), the evidence before the agency shows that use restrictions alone are not sufficient to protect the public and the environment from the unreasonable adverse effects of sodium cyanide.

Accordingly, we respectfully request that the Administrator elect not to reregister, and instead commence cancellation proceedings, for the registered use of sodium cyanide in M-44s.

3. Benefits Assessment

The Administrator's "Benefits Assessment" and sole reliance on the decade-old *Analysis of the Role of the M-44 Device and Compound 1080 Livestock Protection Collars in Predator Management*[46] memo (hereinafter "2009 M-44 Role Analysis Memo") to draw its vague conclusion that the benefits of continued M-44 use "clearly" outweighs the adverse risks of continued use to the public and the environment is severely lacking.

First, the Administrator's reliance solely on the 2009 M-44 Role Analysis Memo is in error. Ample advances in technology, science, and research relating to livestock coexistence practices and non-lethal predator damage control techniques have been made in the past ten years and it is disingenuous for the agency to conclude that merely "[n]o new information is available, and conclusions described in the docket still stand."[47]

Further, even the conclusions drawn in the 2009 M-44 Role Analysis Memo are largely without merit. After describing in detail, the increased efficacy of non-lethal predator control techniques[48] and the declining role of sheep production in the United States more generally,[49] the memo baldly concludes:

> Overall, it is evident that M-44 and [Livestock Protection Collar] devices provide benefits to livestock producers … and if users had to rely on the available alternatives, they *would likely* incur high costs and more predation. In light of the competitive nature of the industry, such a change *could* force producers out of business.[50]

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[46] U.S. Envtl. Protection Agency, Biological and Economic Analysis Division, Memo: Analysis of the Role of the M-44 Device and Compound 1080 Livestock Protection Collars in Predator Management (DP # 356680 and # 356681) (Jan. 6, 2009) (Docket No. EPA-HW-OPP-2010-0752-0027) [hereinafter "M-44 Role Analysis Memo"].

[47] PID at 12; *See e.g.*, Shivik et al. 3003, Lance et al. 2010; Berger 2006, Harper et al. 2008, Musiani et al. 2003, Wielgus and Peebles 2014, Treves et al. 2016 (studies documenting the efficacy of nonlethal alternatives and/or the inefficacy of lethal predator management) (cited in and attached to 2017 petition).

[48] *See* 2009 M-44 Role Analysis Memo at 25–34 (describing the effectiveness of guard animals, vigilance (herding), exclusion (fencing), deterrents, and husbandry management practices in providing non-lethal predator control options).

[49] *Id.* at 5–8 (acknowledging that the overall decline in the sheep industry can be attributed more to changes in consumers' tastes and preferences and increased land and hay prices rather than predation).

[50] *Id.* at 36 (emphasis added).

These conclusions are mere speculation. Even the agency itself acknowledges that it "is unable to quantify" the impacts "or estimate the number of operations that might be impacted" by the elimination of M-44 devices from the predator management toolbox.[51] The agency states that while it "can characterize certain situations," it "is unable to draw clear conclusions as to the economic feasibility of particular predator control techniques."[52] And it acknowledges that "[a] primary uncertainty in this analysis is the difficulty in fully accounting for the costs of various control measures, which include direct and indirect costs."[53] This uncertainty cannot be used to justify the continued use of these dangerous devices when alternatives exist. Indeed, the agency offers no explanation for how a ban could "force some producers out of business."[54] A ban would place all producers in the same position of relying on alternatives, including non-lethal alternatives that have been proven more effective. Given the documented risks and alternatives, the agency cannot reasonably conclude that continued registration of M-44 devices is economically and socially beneficial to the nation at large.

Additionally, the agency flatly acknowledges the severe risks and inefficacies associated with the M-44 device, noting that the device is "not selective, as it will kill any canid predator attracted to the scent whether it is the offending predator or not," and further, that "[t]hese devices may fail at times" and "require frequent inspections and regular maintenance."[55] The agency warns that M-44s demand regular maintenance "as these devices are subject to mechanical malfunctions,"[56] and that there are major downsides to M-44 use:

> There are several drawbacks to the use of M-44 devices. First, M-44 devices can sometimes malfunction or even discharge in the absence of predators, which can lead to accidental takes. Second, the bait may attract and be activated by nontarget carnivores, such as domestic dogs, foxes, or wolves. Non-canids have also been found to pull M-44s.[57]

The Administrator's reliance on blanket assertions that the sheep and cattle industry's monetary value overall renders M-44 use as necessarily beneficial is misplaced.[58] While we agree that a suite of predator management tools must be employed to effectively manage a sheep or cattle operation, we do not agree that the role of M-44s cannot be replaced by the ample array of alternative predator control techniques currently available. The Administrator has provided no scientific or economic support to conclude that M-44 use is beneficial overall.

Accordingly, we request that the Administrator elect not to reregister, and instead commence cancellation proceedings, for the registered use of sodium cyanide in M-44s.

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51 2009 M-44 Role Analysis Memo at 9; PID at 12–13.
52 2009 M-44 Role Analysis Memo at 15.
53 *Id.* at 15.
54 PID at 13.
55 2009 M-44 Role Analysis Memo at 13.
56 *Id.* at 20.
57 *Id.* at 21.
58 PID at 13.

C. Proposed Risk Mitigation and Use Restriction Modifications

We appreciate that the Administrator is taking at least some interim action to reduce risk to the public and non-target wildlife from the harmful impacts of M-44s by proposing label changes that would become effective shortly after publication of an Interim Decision.[59] While we maintain that the Administrator should instead — and ultimately — commence cancellation proceedings for sodium cyanide's registered use in M-44s, in the alternative, we generally support the proposed risk mitigation label changes and offer some additional modifications for the agency to consider.

That said, we note that the lack of enforcement and assurance that label and use restrictions are being followed in the field are a sincere concern to our organizations and many members of the public. Absent proper enforcement, there is no way to ensure that M-44s are being used only in a manner that is consistent with the use restrictions associated with their registrations. As indicated in the August 2017 petition (pages 19-21), many of the harmful incidents adversely affecting humans and their companion animals recently have resulted from M-44 misuse and applicator failure to follow use restrictions. This is a serious and widespread problem.

The EPA's own Draft Risk Assessment acknowledges that non-target exposure consistently occurs despite use restrictions designed to prevent incidental exposure being in place.[60] It provides that "[d]espite these restrictions, incident data have confirmed that endangered mammal and bird species have been killed when encountering M-44 devices."[61] Because past experience shows that even broad restrictions prohibiting use where endangered wildlife might encounter them cannot prevent deaths of endangered wildlife, the devices pose an unreasonable risk and must be cancelled.

Accordingly — and absent commencing cancellation proceedings altogether — we request that the Administrator implement some form of enforcement assurance concurrent with the proposed interim risk mitigation measures.

Proposed General Modifications

We agree with the proposed change for all labels that the word "must" should be used in replacement of the word "shall."[62]

Further, to reduce nontarget exposure, the devices should be modified to require application of more pounds of pressure before firing. Currently, only roughly four pounds of pressure is needed to activate the devices.[63] But that level of pressure allows numerous nontarget deaths of smaller animals, such as raccoons, opossums, skunks and more. Requiring more pressure for activation would reduce nontarget deaths and increase the specificity to canids. The EPA should analyze whether additional modifications to the devices could be made to reduce nontarget exposure.

We further support the proposed restriction changes as follows:

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[59] PID at 13.
[60] DRA at 15.
[61] *Id.* at 4.
[62] PID at 13.
[63] DRA at 9.

Proposed Restriction 8 Modification

We support the revised language generally prohibiting placement of M-44s near occupied residences (except for the residence of the person who requested placement of the device). And we support the restriction that "[w]ithin properties where its use is authorized, the M-44 device must not be used in areas where exposure to the public and family and pets is probable." Indeed, members of the public should not be subject to the risk of death or injury from sodium cyanide exposure while enjoying their neighborhoods and surrounding public lands. For this reason, we are concerned that a 0.25-mile radius is far too small to prevent such exposure and request an increase to at least 1 mile.

It follows that the notification buffer should be increased to between 1 and 3 miles (reading "M-44s cannot be placed between 1 and 3 miles of a residence other than that belonging to the cooperator unless the owner or lessee occupying the residence has been notified beforehand."). Additionally, — and short of a complete ban on public lands — the devices should not be placed in any public area if an adjacent landowner objects within one-week of receiving the notification.

Proposed Restriction 10 Modification

We support the revised language requiring at least one person other than the applicator have knowledge of the exact placement of all M-44 devices in the field. We further request that all M-44s placed in the field be marked and documented with GPS coordinates and that all interested members of the public be informed of the exact coordinates of M-44s placed within 3 miles of their homes. The exact locations of M-44s, as well as their current status upon at least weekly field checking, should be recorded and updated on a website, as well as physically posted in a widely accessible public place nearby, so that interested members of the public can be constantly informed that these deadly devices present potential risk to people and companion animals in the area. We also suggest that notification should be rendered prior to placement to ensure public safety and alleviate the risk of accidental exposure.

Proposed Restriction 12 Modification

We support the revised language clarifying that the water buffer applies even if the water body is frozen.

Proposed Restriction 14 Modification

We support the revised language increasing the buffer for M-44 placement from 50 to 100 feet from public roads or pathways. We further request that M-44s be banned from public lands and authorized for use only on private lands with permission of the landowner and all nearby residents. Members of the public should not be exposed to the deadly risk of sodium cyanide exposure from M-44s while recreating, working on, or simply enjoying our public lands, whether federal, state, municipal, county, or local government public lands.

Proposed Restriction 18 Modification

We support the revised language clarifying that an applicator should be responsible for inspecting M-44s at least once per week, if not daily. We further suggest that the applicator be accompanied by another person to verify the exact location and status of each device to add further accountability.

Alternatively, the EPA should consider requiring applicators to use a remote monitoring system that shows when the device has been triggered to decrease the risk of secondary toxicity to nontarget scavengers.

Proposed Restriction 21 Modification

We support the revised language requiring M-44 devices be stored under lock and key even when in transit.

Proposed Restriction 23 Modification

We support the revised language relating to warning signs and their proper placement as freestanding signs even if no fence line or visible boundary line exists. We also agree that additional elevated warning signs should be placed within at least 15 feet of the device. We further request that the devices be additionally marked by a visible flag or brightly colored stake that is at least 2 feet tall within 1 foot of the device to enhance public notification and reduce exposure risk. The flag or stake should include the international symbol for hazard or poison as well; for example:

Several accidental exposures to people inadvertently encountering the devices demonstrates the necessity of such a warning. Wildlife Services reports that five people have been injured after unknowingly stepping on the devices.[64] Consider also that the incident in Pocatello, Idaho — where a teenager was temporarily blinded after touching the device — may not have occurred if the device was properly labeled as poison.

Proposed Restriction 26 Modification

We support the revised language requiring more detailed recordkeeping to document both date of placement and date of removal for each device. We further request that additional information be recorded documenting each inspection of the device. Weekly recordkeeping reports should document whether the device had been triggered and the resulting injury, death, or no-impact that occurred, as well as any visible animal tracks or human footprints that ventured near the device. All records should be made readily available to the public in a timely manner through a website, as explained above, so people can know the real-time status of active M-44 devices in their area. All Program Accountability Reports or similar reports that may be required by state and local governments documenting Wildlife Services' M-44 use should also be made readily available to the public, both upon request and online.

[64] U.S Dep't of Agriculture, APHIS, Wildlife Services, The Use of Sodium Cyanide in Wildlife Damage Management (May 2017) at 23.

IV. Conclusion

Because M-44s are highly toxic, deadly devices that pose unreasonable risks of adverse effects to humans, non-target species, and the environment, we respectfully request that the EPA not reregister and, instead, commence cancellation proceedings for sodium cyanide registrations authorizing use in the lower 48 states during the pending registration review process. In the alternative, we request that the EPA impose additional, stricter use restriction modifications than those put forward in the Proposed Interim Decision.

Respectfully Submitted,

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TO THE U.S. ENVIRONMENTAL PROTECTION AGENCY

PETITION TO CANCEL REGISTRATIONS OF M-44 CYANIDE CAPSULES (SODIUM CYANIDE)

EPA REGISTRATION NOS. 56228-15, 35978-1, 35975-2, 39508-1, 33858-2, 13808-8 & CA840006

Photo by Tom Koerner, USFWS.

AUGUST 2017

AUTHORED BY:

WILDEARTH GUARDIANS
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Via Electronic and Certified Mail

August 10, 2017

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Dear Administrator Pruitt, Acting Assistant Administrator Cleland-Hamnett, and Acting Director Keigwin,

WildEarth Guardians, the Center for Biological Diversity, and several other wildlife and animal protection organizations seek a ban on use of M-44 cyanide capsules (sodium cyanide) in the lower 48 states. Sodium cyanide is a highly toxic pesticide registered for restricted use under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. §§ 136 et seq.[1] Sodium cyanide is used in M-44 ejector devices — also known as "cyanide bombs" — to kill coyotes (*Canis latrans*), red fox (*Vulpes vulpes*), gray fox (*Urocyon cinereoargenteus*), and wild dogs suspected of preying on livestock.

Because of the dangers posed by sodium cyanide to wildlife and people, we hereby petition the U.S. Environmental Protection Agency (EPA), with respect to sodium cyanide registrations authorizing use in the lower 48 states, to: (1) Cancel all active and pending

[1] Petitioners request action be taken to cancel all active registrations for M-44 cyanide capsules (sodium cyanide) in the lower 48 states and hereinafter reference all active registrations collectively when using the term "sodium cyanide" or "M-44 devices," including EPA Registration No. 56228-15 (APHIS), EPA Registration No. 35978-1 (Wyoming), EPA Registration No. 35975-2 (Montana), EPA Registration No. 39508-1 (New Mexico), EPA Registration No. 33858-2 (Texas), EPA Registration No. 13808-8 (South Dakota), and EPA Registration No. CA840006 (Sodium Cyanide).

registrations for sodium cyanide pursuant to FIFRA § 136d(b); (2) Suspend all sodium cyanide registrations pending completion of cancellation proceedings pursuant to FIFRA § 136d(c)(1); (3) Invoke a stop order prohibiting all current and future use of sodium cyanide effective immediately pursuant to FIFRA §§ 136k, 136j(a)(2)(G); and (4) Initiate Special Review proceedings for all sodium cyanide registrations pursuant to 40 C.F.R. Part 154. Thank you for your consideration. We look forward to your timely response.

Respectfully submitted,

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I. INTRODUCTION

The Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. § 136 et seq., provides the framework for federal regulation of pesticide use, sale, and distribution. The law is intended to prohibit the use of pesticides that cause unreasonable adverse effects on the environment.[2] The Administrator of the EPA is responsible for carrying out the mandates of the Act.[3] Pursuant to this obligation, the Administrator may limit the use of certain pesticides to prevent unreasonable adverse effects on the environment.[4]

M-44 cyanide capsules (containing a pesticide called sodium cyanide) are registered for restricted use under FIFRA (EPA Registration No's. 56228-15, 35978-1, 35975-2, 39508-1, 33858-2, 13808-8, and CA840006). Wildlife Services, a program of the U.S. Department of Agriculture, Animal and Plant Health Inspection Service (APHIS), is a registered user of sodium cyanide (EPA Registrant No. 56228-15). Other registered users include Wyoming Dept. of Agriculture (No. 35978-1), Montana Dept. of Agriculture (No. 35975-2), New Mexico Dept. of Agriculture (No. 39508-1), Texas Dept. of Agriculture (No. 33858-2), and South Dakota Dept. of Agriculture (No. 13808-8). This Petition hereby requests that the Administrator use his authority to prohibit use of sodium cyanide in the lower 48 states pursuant to FIFRA and the Act's implementing regulations. With respect to the lower 48 states, we request the Administrator: (1) Cancel all active and pending registrations for sodium cyanide pursuant to FIFRA § 136d(b); (2) Suspend all sodium cyanide registrations pending completion of cancellation proceedings pursuant to FIFRA § 136d(c)(1); (3) Invoke a stop order prohibiting all current and future use of sodium cyanide effective immediately pursuant to FIFRA §§ 136k, 136j(a)(2)(G); and (4) Initiate Special Review proceedings for all sodium cyanide registrations pursuant to 40 C.F.R. Part 154.

M-44 Devices and Overview of Use

Sodium cyanide is the pesticide active ingredient used in M-44 devices, which are also known as "cyanide bombs." These devices are not actually bombs, however, because no explosives are used. Instead, an M-44 uses a spring-loaded device that is screwed or pushed into the ground. The device is topped with scented bait to lure animals (such as coyotes, foxes, and other canids) to bite. Once the animal's teeth clench on the bait, a spring shoots a pellet of sodium cyanide into the animal's mouth.

The sodium cyanide combines with available moisture including saliva to make hydrogen cyanide gas, which is readily absorbed by the lungs and poisons the animal by inactivating an enzyme essential to mammalian cellular respiration.[5] That quickly leads to central nervous system depression, cardiac arrest, and respiratory failure.[6]

[2] 7 U.S.C. § 136a(a).
[3] 7 U.S.C. § 136(b).
[4] 7 U.S.C. §§ 136a(c)(5)-(6).
[5] U.S. Fish & Wildlife Service, *Biological Opinion: Effects of 16 Vertebrate Control Agents on Endangered and Threatened Species* (1993) at II-73 [hereinafter "1993 BiOp"].
[6] *Id.* at II-73.

Sodium cyanide is a Category 1 toxicant according to the EPA: the most acute, due to the imminent harm it poses to the environment and to humans.[7] Sodium cyanide is highly soluble in water and highly toxic to most aquatic organisms, and as a result, M-44 capsules may not be used within 200 feet of water.[8]

Wildlife Services and state agencies use M-44s in locales across the country to kill so-called "nuisance" wildlife, including coyotes, gray foxes and red foxes, and free-roaming dogs.[9] M-44s containing sodium cyanide are deployed primarily by Wildlife Services; however, the following states also have authority for their use: South Dakota, Montana, Wyoming, New Mexico, and Texas.[10] According to its 2015 and 2016 data, Wildlife Services uses M-44s in the following states: Colorado, Idaho, Montana, North Dakota, Nebraska, New Mexico, Nevada, Oklahoma, Oregon, Texas, Utah, Virginia, West Virginia and Wyoming.[11]

Impacts of M-44s on Endangered Wildlife

In a 1993 Biological Opinion that analyzed the impacts of sodium cyanide on endangered wildlife, the U.S. Fish and Wildlife Service (FWS) found that any carrion-feeding animal able to activate the M-44 device is at risk. For that reason, FWS placed additional restrictions on use of M-44s to try to reduce the risk to wildlife protected under the Endangered Species Act.

In its 1994 Reregistration Eligibility Decision (RED) pertaining to the use of sodium cyanide capsules in M-44 units, EPA concluded that the M-44 did not pose unreasonable risks to humans or the environment if used in accordance with the 26 use restrictions listed on the label, plus language determined by the FWS to be needed to protect endangered species likely to be jeopardized by use of M-44s.[12]

That analysis by FWS and EPA is decades old. Since then, M-44s have killed numerous non-target, federally protected endangered animals. Even when M-44s are used as intended to kill coyotes and other canids, harm to the environment can occur because of the important ecosystem roles played by these animals.

Availability of Viable Alternatives

The balance of interests clearly weighs in favor of prohibiting M-44s given the numerous viable alternatives to protect livestock from predation. For example, guard animals (including dogs, llamas, and donkeys) can be deployed, herders and range riders can

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[7] U.S. Environmental Protection Agency, Reregistration Eligibility Decision (R.E.D.) Facts: Sodium Cyanide (1994) *available at* https://archive.epa.gov/pesticides/reregistration/web/pdf/3086fact.pdf.
[8] 1993 BiOp at II-73.
[9] 1993 BiOp at II-73.
[10] 1993 BiOp at II-73.
[11] U.S. Dep't of Agriculture, Wildlife Services, *2016 Program Data Reports*, *available at* https://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/sa_reports/sa_pdrs/ct_pdr_home_2016; U.S. Dep't of Agriculture, Wildlife Services, *2015 Program Data Reports*, *available at* https://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/sa_reports/sa_pdrs/ct_pdr_home_2015.
[12] 1993 BiOp at II-74.

be employed, and livestock operators can change animal husbandry practices to lessen the risk of predation. Deterrents, such as sound- and light-emitting frightening devices, can also be used to scare away potential predators.

In short, a number of viable alternative tools to address livestock conflicts exist, eliminating the need for M-44 sodium cyanide capsules altogether.

II. PETITIONERS

WILDEARTH GUARDIANS is a non-profit 501(c)(3) organization dedicated to protecting and restoring the wildlife, wild places, wild rivers, and health of the American West. Guardians has over 215,000 activists and members supporting their efforts to end government-funded programs of cruelty to native wildlife.

The CENTER FOR BIOLOGICAL DIVERSITY is a non-profit 501(c)(3) organization with over 48,500 active members and 1.3 million supporters. The Center and its members are concerned with the conservation of imperiled species and the effective implementation of the ESA. Recognizing that pesticides are one of the foremost threats to the earth's environment, biodiversity, and public health, the Center works to prevent and reduce the use of harmful pesticides and to promote sound conservation strategies.

ADVOCATES FOR THE WEST is a non-profit organization protecting and defending public lands, wildlife, watersheds and air through litigation and negotiation.

The ANIMAL LEGAL DEFENSE FUND's mission is to protect the lives and advance the interests of animals through the legal system.

BORN FREE USA, a non-profit 501(c)(3) organization, believes that every animal matters. Inspired by the Academy Award® winning film *Born Free*, the organization works locally, nationally, and internationally to end wild animal cruelty and suffering, and protect threatened wildlife. Born Free USA also operates one of the country's largest wildlife sanctuaries.

The ENDANGERED SPECIES COALITION is a 501(c)(3) organization working to stop the human-caused extinction of our nation's at-risk species, to protect and restore their habitats, and to guide these fragile populations along the road to recovery. The Coalition is a network of conservation, scientific, education, religious, sporting, outdoor recreation, business and community organizations — and more than 150,000 individual activists and supporters — all dedicated to protecting our nation's disappearing wildlife and last remaining wild places.

The HUMANE SOCIETY OF THE UNITED STATES ("The HSUS") is among the nation's largest animal protection organizations, headquartered in Washington, D.C. Since its establishment in 1954, The HSUS has worked to combat animal abuse and exploitation and promote the welfare of all animals. In particular, The HSUS works extensively to promote the conservation of native carnivores through research, public outreach and education, advocacy and litigation. The HSUS has long advocated humane, non-lethal alternatives to cruel killing techniques including steel-jawed, leg-hold traps, strangling neck snares and the use of poisons such as sodium cyanide.

The INTERNATIONAL FUND FOR ANIMAL WELFARE's mission is to rescue and protect animals around the world. The organization rescues individuals, safeguards populations, and preserves habitat.

The NATURAL RESOURCES DEFENSE COUNCIL (NRDC) is an international nonprofit organization with more than 2 million members and online activists. Since 1970, our lawyers, scientists, and other environmental specialists have worked to protect the world's natural resources, public health, and the environment.

PREDATOR DEFENSE is a national non-profit advocacy organization working to protect native predators and end America's war on wildlife. Our efforts take us into the field, onto America's public lands, to Congress, and into courtrooms.

PROJECT COYOTE is a national non-profit organization and a North American coalition of wildlife educators, scientists, ranchers, and community leaders promoting coexistence between people and wildlife, and compassionate conservation through education, science, and advocacy.

PUBLIC EMPLOYEES FOR ENVIRONMENTAL RESPONSIBILITY (PEER) is a non-profit organization protecting public employees who protect our environment. PEER serves professionals who uphold environmental laws so that public servants may work as "anonymous activists," and their agencies must confront the message, not the messenger.

The SIERRA CLUB is one of America's largest and most influential environmental organizations, with more than 3 million members and supporters. In addition to helping people from all backgrounds explore nature and our outdoor heritage, the Sierra Club works to promote clean energy, safeguard the health of our communities, protect wildlife, and preserve our remaining wild places through grassroots activism, public education, lobbying, and legal action.

The SOUTHWEST ENVIRONMENTAL CENTER works to protect and restore native wildlife and their habitats in the Southwest.

The WESTERN ENVIRONMENTAL LAW CENTER uses the full power of the law to defend and protect the American West's treasured landscapes, iconic wildlife, and rural communities.

WESTERN WATERSHEDS PROJECT is a non-profit environmental group working to protect and restore western watersheds and wildlife.

The mission of WILDLANDS NETWORK is to reconnect, restore and rewild North America so that the diversity of life can thrive. The organization envisions a world where nature is unbroken, and where humans co-exist in harmony with the land and its wild inhabitants.

The WOLF CONSERVATION CENTER (WCC) is an environmental education organization committed to conserving wolf populations in North America through science-based education programming and participation in the federal Species Survival Plans for the critically endangered Mexican gray wolf and red wolf. Through wolves, the WCC teaches the broader message of conservation, ecological balance, and personal responsibility for improved stewardship of our World.

III. LEGAL BASIS FOR PETITIONING

Cancellation, suspension, issuance of a stop order, and initiation of a Special Review for all sodium cyanide registrations in the lower 48 is appropriate at this time pursuant to FIFRA and its implementing regulations.

First, cancellation of a pesticide's registration is warranted where the pesticide, "when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment."[13] Here, the registration for sodium cyanide must be cancelled because, as documented below, its continued use is causing unreasonable adverse effects on the environment, members of the public, and non-targeted companion animals.

Second, suspension of a pesticide's registration is warranted under FIFRA § 136d(c)(1) when such action is necessary to prevent an imminent hazard[14] during the time required for cancellation.[15] Here, as documented below, the registration for sodium cyanide should be suspended pending cancellation proceedings to prevent an imminent hazard to the environment and protected species.

Third, a "stop sale, use, or removal" order pursuant to FIFRA § 136k is appropriate when a registered pesticide is being used in an unlawful manner.[16] As documented below, evidence suggests that sodium cyanide — a restricted use pesticide — is being used in

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[13] 7 U.S.C. § 136d(b); *see also id.* § 136(bb) (providing that "[t]he term 'unreasonable adverse effects on the environment' means (1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide ….").

[14] 7 U.S.C. § 136(l) ("The term 'imminent hazard' means a situation which exists when the continued use of a pesticide during the time required for cancellation proceeding would be likely to result in unreasonable adverse effects on the environment or will involve unreasonable hazard to the survival of species declared endangered or threatened by the Secretary pursuant to the Endangered Species Act of 1973 [16 U.S.C. 1531 et seq.].").

[15] 7 U.S.C. §136d(c)(1) ("If the Administrator determines that action is necessary to prevent an imminent hazard during the time required for cancellation … the Administrator may, by order, suspend the registration of the pesticide immediately.").

[16] 7 U.S.C. § 136k(a) ("Whenever any pesticide or device is found by the Administrator in any State and there is reason to believe on the basis of inspection or tests that such pesticide or device is in violation of any of the provisions of this chapter … or when the registration of the pesticide has been canceled by a final order or has been suspended, the Administrator may issue a written or printed 'stop sale, use, or removal' order to any person who owns, controls, or has custody of such pesticide or device ….").

violation of the pesticide's use restrictions, and thereby, its labeling requirements, which is unlawful under FIFRA § 136j(a)(2)(G).[17]

Fourth, the Administrator may initiate a Special Review pursuant to 40 C.F.R Part 154 when one or more of the risk criteria of 40 C.F.R § 154.7 are met.[18] As evidenced below, the Administrator may find that multiple risk criteria triggering such Special Review for sodium cyanide registrations are present.[19] For example, continued sodium cyanide use: "[m]ay pose a risk of serious acute injury to humans or domestic animals[;]" "[m]ay pose a risk to the continued existence of any endangered or threatened species designated by the Secretary of the Interior or the Secretary of Commerce under the Endangered Species Act of 1973, as amended[;]" and "[m]ay otherwise pose a risk to humans or to the environment which is of sufficient magnitude to merit a determination whether the use of the pesticide product offers offsetting social, economic, and environmental benefits that justify initial or continued registration."[20]

IV. FACTUAL AND SCIENTIFIC SUPPORT FOR PETITION

M-44 Use has Unreasonable Adverse Impacts on the Environment and Presents an Imminent Hazard

Evidence exists that past and present uses of sodium cyanide have unreasonable adverse impacts upon the environment and present an imminent hazard, as those terms are defined by FIFRA and the Act's implementing regulations.[21] M-44 use causes harm to non-target wildlife, federally protected threatened and endangered species, and people and companion animals. The harms caused by M-44 use are not outweighed by the benefits of continued use because viable alternatives exist.

<u>Impacts to Non-target Wildlife</u>

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[17] 7 U.S.C. § 136j(a)(2) (G)("It shall be unlawful for any person — … to use any registered pesticide in a manner inconsistent with its labeling.").
[18] *See* 40 C.F.R. § 154.1 ("The purpose of the Special Review process is to help the Agency determine whether to initiate procedures to cancel, deny, or reclassify registration of a pesticide product because uses of that product may cause unreasonable adverse effects on the environment, in accordance with sections 3(c)(6) and 6 of [FIFRA]. The process is intended to ensure that the Agency assesses risks that may be posed by pesticides and the benefits of use of those pesticides, in an open and responsive manner.").
[19] 40 C.F.R. § 154.7.
[20] 40 C.F.R. §§ 154.7 (1), (3), (4), (6).
[21] 7 U.S.C. § 136(bb) (providing that "[t]he term 'unreasonable adverse effects on the environment' means (1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide ….."); 7 U.S.C. § 136(l) ("The term 'imminent hazard' means a situation which exists when the continued use of a pesticide during the time required for cancellation proceeding would be likely to result in unreasonable adverse effects on the environment or will involve unreasonable hazard to the survival of species declared endangered or threatened by the Secretary pursuant to the Endangered Species Act of 1973 [16 U.S.C. 1531 et seq.]."). *See also Environmental Defense Fund, Inc. v. EPA*, 510 F. 2d 1292, 1297 (D.C. Cir. 1975) (upholding EPA suspension and cancellation order for aldrin and dieldrin and stating: "We have cautioned that the term 'imminent hazard' is not limited to a concept of crisis. 'It is enough if there is a substantial likelihood that serious harm will be experienced during the year or two required in any realized projection of the administrative process.'" (citing *Defense Fund, Inc. v. EPA*, 465 F.2d 528, 540 (D.C. Cir. 1972)).

M-44s are indiscriminate killers that are responsible for the deaths of thousands of non-target animals.

The U.S. Department of Agriculture's Animal Damage Control program (predecessor to APHIS-Wildlife Services) recorded 103,255 animals killed by M-44's between 1976 and 1986, including 4,868 non-target animals (approximately 5% of all animals killed).[22] Non-target species identified as having been killed by M-44s included grizzly bear, black bear, mountain lion, badger, kit and swift fox, bobcat, ringtail cat, feral cat, skunk, opossum, raccoon, Russian boar, feral hog, javelin, beaver, porcupine, nutria, rabbit, vulture, raven, crow, and hawk.[23] In addition, a California condor was found dead near the vicinity of an M-44 in 1986.[24]

A review of the Ecological Incident Information System in 2010 shows 45 terrestrial non-target animal incidents resulting from M-44 use from 1983-2009. The database records mortality for 26 birds, 15 dogs, ten wolves, three foxes, and two bears.[25]

According to Wildlife Services' most recent available data, from 2010-2016, over 2,600 animals were unintentionally taken by M-44s. For example, during that time period, Wildlife Services killed 882 non-target animals in Texas, 635 in Virginia, 336 in West Virginia, 315 in New Mexico, and 283 in Oklahoma.[26]

Wildlife Services' 2016 data shows that 321 animals were unintentionally killed by M-44s *in that year alone*.[27] Included among the non-targeted animals killed in 2016 were: 101 gray fox, 61 red fox, 57 raccoons, one black bear, one fisher, and seven domestic animals (such as family dogs). Such verified deaths almost certainly underestimate the total number of non-target species impacted because the likelihood of locating the carcass of a non-target species is small, especially with respect to small birds and small mammals.

More recently, in February 2017, a wolf died in northeastern Oregon from an M-44 used by Wildlife Services to target coyotes. In March 2017, in two separate incidents, M-44s temporarily blinded a child and killed three family dogs in front of their families in Idaho and Wyoming.

Impacts to Threatened and Endangered Species

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22 1993 BiOp at II-74.
23 *Id.*
24 *Id.*
25 Memorandum dated Sept. 20, 2010 from Valerie Wood, Biologist at the Environmental Fate and Effects Division of EPA, to Kathryn Jakob, Chemical Review Manager at EPA with attached draft "Problem Formulation for the Ecological Risk Assessment, of Sodium Cyanide (M-44)" at 12.
26 U.S. Dep't of Agriculture, Wildlife Services, *2016 Program Data Reports*, *available at* https://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/sa_reports/sa_pdrs/ct_pdr_home_2016 (last visited July 21, 2017).
27 U.S. Dep't of Agriculture, Wildlife Services, *Program Data Report G – 2016 Animals Dispersed/Killed or Euthanized/Removed or Destroyed/Freed*, *available at* https://www.aphis.usda.gov/wildlife_damage/pdr/PDR-G_Report.php?fy=2016&fld=KILLED_EUTH&fld_val=0 (last visited June 5, 2017).

M-44s also put federally protected threatened and endangered species at greater risk. Registered use of M-44s has unintentionally killed a threatened grizzly bear, endangered California condors, wolves and other species protected under the Endangered Species Act (ESA). M-44s placed in the habitat of Canada lynx, a threatened species under the ESA, or in the habitat of wolverine, a candidate species for ESA protection, further place these imperiled species at risk of extinction.

Specifically, according to documents received by the Center pursuant to the Freedom of Information Act, in 1978 a threatened grizzly bear in Montana died from an M-44. In 1983, an endangered California condor died from an M-44 in Kern County, California. In 1995, an endangered wolf in the panhandle of Idaho died from an M-44 set for coyotes. In March of 2001, an endangered wolf died from an M-44 in South Dakota. Two years later, in March of 2003, another wolf died in an undisclosed location. In March of 2005, a bald eagle, protected under the ESA at that time, died from an M-44 in McHenry County, North Dakota. In January of 2007, two wolves died from M-44s in Idaho near Riggins. In December of 2008, an endangered wolf was killed from an M-44 north of Cokeville, Wyoming, in Lincoln County. In May of 2013, a federally protected bald eagle died from an M-44 in Richland County, North Dakota.[28]

The number of federally-protected animals killed by M-44s are likely under-represented here as these incidents only reflect deaths reported to the EPA. Many killed animals are likely never discovered as they can die some distance from the M-44 device, and other animals could be discovered but not reported.

The incidents detailed here do not include other protected non-endangered wildlife, such as state-listed or "special concern" species, killed by M-44s. As just one additional example, a protected[29] wolf died in 2017 from an M-44 device in northeastern Oregon.[30]

Threats to People and Companion Animals

Sodium cyanide is a Category 1 toxicant because it is highly lethal to people and domestic animals in addition to native wildlife. M-44s put people and their companion animals unnecessarily at risk of being severely injured, or even killed.

In one tragic incident in March of 2017, a 14-year old boy was poisoned when he unsuspectingly tugged on an M-44 device while hiking behind his home in Idaho.[31] The boy watched in horror as his golden retriever convulsed and died within only minutes of the

[28] Incident reports and other documentation are on file with author Collette Adkins and included with this petition.
[29] Wolves throughout the State of Oregon are considered "a special status game mammal, protected by the Oregon Wolf Plan." Oregon Dep't of Fish & Wildlife, *Frequently Asked Questions about Wolves in Oregon*, http://www.dfw.state.or.us/Wolves/faq.asp (last visited Aug. 9, 2017).
[30] Oregon Dep't of Fish & Wildlife, *Press Release: Wolf Dies in Unintentional Take in Northeast Oregon* (Mar. 2, 2017) http://www.dfw.state.or.us/news/2017/03_mar/030217.asp.
[31] Cristina Corbin, *USDA Must Rethink Cyanide Bombs That Injured Boy, Killed Pets, Lawmaker Says*, FOX NEWS U.S. (Mar. 21, 2017) http://www.foxnews.com/us/2017/03/21/usda-must-rethink-cyanide-bombs-that-injured-boy-killed-pets-lawmaker-says.html.

device being activated. This incident sparked a public outcry,[32] led to a statewide moratorium, and the introduction of federal legislation[33] to ban the devices from further use nationwide. Sadly, this tragic incident is only one of many that have occurred in the past and are likely to occur in the future if the devices remain in use.

In another recent incident, in March of 2017, M-44s killed two family dogs while the family hiked together on a prairie on public lands in Wyoming.[34] That incident not only put the dogs at risk but also the family members who were exposed to sodium cyanide when they tried to save the dogs by washing them in a creek and when they hugged and kissed their beloved dying pets.

In 2016 alone, Wildlife Services admitted to unintentionally killing seven domestic animals with M-44s.[35] In addition, in 2016, Wildlife Services reported unintentionally killing 22 dogs that were classified as feral, free-ranging or hybrids. Many of these dogs were likely family dogs running off-leash. As of June, at least three domestic dogs were killed by M-44s in 2017.[36] Appendix B, which is attached, provides a list — compiled by Wildlife Services — of dogs unintentionally killed by M-44s.

A number of employees and unsuspecting members of the public have also been put at risk from sodium cyanide's toxic effects. The Center received documentation of several such incidents in response to a request under the Freedom of Information Act. For example, in December of 1999, a private landowner tried to remove an M-44 placed on property that he was leasing and accidentally triggered the device. He tasted the poison in his mouth and his wife drove him to the hospital, where he received medical attention. In November of 2002, a woman accidentally triggered an M-44 device placed on her property. She experienced increased respiratory rate and eye irritation but was able to drive herself to the hospital. In May of 2007, a person spraying for mosquitoes accidentally stepped on a M-44 device and sodium cyanide sprayed into his eyes causing burning and irritation, as well as disorientation. He received emergency medical assistance, and several others, including a county sheriff, came to the scene and had to shower because of exposure to sodium cyanide. In February of 2011, a border patrol agent in Kinney County, Texas, kicked and then tugged at an unknown object, which turned out to be a M-44. The device exploded in his gloved hands and he called an ambulance, which brought him to the hospital for medical attention.[37]

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32 Sarah V. Schweig, *Family's Dog Was Just Killed By This Tool — And the U.S. Government Put It There*, THE DODO (Mar. 20, 2017) https://www.thedodo.com/usda-m44-kills-idaho-dog-2322197701.html.
33 *See* Press Release: Rep. Peter DeFazio Introduces Legislation to Ban Lethal Poisons Compound 1080, Sodium Cyanide from Predator Control (Mar. 30, 2017) http://defazio.house.gov/media-center/press-releases/rep-peter-defazio-introduces-legislation-to-ban-lethal-poisons-compound.
34 http://www.predatordefense.org/features/m44_WY_Amy_dogs.htm
35 U.S. Dep't of Agriculture, Wildlife Services, *Program Data Report G – 2016 Animals Dispersed/Killed or Euthanized/Removed or Destroyed/Freed*, *available at* https://www.aphis.usda.gov/wildlife_damage/pdr/PDR-G_Report.php?fy=2016&fld=KILLED_EUTH&fld_val=0 (last visited June 5, 2017).
36 Cristina Corbin, *USDA Must Rethink Cyanide Bombs That Injured Boy, Killed Pets, Lawmaker Says*, FOX NEWS U.S. (Mar. 21, 2017) http://www.foxnews.com/us/2017/03/21/usda-must-rethink-cyanide-bombs-that-injured-boy-killed-pets-lawmaker-says.html.
37 Incident reports and other documentation are on file with author Collette Adkins and included with this petition.

Other reports of incidents have been gathered by the co-petitioning non-profit organizations, Predator Defense and The Humane Society of the United States. Dozens of these incidents are listed in Appendix A (attached). For example, in May of 2003, an M-44 device exploded and harmed a man who was rock hounding in Uintah County, Utah. His family did not know what hit him because of the lack of warning signs in the area. He immediately experienced disorientation and was unable to speak. His wife explains that he suffered for many years and had his life cut short because of the encounter.[38] Another incident involved a woman who was exposed to sodium cyanide after trying to resuscitate her dog, who died from an M-44 set on her land without her permission.[39] She immediately tasted the poison in her mouth and then felt disorientated. Over the next several months she experienced tingling in her arms and insomnia. Another incident involves a rancher who pulled on what he thought to be just a pipe sticking out of the ground but was actually an M-44 device that Wildlife Services set on his property without his permission.[40] When the device exploded, it badly cut and burned his hand. He experienced pain in his hand for several months during the slow healing process.

Several other reported incidents include pesticide applicators, which carry antidotes in case of sodium cyanide exposure. For example, in May 2001, an applicator accidentally triggered the device. He experienced temporary blindness in one eye, as well as blisters on his tongue and lips and went to the emergency room to receive medical attention. In January 2002, an applicator tried to cover an M-44 with a concrete block because he knew of hunting dogs in the area. He accidentally triggered the device and the sodium cyanide capsule hit his face and eye. He flushed his eyes and went to the hospital for medical attention. In March 2002, an applicator accidentally triggered an M-44 when he reached into a bucket in his vehicle that held the assembled device. He experienced burning of his eyes and could taste the poison in his mouth, and he drove himself to the emergency room, where he received medical assistance. In April 2005, an applicator accidentally triggered the device while installing it and administered the antidote. In January 2007, an applicator working on behalf of Wildlife Services in Oklahoma triggered an M-44. He experienced eye irritation and disorientation but was able to administer the antidote and drive himself to the hospital. In November 2008, an applicator accidentally triggered the device and the sodium cyanide capsule hit him in the face. After tasting the poison, he administered the antidote and went to the hospital for medical attention.[41]

Alternatives to Sodium Cyanide

M-44s are indiscriminate killing devices that are not needed in modern wildlife management because ample viable alternatives currently exist.

Numerous, proven effective and nonlethal methods of reducing conflicts with coyotes and other canids exist. For example, electric fences (that can be solar powered for use in remote areas), fladry (flags tied to ropes or fences), guard animals, range riders, strobe

[38] https://www.predatordefense.org/docs/m44_letter_Slaugh_DeFazio.pdf
[39] https://www.predatordefense.org/docs/m44_letter_Kingsley_DeFazio_01-09-07.pdf
[40] https://www.predatordefense.org/docs/m44_letter_Guerro_DeFazio.pdf
[41] Incident reports and other documentation are on file with author Collette Adkins and included with this petition.

lights and noisemakers can be used in lieu of M-44s to effectively deter coyotes and other so-called "problem wildlife" from disturbing livestock. Indeed, numerous studies have demonstrated the effectiveness of nonlethal methods to protect livestock from predators (e.g. Shivik et al. 2003[42]; Lance et al. 2010[43]).

Moreover, numerous scientific studies seriously call into question the efficacy of lethal predator control (e.g., Berger 2006[44], Harper et al. 2008[45]; Musiani et al. 2003[46]). For example, in a study based upon a review of 25 years of livestock depredation data, Wielgus and Peebles (2014)[47] found that with increased predator persecution, livestock losses *increased* in the following year. Additionally, Treves et al. (2016),[48] a meta-review of 24 studies, showed little or no scientific support for the efficacy of killing predators to protect livestock. Just as many livestock are likely to die, or in some cases even more, after predators are killed.

Scientists explain that indiscriminate killing of coyotes disrupts the stability and equilibrium of their social structure, triggering compensatory breeding and an increase in the coyote population.[49] Specifically, younger pairs begin to breed and juvenile males move in to fill the gap. Increasing the number of juvenile males in a destabilized population increases the likelihood of predation on wild ungulates and on livestock.[50]

While we do not condone — nor does the science support — the use of lethal techniques to control predators, even if Wildlife Services and state agencies insist on using lethal methods to target coyotes and other canids, more selective and more effective alternatives to M-44s are available. Firearms can be used with relatively minimal risk to people and non-targets as long as the shooter makes a positive identification before shooting. Traps, such as cage traps, can be used with specifications to reduce non-target

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[42] Shivik, J. A., A. Treves, and P. Callahan. 2003. *Nonlethal techniques for managing predation: Primary and secondary repellents*. CONSERVATION BIOLOGY 17: 1531-1537, *available at* http://wscinof.dreamhosters.com/wp-content/uploads/SHIVAKNon-Lethal.pdf.

[43] Lance, N.J., S.W. Breck, C. Sime, P. Callahan, and J.A. Shivik. 2010. *Biological, technical, and social aspects of applying electrified fladry for livestock protection from wolves* (*Canis lupus*). WILDLIFE RESEARCH 37: 708-714, http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=2257&context=icwdm_usdanwrc.

[44] Berger, K.M. 2006. *Carnivore-Livestock Conflicts: Effects of Subsidized Predator Control and Economic Correlates on the Sheep Industry*. CONSERVATION BIOLOGY 20: 751-761.

[45] Harper, E.K., W.J. Paul, and D.L. Mech, et al. 2008. *Effectiveness of lethal, directed wolf-depredation control in Minnesota*. JOURNAL OF WILDLIFE MANAGEMENT 72: 778–84.

[46] Musiani, M., C. Mamo, L. Boitani, C. Callaghan, C. C. Gates, L. Mattei, E. Visalberghi, S. Breck, and G. Volpi. 2003. *Wolf depredation trends and the use of fladry barriers to protect livestock in western North America*. CONSERVATION BIOLOGY 17: 1538-1547, http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1616&context=icwdm_usdanwrc.

[47] Wielgus, R. and K. Peebles. 2014. *Effects of Wolf Mortality on Livestock Depredations*. PLOS ONE 9: e113505, http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0113505.

[48] Treves, A., M. Krofel, J. McManus. 2016. *Predator control should not be a shot in the dark*. FRONTIERS IN ECOLOGY AND THE ENVIRONMENT 14: 380-388, *available at* http://faculty.nelson.wisc.edu/treves/pubs/Treves_Krofel_McManus.pdf.

[49] *See e.g.*, Letter from Dr. Robert Crabtree, Yellowstone Ecological Research Center (Revised Draft June 21, 2012), *available at* http://www.predatordefense.org/docs/coyotes_letter_Dr_Crabtree_06-21-12.pdf (presenting research showing that indiscriminate killing of coyotes results in population booms with consequent increases in livestock and wild ungulate predation).

[50] *Id.*

capture, and as long as traps are frequently checked (at least once every 24-hours), non-target animals may often be released without lethal injuries.

An analysis of Wildlife Services' own data demonstrates that alternatives to M-44s are more effective for capturing coyotes and other canids. For example, in 2015, Wildlife Services reportedly killed 68,905 coyotes. Wildlife Services killed just 18.7 percent of these coyotes using M-44s. Using the more effective — and more selective — technique of shooting coyotes with firearms, Wildlife Services killed 27,181 coyotes in 2015. That's nearly 40 percent of the total number of coyotes killed that year.[51] In short, given the alternatives to M-44s, continued M-44 use is economically unjustified.

Ecological Benefits of Conserving Predators Targeted by M-44s

Prohibiting the use of M-44s would benefit the health of ecosystems and native wildlife populations altogether. Carnivores targeted by M-44s, such as coyotes and foxes, play an essential role in maintaining healthy ecosystems. Predator species modulate prey populations and increase the health of those populations. The presence of carnivores on the landscape increases the biological diversity and overall functionality of ecosystems. Indeed, numerous studies analyze how carnivore removal, in particular, can cause a wide range of unanticipated impacts that are often profound, including on native plant communities, wildfire and biogeochemical cycles, the spread of disease or invasive species, and more (e.g. Beschta and Ripple 2009[52]; Levi et al. 2012[53]; Bergstrom et al. 2013[54]; Bergstrom 2017[55]).

Mesopredator species, like coyotes, are essential to maintaining ecological balance. Coyotes play a keystone role in the American West's native ecosystems by preying upon smaller carnivores such as skunks, foxes, and raccoons.[56] This predation indirectly benefits the prey of smaller carnivores. For instance, the resulting decreased nest predation by smaller carnivores increases ground-nesting birds like the imperiled greater sage grouse.[57] Coyotes also increase the diversity of rodent species by increasing the competition amongst smaller carnivores.[58]

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[51] U.S. Dep't of Agriculture, Wildlife Services, *2016 Program Data Reports*, *available at* https://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/sa_reports/sa_pdrs/ct_pdr_home_2016 (last visited July 21, 2017).

[52] Beschta, R.L., and W.J. Ripple. 2009. *Large predators and trophic cascades in terrestrial ecosystems of the western United States*. BIOL. CONSERV. 142(11): 2401–2414.

[53] Levi, T., A.M. Kilpatrick, M. Mangel, and C.C. Wilmers. 2012. *Deer, predators, and the emergence of Lyme disease*. PROC NATL ACAD SCI 109(27): 10942–10947.

[54] Bergstrom, B.J., L.C. Arias, A.D. Davidson, A.W. Ferguson, L.A. Randa, and S.R. Sheffield. 2014. *License to kill: reforming federal wildlife control to restore biodiversity and ecosystem function*. CONSERVATION LETTERS.

[55] Bergstrom, B.J. 2017. *Carnivore conservation: shifting the paradigm from control to coexistence*. J. MAMMAL. 98 (1): 1-6.

[56] Crooks, K.R. and M.E. Soule. 1999. *Mesopredator Release and Avifaunal Extinctions in a Fragmented System*. 400 J. NATURE 563–566; Henke, S.E. and F. C. Bryant. 1999. *Effects of Coyote Removal of the Faunal Community in Western Texas*. 63 J. WILDLIFE MGMT. 1066–1081.

[57] Mezquida, E.T. et. al. 2006. *Sage-Grouse and Indirect Interactions: Potential Implications of Coyote Control on Sage-Grouse Populations*. 108 J. CONDOR 747–759.

[58] Ripple, W.J. and R. L. Beschta. 2006. *Linking a Cougar Decline, Trophic Cascade, and Catastrophic Regime Shift in Zion National Park*. 133 J. BIOLOGICAL CONSERVATION 397–408.

In summary, the harms associated with continued use of M-44 sodium cyanide devices far outweigh the benefits of that use.

M-44s are Being Used Illegally, In Violation of Labeling Requirements and FIFRA

The labels[59] for registered sodium cyanide products require that users comply with all twenty-six use restrictions outlined in the Use Restriction Bulletin.[60] Even though FIFRA requires strict adherence to pesticide labels,[61] numerous incidents involving accidental exposure to sodium cyanide show that the registered users do not consistently abide by a number of these use restrictions.

The recent incidents in Idaho and Wyoming provide ample evidence demonstrating how registered users are violating the label requirements and other use restrictions when placing M-44s. The incident in Pocatello, Idaho involved an illegally-placed M-44 that injured a teen-aged boy, killed his dog and exposed several family members to sodium cyanide. Media reports and written accounts from the family demonstrate violations of the following use restrictions:

- "The M-44 device shall not be used: (1) in areas within national forests or other Federal lands set aside for recreational use, (2) areas where exposure to the public and family and pets is probable, (3) in prairie dog towns, or (4) except for the protection of Federally designated threatened or endangered species, in National or State Parks; National or State Monuments; federally designated wilderness areas; and wildlife refuge areas";[62]
- "Bilingual warning signs in English and Spanish shall be used in all areas containing M-44 devices . . . Main entrances or commonly used access points to areas in which M-44 devices are set shall be posted with warning signs to alert the public to the toxic nature of the cyanide and to the danger to pets. Signs shall be inspected weekly to ensure their continued presence and ensure that they are conspicuous and legible . . . An elevated sign shall be placed within 25 feet of each individual M-44 device warning persons not to handle the device"; and[63]
- "In all areas where the use of the M-44 device is anticipated, local medical people shall be notified of the intended use. This notification may be made through a poison control center, local medical society, the Public Health

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[59] *See e.g.,* Label for EPA Registration No. 56228-15 ("Users of this product must follow all requirements of product labeling, including but not limited to, all Use Restrictions, Directions for Use, Precautionary Statements, first aid and antidotal measures, information on endangered species, requirements for posting warning signs, and Storage and Disposal instructions."). See also the labels for EPA Registration No. 35975-2, EPA Registration No. 39508-1, EPA Registration No. 13808-8, EPA Registration No. 33858-2, and EPA Registration No. 35978-1.
[60] U.S. Dep't of Agriculture, Animal & Plant Health Inspection Service, *WS Directive 2.415, M-44 Use and Restrictions* (revised June 15, 2017) [hereinafter "M-44 Use Restrictions"] *available at* https://www.aphis.usda.gov/wildlife_damage/directives/2.415_m44_use%26restrictions.pdf.
[61] 7 U.S.C. § 136j(a)(2)(G).
[62] M-44 Use Restrictions at 3.
[63] *Id.* at 10–11.

> Service, or directly to a doctor or hospital. They shall be advised of the antidotal and first-aid measures required for treatment of cyanide poisoning. It shall be the responsibility of the supervisor to perform this function."[64]

It cannot be disputed that the M-44 was placed in an "area[] where exposure to the public and family and pets is probable." Fourteen-year-old Canyon Mansfield was walking the family Labrador, Casey, on a hill just 300 yards behind their home on public land managed by the Bureau of Land Management (BLM) in the outskirts of Pocatello, Idaho.[65] (That placement also violated a November 2016 pledge by Wildlife Services in Idaho not to use M-44s on public land in Idaho.[66])

As for the requirement for conspicuous warning signs, Dan Argyle, a captain in the Bannock County Sheriff's Office, told National Geographic that "no warning signs were observed at the scene . . . ."[67] And Canyon Mansfield explains: "No signs like these were near the cyanide bomb that took my dog away from me."[68]

It has been reported that Wildlife Services made no notifications of the intended use of M-44s to local medical professionals.[69] Canyon Mansfield's father, Dr. Mark Mansfield explains: "We didn't know anything about it. No neighborhood notifications, and our local authorities didn't know anything about them . . . The sheriff deputies who went up there didn't even know what a cyanide bomb was." The Center requested, under the Freedom of Information Act, copies of written materials serving as proof that the required notifications to medical professionals were made in Idaho. Responsive records indicate that Wildlife Services notified Idaho hospitals *after* the Pocatello incident, in July 2017, and that Wildlife Services has not made these notifications on an annual basis, as the prior notification to Idaho hospitals occurred in 2013.

The incident north of Casper, Wyoming that killed two family dogs also demonstrates a violation of the requirement for warning signs.[70] A media report provides that a "few days after the dogs died in Wyoming, Daniel Helfrick returned to the area, looking for signs they might have missed to warn them of the cyanide traps. He didn't see any."[71] A personal account of the tragic incident by one of the involved family members provides further evidence that no signs were posted.[72]

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[64] *Id.* at 12.
[65] http://news.nationalgeographic.com/2017/04/wildlife-watch-wildlife-services-cyanide-idaho-predator-control/.
[66] http://fox13now.com/2017/03/21/cyanide-bomb-that-killed-dog-owner-placed-illegally-by-wildlife-services/.
[67] http://news.nationalgeographic.com/2017/04/wildlife-watch-wildlife-services-cyanide-idaho-predator-control/.
[68] https://www.predatordefense.org/docs/m44s_canyons_story.pdf.
[69] http://www.theblaze.com/news/2017/03/21/cyanide-device-explodes-killing-familys-dog-they-cant-believe-who-planted-it-behind-their-home/.
[70] http://www.wyofile.com/column/cyanide-bomb-kills-two-casper-dogs/.
[71] http://www.wyofile.com/column/cyanide-bomb-kills-two-casper-dogs/.
[72] https://www.predatordefense.org/features/m44_WY_Amy_dogs.htm.

In addition, the March 2002 incident, where an applicator was injured when he reached into a bucket of assembled M-44s, likely occurred because he was not properly trained in the safe handling of the devices.[73]

Risk Criteria Triggering Initiation of a Special Review Are Present

FIFRA's implementing regulations at 40 C.F.R. Part 154 authorize the Administrator to initiate a Special Review of a registered pesticide if any one of the risk criteria outlined in 40 C.F.R. Part 154.7 are met.[74] In relevant part, such risk criteria include the following:

1. The Administrator finds the registered pesticide "[m]ay pose a risk of serious or acute injury to humans or domestic animals";[75]
2. The Administrator finds the registered pesticide "[m]ay result in residues in the environment of nontarget organisms at levels which equal or exceed concentrations acutely or chronically toxic to such organisms, or at levels which produce adverse reproductive effects in such organisms";[76]
3. The Administrator finds the registered pesticide "[m]ay pose a risk to the continued existence of any endangered or threatened species designated by the Secretary of the Interior or the Secretary of Commerce under the Endangered Species Act of 1973, as amended";[77]
4. The Administrator finds the registered pesticide "[m]ay result in the destruction or other adverse modification of any habitat designated by the Secretary of the Interior or the Secretary of Commerce under the Endangered Species Act as a critical habitat for an endangered or threatened species";[78] and/or
5. The Administrator finds the registered pesticide "[m]ay otherwise pose a risk to humans or to the environment which is of sufficient magnitude to merit a determination whether the use of the pesticide product offers offsetting social, economic, and environmental benefits that justify . . . continued registration."[79]

As demonstrated throughout this Petition — and further elaborated upon below — several of these risk criteria are met by use of M-44s.

<u>M-44s Pose Risk of Serious or Acute Injury to Humans and Domestic Animals</u>

As explained above and demonstrated by several recent incidents involving injury to people and their companion animals, M-44s pose a risk of serious injury – and even death – to humans and domestic animals, including family dogs. For this reason alone, a Special Review should be initiated.

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[73] M-44 Use Restrictions at 1.
[74] *See* 40 C.F.R. § 154.1 ("The purpose of the Special Review process is to help the Agency determine whether to initiate procedures to cancel, deny, or reclassify registration of a pesticide product because uses of that product may cause unreasonable adverse effects on the environment, in accordance with sections 3(c)(6) and 6 of [FIFRA]. The process is intended to ensure that the Agency assesses risks that may be posed by pesticides and the benefits of use of those pesticides, in an open and responsive manner.").
[75] 40 C.F.R. § 154.7(a)(1).
[76] 40 C.F.R. § 154.7(a)(3).
[77] 40 C.F.R. § 154.7(a)(4).
[78] 40 C.F.R. § 154.7(a)(5).
[79] 40 C.F.R. § 154.7(a)(6).

M-44s Pose Harmful Risks to Protected Species

As indicated above, M-44s have killed federally protected threatened and endangered species, including a grizzly bear, wolves, and a California condor, among other ESA-protected imperiled animals. These deaths also compel initiation of a Special Review.

M-44s Pose Other Risks to Humans and the Environment Meriting Further Consideration

The Administrator may initiate a Special Review at his discretion if the registered pesticide poses any other risk to humans and the environment warranting such review. In combination with the other risk criteria, the dangers posed to unsuspecting members of the public and non-targeted wildlife are of sufficient magnitude to warrant such review for M-44 sodium cyanide capsules. Specifically, those incidents involving harm to people that do not rise to the level of "serious or acute injury" are worthy of consideration in a Special Review, especially considering that these incidents occur routinely. The deaths of thousands of non-target animals from M-44s also weigh in favor of initiating a Special Review.

V. Conclusion

In sum, pursuant to FIFRA, 7 U.S.C. § 136d(b), the Administrator should cancel all registrations for M-44 cyanide capsules (sodium cyanide) because the pesticide presents an unreasonable adverse impact to the environment. Further, pursuant to FIFRA § 136d(c)(1), the Administrator should suspend all sodium cyanide registrations pending cancellation proceedings because an imminent hazard exists. The Administrator should also issue a stop order, pursuant to FIFRA §§ 136k, 136j(a)(2)(G), because registered users, including Wildlife Services, are using sodium cyanide, a restricted use pesticide, in violation of the product's labeling requirements, and thereby, in violation of the law. Finally, the Administrator should initiate a Special Review proceeding for all sodium cyanide registrations because multiple risk criteria of 40 C.F.R § 154 are met.

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Death Certificate for Dennis Slaugh, M-44 Poisoning Victim

STATE OF UTAH
CERTIFICATION OF VITAL RECORD

CERTIFICATE OF DEATH

State File Number: 2018002960

Dennis Ray Slaugh

DECEDENT INFORMATION

Date of Death:	February 24, 2018	Time of Death:	11:05
City of Death:	Murray	County of Death:	Salt Lake
Age:	75	Date of Birth:	September 1, 1942
Place of Birth:	Vernal, Utah	Sex:	Male
Armed Services:	No	Marital Status:	Married
Spouse's Name:	Dorothy Lorraine Hullinger	Usual Occupation:	Heavy Equipment Operator
Industry/Business:	Uintah County	Education:	Associate Degree
Residence:	Vernal, Utah	Parent or Father:	Mervin Jay Slaugh
Parent or Mother:	Mary Emily Bowden	Facility Type:	Hospital Inpatient
Facility or Address:	Intermountain Medical Center		

INFORMANT INFORMATION

Name: Dorothy Lorraine Slaugh — Relationship: Wife
Mailing Address: 4483 North Dryfork Canyon Rd, Vernal, Utah 84078

DISPOSITION INFORMATION

Method of Disposition: Cremation
Place of Disposition: Basin Cremation Center Inc, Vernal, Utah
Date of Disposition: March 2, 2018

FUNERAL HOME INFORMATION

Funeral Home: Ashley Valley Funeral Home
Address: 410 North 800 West, , Vernal, Utah 84078
Funeral Director: Jacob Phillips

MEDICAL CERTIFICATION

Medical Professional: Eric Anding MD, 5121 South Cottonwood Street, Murray (Salt Lake), Utah 84107

CAUSE OF DEATH

Acute Myocardial Infarction
Due to (or as a consequence of): Coronary Artery Disease
Other significant conditions: Cyanide Poisoning / Exposure From M44 Device 2003
Tobacco Use: Non-user
Medical Examiner Contacted: Yes   Autopsy Performed: No   Manner of Death: Natural

Date Registered: February 27, 2018
Date Issued: May 11, 2018

AMENDMENT HISTORY

02/27/2018 Spouse Last Name from **Slaugh** to **Hullinger**
03/06/2018 Decedent Date of Birth from **01/01/1942** to **09/01/1942**
05/10/2018 Conditions Contributing to Death from **Cyanide Poisoning / Exposure from M44 Device 2002** to **Cyanide Poisoning / Exposure From M44 Device 2003**

This is an exact reproduction of the facts registered in the Utah State Office of Vital Records and Statistics.
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Richard J. Oborn, MPA
State Registrar
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Jordan D. Mathis
Director/ Health Officer
TriCounty Health
Department

ANY ALTERATION OR ERASURE VOIDS THIS CERTIFICATE

Featured Incidents of Pet Killings and Human Poisonings Caused by M-44s

Published by Predator Defense, www.predatordefense.org, Sept. 13, 2018

The list below highlights just a few of the documented incidents of people and domestic animals injured or killed by the M-44 cyanide devices used by USDA Wildlife Services. It was compiled from agency documents, news reports, and various other sources. The real M-44 body count is in the thousands, and far exceeds the numbers officially reported by Wildlife Services. See explanatory note on official counts in separate report titled *"USDA Wildlife Services Yearly Summary Statistics of Domestic Dog Killings by M-44s."*

February 2018: Dennis Slaugh of Vernal, Utah, dies. Slaugh was poisoned by an M-44 in 2003, and his death certificate listed cyanide poisoning from M-44 as a contributing cause (see death certificate).

March 2017: A dog and a 14-year-old boy triggered an M-44 in Idaho. The boy, along with several emergency personnel, were exposed to cyanide. The boy suffered long-term, adverse health effects. His dog died in front of him. Were it not for wind direction, the boy might also have have died. No warning signs were posted.[i]

March 2017: Two dogs were killed in Wyoming by an M-44 during a walk with their family.[ii]

February 2011: An M-44 was placed 918 from a residence without the family's knowledge, killing their dog and violating three EPA use restrictions.[iii,iv]

February 2010: A dog was killed in Nebraska by an M-44 set by Wildlife Services on the dog owner's rangeland/pasture.[v]

April 2010: A dog wearing collar and tags was killed in W. Virginia by an M-44 set on neighboring land. The Wildlife Services agent buried her without notifying the family.[vi]

January 2008: A dog was killed by an M-44 in N. Dakota.[vii]

January 2008: A man in Texas was injured by an M-44 placed without his knowledge on grazing land.

February 2008: A beagle was killed by an M-44 in Virginia.[viii]

February 2008: A dog was killed by an M-44 in New Mexico.[ix]

April 2008: A dog in N. Dakota was killed by an M-44 set on rangeland/pasture.[x]

June 2008: A pit bull was killed in Virginia by an M-44 in a livestock pasture/hayfield.[xi]

January 2007: A dog was killed by an M-44 in North Dakota.[xii]

March 2007: A Border collie was killed by an M-44 in Virginia.[xiii]

April 2007: A Border collie puppy was killed by an M-44 in Virginia.[xiv,xv]

May 2007: A worker in Texas accidentally triggered an M-44. The cyanide was ejected into the man's eyes and he subsequently experienced burning and irritated eyes as well as disorientation.[xvi,xvii]

June 2007: A Great Pyrenees was killed by an M-44 in New Mexico.[xviii]

January 2006: A Golden retriever was killed by an M-44 in Virginia.[xix,xx]

February 2006: A Labrador retriever was killed in Utah when she triggered an M-44 set a foot from a road.[xxi]

April 2006: A young German shepherd was killed when he triggered an M-44 on public land in Utah.[xxii,xxiii]

March 2005: An Australian Shepherd was killed in New Mexico by an M-44 set by Wildlife Services on rangeland.[xxiv]

March 2005: A dog was killed in New Mexico by an M-44 set by Wildlife Services on ranch land.[xxv]

April 2005: A Border collie in New Mexico was killed by an M-44 set on the owner's ranch property.[xxvi,xxvii]

December 2005: A certified therapy dog who worked with at-risk youth was killed in front of a girl's group by an M-44 set 10 feet from a public road.[xxviii,xxix]

January 2004: A dog was killed by an M-44 set by Wildlife Services in New Mexico on the ranch of the dog owner's relative.[xxx]

February 2004: An Irish setter was likely killed by an M-44 in Virginia.[xxxi]

March 2004: A dog in Idaho was found dead within 200 yards of an M-44 set by Wildlife Services in a nearby sheep pasture.[xxxii,xxxiii]

March 2004: A German shepherd was killed by an M-44 in New Mexico.[xxxiv]

May 2003: Dennis Slaugh was poisoned and permanently disabled when he triggered an M-44 on public land in Utah. He was forced to retire from his job.[xxxv]

January 2002: A rancher in Nebraska was injured by the accidental discharge of an M-44 that had been set by Wildlife Services on his property.[xxxvi]

February 2002: A dog was killed by an M-44 set by Wildlife Services.[xxxvii]

February 2002: A Labrador retriever was killed in Virginia by an M-44 set by Wildlife Services on a neighbor's cattle pasture.[xxxviii]

February 2002: A dog was killed in New Mexico by an M-44 set by Wildlife Services on rangeland/pasture.[xxxix]

February 2002: A dog triggered an M-44 in Oregon placed on a neighboring ranch by Wildlife Services.[xl]

February 2002: A dog was killed by an M-44 set by Wildlife Services.[xli]

February 2002: A dog was killed by an M-44 set by Wildlife Services on the farm of the dog owner's relative.[xlii]

February 2002: A dog in Oregon took 8 hours to die after exposure to an M-44 set on property next door to her home and without her knowledge. During a subsequent investigation WS requested that Oregon authorities "consider the info provided during the investigation be confidential *and not disclosed as public record* [emphasis added]." WS also refused to release a copy of the incident report to the dog's owner.[xliii,xliv,xlv]

April 2002: A dog was killed by an M-44 set by Wildlife Services on a neighboring farm in Virginia.[xlvi]

June 2002: A black Angus cow was killed in West Virginia by an M-44 set by Wildlife Services in a pasture.[xlvii]

November 2002: A woman was injured after trying to remove an M-44 set by Wildlife Services on her neighbor's property.[xlviii]

May 2001: A dog in Colorado was killed by an M-44 set by Wildlife Services on a neighboring ranch "outside the provisions authorized by state law".[xlix]

April 2001: A dog in Nebraska was killed by an M-44 set by Wildlife Services on rangeland/pasture.[l]

January 2000: A dog in Oregon was killed after triggering an M-44 set 100 yards from the owner's home. The device was one of six that had been planted in a tree farm frequented by local children.[li,lii,liii]

February 2000: A dog in New Mexico activated an M-44 set on rangeland/pasture by Wildlife Services.[liv]

March 2000: A dog in Colorado was killed by an M-44 set on private property without the knowledge of the owners. The family, including a three-year-old girl, watched as the dog suffered and died. A state investigation found that Wildlife Services had not only trespassed, but broken a suite of federal rules regulating M-44s.[lv]

May 2000: A Border collie in West Virginia was killed by an M-44 set by Wildlife Services in a sheep pasture.[lvi]

September 2000: A county surveyor in Utah discharged an M-44 after mistaking it for a survey marker.[lvii]

March 1999: A man and his three-year old daughter were walking with their dog on their property in Colorado when it triggered an M-44 and later died. A WS staffer had placed two traps on their land, trespassing and breaking a suite of federal rules.[lviii]

April 1999: A dog was killed in Virginia when he triggered an M-44 set by Wildlife Services on a neighboring farm. The owner also found another dog's body at the device. A third dog also encountered an M-44 and returned home with red and swollen eyes as well as a swollen mouth and a peculiar odor. The owner himself likely experienced secondary poisoning.[lix,lx]

August 1999: An individual helping a Wildlife Services employee look for and remove M-44s accidentally fired one of the devices.[lxi]

September 1999: A hunting dog was killed in Virginia by anM-44 set by Wildlife Services. M-44s were not permitted for use in that state from September 1 to January 7, but the Wildlife Services employee had failed to remove them.[lxii]

September 1999: A dog was killed in Oregon by an M-44 set by Wildlife Services.[lxiii]

October 1999: A Wildlife Services employee in Texas accidentally discharged an M- 44 as he was setting it. He had to be airlifted to a facility for treatment.[lxiv]

October 1999: A dog was killed in Utah by an M-44 set by Wildlife Services.[lxv]

December 1999: Two dogs were killed by M-44s during a hunting trip in New Mexico on state lands.[lxvi]

December 1999: A citizen in Nebraska accidentally discharged an M-44 as he attempted to move it with a pair of pliers while he was repairing fence wire.[lxvii]

February 1998: A dog in Utah was killed by anM-44set by Wildlife Services on BLM land that adjoined the owner's private yard. No one was notified about Wildlife Services' activities.[lxviii,lxix,lxx]

November 1998: A man in Texas, working on private land, was injured when he grabbed what he thought was a rusted metal rod to pull it from the ground and an M- 44 exploded in his hand.[lxxi]

December 1998: A dog was killed in Oregon by an M-44 set by Wildlife Services.[lxxii]

April 1995: A hunter in Idaho accidentally discharged an M-44 that had been set by Wildlife Services.[lxxiii]

Fall 1994: A dog in Oregon was walking with its family when it triggered an M-44 set on the property without their knowledge. The owner, not knowing why her dog was in respiratory distress, attempted to help it and suffered secondary cyanide poisoning from inhalation. The dog suffered for 15 minutes before dying.[lxxiv]

August 1993: Two bow hunters in Utah pulled M-44s set by Wildlife Services.[lxxv]

April 1990: A dog in New Mexico accompanying a ranch hand triggered an M-44. After attempting mouth-to-mouth resuscitation on the dog, who died within a few minutes, the man quickly experienced loss of breath, a swollen tongue, a fast heart rate, numb lips, and curling fingers on one hand. He was transported to a hospital where he was treated and placed in intensive care.[lxxvi]

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[i] Idaho State Journal David Ashbury March 16 2017 Pocatello boy watches family dog die after cyanide bomb explodes. http://idahostatejournal.com/news/local/pocatello-boy-watches-family-dog-die-after-cyanide-bomb-explodes/article_d0003a2f-6b7f- 5d31-b427-68db03d3b93a.html

[ii] http://www.predatordefense.org/features/m44_WY_Amy_dogs.htm

[iii] Predator Defense, http://www.predatordefense.org/m44s_bella.htm

[iv] Tom Knudson, "Efforts to investigate Wildlife Services' methods continue," The Sacramento Bee, June 25, 2012.

[v] USDA-APHIS-WS, Adverse Effects Incident Information Report.

[vi] Letter from James R. Gardner to Commissioner Gus Douglas, West Virginia State Department of Agriculture, April 21, 2010.

[vii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[viii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[ix] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[x] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[xi] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[xii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[xiii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[xiv] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[xv] USDA-APHIS-WS, Report of Injury or Death of Non-target Animal.

[xvi] USDA-APHIS-WS, Adverse Effects Incident Information Report and Human Incident Supplemental Report

[xvii] Brazoria County Sheriff Incident/Offense Report, 22 May 2007.

[xviii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[xix] USDA-APHIS-WS, Adverse Effects Incident Information Report.

[xx] USDA-APHIS-WS, Report of Injury or Death of Nontarget Animal.

[xxi] Mike Stark, "Dog died at cyanide trap set in an off-limits area," Associated Press, 01 June, 2008.

[xxii]Memo from Michael J. Bodenchuk, Utah State Director, Wildlife Services to Ms. Barbara Knotz, 21 June 2006.
[xxiii]"Utah couple challenges USDA use of cyanide bombs," Associated Press, 20 August 2006.
[xxiv]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxv]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxvi]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxvii]USDA Work Task form, 15 April 2005.
[xxviii]USDA-APHIS-WS, Adverse Effects Incident Information Report.
[xxix]Born Free USA, http://www.bornfreeusa.org/database/trapping_incident.php?id=110
[xxx] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxxi]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxxii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxxiii]USDA-APHIS-WS, Report of Injury or Death of Non-target Animal.
[xxxiv]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxxv]Christopher Ketcham, "America's secret war on wildlife," Men's Journal, January 2008, p. 49.
[xxxvi]USDA-APHIS-WS, Adverse Effects Incident Information Report and Human Incident Supplemental Report.
[xxxvii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxxviii] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xxxix] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xl] USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xli]USDA-APHIS-WS, Report of Injury or Death of Non-target Animal.
[xlii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Human Incident Supplemental Report.
[xliii]Letter from Danielle Clair to Congressman Peter DeFazio, 18 February 2002.
[xliv]Letter from Mark Jensen, Oregon Assistant State Director, Wildlife Services, to Dale Mitchell, Assistant Administrator, Oregon Department of Agriculture, 15 April 2002.
[xlv]Letter from Congressman Peter DeFazio to Bill Clay, Deputy Administrator of Wildlife Services, 24 May 2002.
[xlvi]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xlvii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[xlviii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Human Incident Supplemental Report.
[xlix]Memo from Craig Coolahan, Colorado State Director, USDA-APHIS-WS to Martin Mendoza, Director, OSS, 16 May 2001.
[l]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[li]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[lii]Keri Watson and Greg Hanscom, "Poison traps kill unintended victims," High Country News, March 13, 2000.
[liii]Predator Defense, http://www.predatordefense.org/docs/m44_article_Buddy_Tippetts.pdf
[liv]USDA-APHIS-WS, Adverse Effects Incident Information Report.
[lv]Keri Watson and Greg Hanscom, "Poison traps kill unintended victims," High Country News, March 13, 2000.
[lvi]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.

[lvii]Memo from Michael J. Bodenchuk to Michael V. Worthen and Thomas R. Hoffman, 28 September 2000.
[lviii]High Country News 3/1300 Poison traps kill unintended victims https://www.hcn.org/issues/174/5628
[lix]Written account from Gary Tucker, 20 May 1999.
[lx]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[lxi]Accident Report signed by Alan May, District Supervisor, 16 August 1999.
[lxii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[lxiii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[lxiv]USDA-APHIS-WS, Adverse Effects Incident Information Report and Human Incident Supplemental Report.
[lxv]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[lxvi]Keri Watson and Greg Hanscom, "Poison traps kill unintended victims," High Country News, March 13, 2000.
[lxvii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Human Incident Supplemental Report.
[lxviii]Memo from Nick Sandberg, Bureau of Land Management to Mike Bodenchuck, USDA-APHIS-WS, 19 March 1998.
[lxix]U.S. Department of the Interior, Bureau of Land Management Incident Record, Case No. UT-069-98-03.
[lxx]U.S. Department of the Interior, Bureau of Land Management, San Juan Resource Area, Conversation Confirmation Report, 03 March 1998.
[lxxi]Predator Defense, http://www.predatordefense.org/docs/m44_letter_Guerro_DeFazio.pdf
[lxxii]USDA-APHIS-WS, Adverse Effects Incident Information Report and Domestic Animal, Fauna, or Flora Incident Supplemental Report.
[lxxiii]Memo from Dr. Peter L. Joseph, USDA-APHIS-Biotechnology, Biologies, and Environmental Protection to Mr. Robert A. Forrest, U.S. Environmental Protection Agency, 25 April 1995.
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[lxxv]Memo from James A. Winnat, Utah State Director, USDA-APHIS-WS to ADC employees, 09 September 1993.
[lxxvi]Memo from Larry J. Killgo, State Director, ADC, Albuquerque, NM to District Supervisor, ADC, Roswell, NM, 01 May 1990.

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Carnivore conservation: shifting the paradigm from control to coexistence

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For 90 years, the American Society of Mammalogists (ASM) has made science-based challenges to widespread lethal control of native mammals, particularly by the United States federal government targeting carnivores in the western states. A consensus is emerging among ecologists that extirpated, depleted, and destabilized populations of large predators are negatively affecting the biodiversity and resilience of ecosystems. This Special Feature developed from a thematic session on predator control at ASM's 2013 annual meeting, and in it we present data and arguments from the perspectives of ecology, wildlife biology and management, social science, ethics, and law and policy showing that nonlethal methods of preventing depredation of livestock by large carnivores may be more effective, more defensible on ecological, legal, and wildlife-policy grounds, and more tolerated by society than lethal methods, and that total mortality rates for a large carnivore may be driven higher than previously assumed by human causes that are often underestimated.

Key words: carnivores, depredation, nonlethal control methods, predator control

"…this is why the caribou and the wolf are one; for the caribou feeds the wolf, but it is the wolf who keeps the caribou strong."

Eskimo legend as told to Farley Mowat (Mowat 1973:85)

This Special Feature developed from a special thematic session on mammalian predator control at the 94th annual meeting of the American Society of Mammalogists (ASM) held in June 2013 in Philadelphia. Sponsored by the ASM Conservation Committee, the thematic session explored a range of perspectives—from wildlife managers, carnivore biologists, and sociologists—on issues of managing human conflicts involving native large carnivores. For 90 years, ASM has presented science-based critiques of lethal control of native wildlife—particularly large carnivores—by the United States federal government, starting with its 1st published Society resolution (Jackson 1924) and continuing to the present (ASM 2012; others reviewed in Bergstrom et al. 2014). Additionally, prominent early ASM members, including Aldo Leopold, C. Hart Merriam, and E. Raymond Hall, individually published letters stating that lethal control of large carnivores, particularly in the western United States, was driven by politics rather than science and was excessive in its direct effects on targeted as well as nontargeted species of native mammals (Bergstrom et al. 2014). These concerns by early 20th century mammalogists were well founded, given that, first, grizzly bears (*Ursus arctos horribilis*), and then, by the 1930s, gray wolves (*Canis lupus*) were extirpated from the western contiguous states by private and government agents (Robinson 2005).

The 1973 Endangered Species Act (16 U.S.C. 1531–1544, 87 Stat. 884, as amended—Public Law 93–205) alleviated concerns of American mammalogists that their government would allow or directly cause extinction or wide-scale extirpations of native mammals. However, in the United States as well as globally, most large carnivores have experienced substantial range contractions and population reductions; in fact, the American black bear (*Ursus americanus*) is the world's only large terrestrial carnivore species that has a global population of more than 200,000 and is one of the very few whose population trend is not "decreasing" (Ripple et al. 2014). Even in areas still occupied by large carnivores, predator removal locally in less-developed landscapes causes concern about nontarget mortality of certain rare species and indirect effects on biodiversity and ecosystem function from disruption of "top-down forcing" (sensu Estes et al. 2011; Bergstrom et al. 2014). In the United States, legal public harvest takes 2.5 million native carnivores annually (Association of Fish and Wildlife Agencies 2014). Additional human-caused mortality of carnivores due to poaching and road-kill is hard to quantify but may be higher than commonly assumed. Vehicles on roads, for example, have killed 13% of the gray wolf (*C. lupus*) population annually in Wisconsin (Treves et al., this issue). Lethal control of large carnivores in the United States by professional federal, state, and private agents constitutes a fraction of the total human-caused mortalities nationwide, but they are done primarily to benefit livestock producers in western states, often intensely at a very local scale (e.g., 884 coyotes [*Canis latrans*] killed on a single ranch in Nevada in a 2-year period by aerial gunning—Knudson 2015), and they can result in removal of 1 or more carnivore species from local ecosystems (Bergstrom et al. 2014).

Wildlife Services, a division of the United States Department of Agriculture's Animal Plant Health Inspection Services, is tasked by law "to provide Federal leadership and expertise to resolve wildlife conflicts to allow people and wildlife to coexist" (Wildlife Services 2015). Wildlife Services' research scientists do important studies on nonlethal methods of reducing carnivore–livestock conflict (e.g., Stone et al., this issue), but its field operations in the western United States have been criticized for their over-reliance on lethal means of resolving wildlife conflicts with livestock (Government Accountability Office [GAO] 1995; Niemeyer 2010; ASM 2012; Bergstrom et al. 2014). In Fiscal Year 2013, Wildlife Services killed > 75,000 coyotes (not counting 366 dens destroyed), 320 gray wolves, 345 cougars (*Puma concolor*), 3,546 red and gray foxes (*Vulpes vulpes* and *Urocyon cinereoargenteus*, respectively), and 372 badgers (*Taxidea taxus*—Wildlife Services 2015). The annual number of control kills of coyotes has remained remarkably constant since 1939, varying between 50,000 and 110,000 and has exceeded 70,000 annually since 1985 (Berger 2006; Bergstrom et al. 2014). Also typical, Wildlife Services in Fiscal Year 2013 *unintentionally* killed 397 river otters (*Lontra canadensis*), 14 kit foxes (*Vulpes macrotis*), and 41 swift foxes (*V. velox*—Wildlife Services 2015). Wildlife Services does not monitor populations of species it targets for control nor those unintentionally killed, but one of the few published estimates of an overall mortality rate is that Wildlife Services, along with state managers, removed 23.2% of the estimated coyote population of Wyoming in 1994–1995 (Taylor et al. 2009). This level of human-caused mortality of mammalian predators may have negative unintended consequences for native ecosystems and biodiversity. Lethal control of carnivores may also be unnecessary and counterproductive to its ostensible goals (see Treves et al. 2016 for a recent review). We will explore these consequences in this Special Feature. We invited individual research scientists from the National Wildlife Research Center (the research arm of Wildlife Services) to contribute a science-based defense of lethal control of native carnivores to this Special Feature, but they each, as well as the center, collectively via their director, declined the offer (L. Clark, in litt., 13 November 2013).

There are 5 categories of reasons why mammalogists and conservation biologists should be interested in guiding governments—and society at large—toward replacing localized predator removal or population reduction (lethal control) with nonlethal means of wildlife conflict resolution: 1) potential disruption of top-down forcing and consequent loss of ecosystem resilience and biodiversity; 2) "bycatch" or unnecessary killing of nontarget species of mammals and other wildlife that occurs with nonselective methods of lethal control; 3) population reduction of certain species of native wildlife valued by many parts of society for the benefit of a few favored interest groups; 4) ineffectiveness of lethal control of predators at either reducing livestock depredation or, secondarily, enhancing game populations, over the long term; and 5) ethical considerations about both the intrinsic value of carnivores and humane methods of killing them. Some of these deserve brief attention in this overview, and others will be dealt with in more detail in the 5 other papers in this Special Feature, including new empirical evidence for the efficacy of nonlethal methods as alternatives to lethal predator control.

The Important Role of Both Apex Predators and Mesopredators in Maintaining Ecosystem Function

With this topic currently under considerable empirical and theoretical scrutiny, the evidence assembled as of 2011 led 23 prominent ecologists to conclude that loss of apex predators was a major driver of destabilization and collapse of their native ecosystems, leading to pandemics, irruptions of invasive species, and lost ecosystem services (Estes et al. 2011). Aldo Leopold was one of the 1st biologists to argue that mammalian predators played an indispensable role in controlling ungulate prey, thus preventing depletion of their resources, citing the irruption of the early 20th century herd of Kaibab deer (*Odocoileus hemionus*) after widespread predator removal (Leopold 1943). A recent review of several lines of evidence concluded that Leopold was right (Binkley et al. 2006). The poor condition of rangelands in much of the western United States can be attributed partly to native ungulates whose predators have been depleted (Beschta et al. 2013). Hebblewhite et al. (2005) documented that top-down forcing exerted by wolves on browsing prey had indirect positive effects on songbird communities in the Canadian Rockies. Restoration of a putative wolf-driven trophic cascade has restored certain riparian plant and animal communities in Yellowstone National Park (e.g., Ripple and Beschta 2012; though see Mech 2012). Top-down forcing (also known as a trophic cascade, i.e., the many indirect effects predation has on lower trophic levels and the ecosystem as a whole) by wolves may be enhanced by facilitative interactions with sympatric large carnivores (e.g., cougar—Atwood et al. 2007), or it may be dampened in more human-dominated landscapes (Muhly et al. 2013). A possible indirect effect of wolf predation is to reduce abundance of songbirds and rodents in a 4-species interaction chain, by releasing the lowest of the 3 trophic levels of carnivores (Levi and Wilmers 2012). In some systems, an apex large carnivore causing mesocarnivore suppression and, indirectly, small-carnivore release may be the more natural state. Removal of the apex carnivore, conversely, causes mesocarnivore release and small-carnivore suppression, which allows an irruption of rodent populations. Such an altered trophic cascade is exemplified by the recent colonization of eastern North America by coyotes following extirpation of wolves and may explain the rapid increase in the incidence of Lyme disease (Levi et al. 2012). Lethal control of the Australian apex predator the dingo (*Canis dingo*) has caused similar state shifts, resulting in dominance of introduced mesopredators and herbivores, which then cause damage to native plant and animal communities (Wallach et al. 2010).

Ineffectiveness and Unintended Consequences of Predator Removal

The consistent annual efforts by Wildlife Services at lethal control of coyotes in the western United States, described above, did not succeed in ameliorating the long decline of the nation's sheep industry, which began in the post-war years (Berger 2006). And, local-scale removal of coyotes has been found to cause population irruptions and reduced diversity in rodent communities (Henke and Bryant 1999). Use of public harvest of cougars in Washington state to remediate livestock depredation was found to be ineffective (Peebles et al. 2013). Similarly, recreational hunting of Eurasian lynx (*Lynx lynx*) was found to have little effect on sheep depredation unless of a magnitude to cause lynx population decline (Herfindal et al. 2005). Lethal control of gray wolves in the western United States could have such unintended consequences as shifting depredation from cattle to sheep (by mesopredator release of coyotes) and increasing mortality of pronghorn (*Antilocapra americana*) fawns (Berger et al. 2008; Bergstrom et al. 2014). Lethal control of gray wolves in the northern Rocky Mountains, causing total mortality of up to 25% of the estimated population, was found actually to increase depredation on livestock (Wielgus and Peebles 2014; but see Bradley et al. 2015). There are 3 reasons that predator removal is likely to have no long-term effect—or even adverse effects—on depredation of livestock: vacant territories are quickly recolonized (Knowlton et al. 1999; Treves and Naughton-Treves 2005); immigration rate of breeding pairs into the area experiencing lethal control can increase (Sacks et al. 1999); and immigrants are more likely to be subadults, which have a greater propensity for livestock depredation than older adults (Peebles et al. 2013). Simulation results suggest that even moderate nonselective predator control can potentially increase densities of the targeted carnivore species, because nontarget deaths of co-occurring carnivore species decrease competition for the targeted species (Casanovas et al. 2012). Use of nonselective, lethal predator-control methods (e.g., trapping and poison baits) by Wildlife Services has resulted unintentionally in the deaths of individuals of 150 species of vertebrates since 2000 (Knudson 2012) and at least 12 taxa of mammals protected (or candidates for protection) under the Endangered Species Act since 1990 (Bergstrom et al. 2014). Selective local removal of carnivores such as coyotes may eliminate the bycatch problem, but it can still trigger mesopredator release with unintended negative consequences (Mezquida et al. 2006).

The ASM has supported lethal control of large carnivores in certain cases where preservation of critically endangered wildlife species demands it (such as cougar predation on isolated populations of peninsula bighorn sheep, *Ovis canadensis nelsoni*—ASM 2012; Stephenson et al. 2012), but culling apex predators to enhance common game species may be unnecessary at best and harmful at worst. To the latter point, it is well known that wolves preferentially prey on older and diseased individuals (Mech and Peterson 2003; Wright et al. 2006), so natural predation is an important selective agent for the prey. To the former point, recent studies have concluded that gray wolf populations are intrinsically density dependent. That is, rather than being prey-limited, wolf densities are regulated through social interactions, with increasing interpack aggression and mortality at higher densities (Cariappa et al. 2011; Cubaynes et al. 2014). Large mammalian carnivores have been found to limit prey populations, broadly and in specific predator–prey interactions (Binkley et al. 2006; Ripple and Van Valkenburg 2010; Christianson and Creel 2014), but the effect of reduction or removal of predators on densities and dynamics of prey populations in any specific case can be hard to predict. Experiments removing coyotes and cougars in Idaho showed winter weather to be much more important than predation in predicting population trends of mule deer (*O. hemionus*—Hurley et al. 2011). A 7-year effort to remove all mammalian nest predators of ground-nesting birds (coyotes being the largest) from study sites in the southeastern United States concluded that removal of mammalian predators had no net effect on nest predation, primarily because of compensatory increases in predation by snakes (Ellis-Felege et al. 2012). A meta-analysis of 113 predator removal experiments (which was a taxonomically broad sample of animal predators) found that the intended beneficiary prey populations declined in 54 of them (Sih et al. 1985). This illustrates the multiple indirect pathways of potential top-down forcing that may be altered by removal of an apex predator from a complex food web, producing many possible outcomes for prey dynamics. For a mammalian carnivore example, 1 such pathway is through "apparent competition" with an alternate ungulate prey species, mediated through a different predator that increases compensatorily (Serrouya et al. 2015). Another pathway involves release of a mesopredator that preys preferentially on neonates of the same ungulate prey species (Prugh and Arthur 2015).

Effectiveness of Nonlethal Control of Depredation

Use of nonlethal methods (such as guardian animals and livestock protection collars) to prevent livestock depredation by leopards (*Panthera pardus*), caracals (*Caracal caracal*), and jackals (*Canis mesomelas*) in South Africa was found to be less expensive and more effective than lethal predator control (McManus et al. 2014). In this Special Feature, Stone et al. (this issue) document that, over a 7-year pilot project in prime wolf habitat in Idaho, the adaptive use of a suite of nonlethal deterrent strategies reduced sheep depredation by more than

3-fold compared to sheep allotments in Idaho that used lethal controls over the same time period. Presenting results from a large cattle station in Australia, where full implementation of such nonlethal strategies may be prohibitive, Wallach et al. (this issue) argue that simply ending lethal control of dingoes reduced depredation by allowing the social structure of the predator to stabilize, and additionally that cattle mortality can be reduced most effectively by improving husbandry practices. These 2 studies do not meet the "gold standard" of replicated, randomized experimental design (which few predator-control studies do—Treves et al. 2016), because the latter would have been impossible without intentional further killing of important apex predators of great conservation value (in the case of Idaho gray wolves still legally protected for most of the study). Nonetheless, their results are valuable in providing insights into workable alternatives to lethal control for solving wildlife–livestock conflicts. Both of these studies suggest that stable, naturally regulated populations of social carnivores not significantly exploited by humans are the preferred option for both reducing livestock depredation and restoring the functional role of apex predators to ecosystems. These findings for large canids mirror those for cougars, in which excessive harvest replaces adult males with immigrating adolescent males, which are more prone to depredate (Peebles et al. 2013).

Members of ASM Are Acutely Aware of Guidelines on Humane Treatment

There has been much discussion in recent years within the Society about the ethical constraints and obligations pertaining to working with live mammals. While we have striven to ensure that Animal Care and Use regulations imposed on us by extrinsic bodies are not overly onerous and do not prevent us from vigorous pursuit of our science, we nonetheless all feel the obligation to abide by a set of rules for humane treatment of our mammalian study subjects. Not a paper is published in this journal presenting original results from live animal subjects that does not state that the study adhered to these ASM-adopted guidelines (Sikes et al. 2016). Ironically, ASM's guidelines were developed in large part in response to oversight by United States Department of Agriculture-monitored institutional Animal Care and Use committees at universities where many of us work, yet the agencies in the United States Department of Agriculture, including Wildlife Services, are not obligated to abide by the guidelines that their agency helped produce. Although they follow guidelines of the American Veterinary Medical Association on euthanasia, Wildlife Services claims their "management and operational programs are exempt from Animal Welfare Act (1966, 7 U.S.C. 2131, 9CFR) compliance" (Clay 2012:8).

In this Special Feature, Slagle et al. (this issue) show that, while the United States public accepts that predators may need to be controlled, there is low and declining acceptance of lethal predator-control methods, which are regarded as inhumane. Governments at the federal, state, and local levels are tasked with serving broad constituencies, and in the case of native wildlife, which are a public trust asset (Bruskotter et al. 2011; Treves et al. 2015), they should be responsive to these public attitudes. In practice, some government resource agencies or the appointed government boards that rule them, or both, have traditionally favored narrower constituencies within the public. State wildlife or game agencies have elected to provide hunting opportunities for certain species, including large carnivores, even if citizens opposed to hunting a particular species of large carnivore greatly outnumber those wishing to hunt it. A case in point is the state of Michigan recently approving a wolf hunt following removal of federal protection by the Endangered Species Act, and in this Special Feature, Vucetich et al. (this issue) argue that the North American Model of wildlife management, to which the profession is supposedly bound, does not support the hunt. In a society in which lethal control of predators is viewed increasingly negatively and scientific consensus is emerging that social carnivores occupying apex-predator trophic levels function best and depredate least when not lethally exploited, killing native large carnivores is an issue that will become increasingly controversial and should receive increasing scientific scrutiny.

Finally, insofar as most states, probably for the foreseeable future, will continue to include large carnivore hunting among their wildlife management tools, it is important that decision-makers in wildlife agencies have valid data on mortality rates from all mortality sources and on the further effects of anthropogenic mortality on recruitment (which may be negative), so that harvest quotas may not push total mortality beyond a sustainable level (see Creel et al. 2015). To that end, Treves et al. (this issue) show that well over a third of mortality of wolves over the past 3 decades in Wisconsin was due to poaching and another 13% was due to vehicle collision, suggesting that total mortality of the population, which was subsequently exposed to harvest, is higher than the management agency assumes. Setting wildlife management goals at reducing carnivore mortality to at most sustainable levels, and eliminating human-caused mortality wherever possible, is in line with the best current ecological, social, and ethical scholarship, as papers in this Special Feature attest.

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SCIENTIFIC REPORTS

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Limited evidence on the effectiveness of interventions to reduce livestock predation by large carnivores

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Successful coexistence between large carnivores and humans is conditional upon effective mitigation of the impact of these species on humans, such as through livestock depredation. It is therefore essential for conservation practitioners, carnivore managing authorities, or livestock owners to know the effectiveness of interventions intended to reduce livestock predation by large carnivores. We reviewed the scientific literature (1990–2016), searching for evidence of the effectiveness of interventions. We found experimental and quasi-experimental studies were rare within the field, and only 21 studies applied a case-control study design (3.7% of reviewed publications). We used a relative risk ratio to evaluate the studied interventions: changing livestock type, keeping livestock in enclosures, guarding or livestock guarding dogs, predator removal, using shock collars on carnivores, sterilizing carnivores, and using visual or auditory deterrents to frighten carnivores. Although there was a general lack of scientific evidence of the effectiveness of any of these interventions, some interventions reduced the risk of depredation whereas other interventions did not result in reduced depredation. We urge managers and stakeholders to move towards an evidence-based large carnivore management practice and researchers to conduct studies of intervention effectiveness with a randomized case-control design combined with systematic reviewing to evaluate the evidence.

Predation on domestic animals is an important factor influencing the coexistence between large carnivores and humans[1]. In order to mitigate the negative impact of large carnivores on livestock, modern societies (through governments), non-governmental organisations (NGOs), and individuals invest logistical and budgetary efforts (i.e., public and private money) in a large number of preventive measures (hereafter interventions) that are believed to reduce the risk or impact of depredation (i.e., carnivores attacking, injuring, or killing domestic animals). Selecting the correct intervention to implement in each unique case is challenging. Choosing incorrectly can for instance result in multiple negative consequences, such as higher economic costs than expected due to potential livestock losses and the need for additional interventions, or exacerbate conflicts between different stakeholders. The choice of intervention can make the difference between life and death to domestic animals as well as carnivores[1–4]. Choosing the appropriate intervention is also important for establishing trust in carnivore managing authorities. Mistrust in authorities and/or management strategies can create feelings of frustration, anger, or fear. Feelings of this kind may ultimately enhance the negative view of carnivore conservation and management, and undermine coexistence between humans and large carnivores in multi-use landscapes[5–7]. Additionally, negative feelings could be spurred by a lack of reliable information on the expected costs, side-effects, and effectiveness of interventions. Particularly so in cases when livestock losses continue after the implementation of an intervention, or when the cost of the intervention is high compared to the expected cost of depredation.

Authorities, wildlife managers, and owners of domestic animals face a wide variety of potential interventions to protect domestic animals from large carnivores[8–10]. Interventions range from lethal (e.g., culling) to non-lethal methods (e.g., fences), overarching policy goals (e.g., carnivore population caps), interventions funded by

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authorities or initiated and undertaken by the affected people (e.g., compensation systems and increased guarding, respectively), to information dissemination (e.g., public meetings). Lethal interventions have traditionally been widely utilized and played an important role in the reduction and extirpation of large carnivore populations around the world, including Europe and North America[1]. Lethal interventions currently receive less public support than in the past[11,12] and their implementation on endangered populations contributes to controversy over their use. Some non-lethal interventions have long histories of use, such as livestock guarding dogs[13], whereas others are based on new technologies, such as scaring devices or electric fencing[14,15]. Carnivore depredation on livestock has occurred for thousands of years, at least since the expansion of livestock husbandry after domestication ca. 11,000 BP[16]. Due to the long and extensive use of various interventions, it could be expected that the best interventions to reduce the impact of large carnivore depredation are by now well tested and identified. Yet, scientific evaluations of interventions are still surprisingly scarce[17,18] and, in general, our understanding of their efficacy is based on narrative review[19].

With this study we aim to move closer to the evidence-based practice and systematic reviewing process that has increased the efficiency of medical interventions[20], and which is actively increasing the efficiency and trustworthiness of biodiversity conservation[21]. We look for evidence of the outcome of implementing interventions, to assess to what extent interventions reduce the risk and impact of attacks by large carnivore on livestock, i.e. how effective interventions are to prevent depredation. This information is critical for the owners of domestic animals, who need to know if the interventions that they spend time and money on actually prevent the loss of livestock. A true systematic review[22] is beyond the scope of this paper, but we use a structured methodology based on systematic review procedures. Our aim is to answer the fundamental question "What works?" regarding interventions designed to reduce the risk and impact of large carnivore depredation on domestic animals. The objective is to provide a quantitative assessment of the efficacy of evaluated interventions by exhaustively reviewing empirical studies, without building upon previous reviews[8,17,18], and using a transparent and replicable methodology. Based on our findings, we discuss what scientific evidence is currently available and where there might be room for improvements in large carnivore management science.

Results

Our initial literature search returned 27,781 publications, of which 562 were read in full (see Methods for search and screening criteria). Only 21 (3.7%) of the 562 reviewed publications fulfilled our inclusion criteria (see Supplementary Table S1). The limited number of studies selected (mean number of studies by intervention group: 3.2, range 1–6, Supplementary Table S1) meant we were unable to carry out meta-analysis on the effectiveness of interventions. Some publications included more than one study, such as testing different interventions in one setting[23,24], or one intervention tested on several carnivore species or livestock types[25–29]. Out of the 30 carnivore species considered, the final 21 publications focused on 10 species (see Supplementary Table S1). All publications included at least one study of an intervention that reduced the risk of carnivore attack, whereas five publications (24%) also included studies where interventions had either no effect, or led to an increase in livestock depredation.

Change livestock type. One study evaluated the effect of livestock breed on livestock depredation[26] and found that the heavier breed, *Dala sheep* (focal group), suffered more losses to wolverine (*Gulo gulo*) depredation than did the lighter *Norwegian short-tailed sheep* (relative risk $RR_{\text{wolverine depredation}} = 0.72$), *Norwegian fur-bearing sheep* ($RR_{\text{wolverine depredation}} = 0.47$), and *Rygja sheep* ($RR_{\text{wolverine depredation}} = 0.63$) breeds (Fig. 1). This suggests that the choice of livestock breeds in each particular context (considering also the carnivore species present in the area) can have an effect on losses. *Dala sheep* are heavier than the *Norwegian short-tailed*, the *Norwegian fur-bearing*, and the *Rygja* breeds, with ewe weights of 80–90 kg, 60–70 kg, 60 kg, and 75 kg, respectively[30,31].

Enclosure. Six studies focused on the effect of livestock confinement to avoid carnivore depredation. Livestock are often confined during the night, when carnivores are most active. A negative effect on livestock depredation, was found for keeping sheep in night barns in Slovenia[32] ($RR_{\text{bear depredation}} = 0.04$, $RR_{\text{wolf depredation}} = 0.25$) and also for using night corrals to protect livestock from pumas *(Puma concolor)* in southern Brazil[33] ($RR_{\text{puma depredation}} = 0.25$, Fig. 1). The most effective enclosure design appears to be context-dependent, and dependent on the carnivore guild, as different species apply different tactics to enter enclosures (e.g. climbing, digging or jumping). A pole construction kept spotted hyenas *(Crocuta crocuta)* out of sheep and goat enclosures better than a bush fence construction[28] ($RR_{\text{hyena depredation}} = 0.26$), but the enclosure of pole construction left livestock more susceptible to leopard *(Panthera pardus)* depredation than bush fence enclosures[28] ($RR_{\text{leopard depredation}} = 3.50$). In another study with multiple carnivores, including leopards, the transparency of the enclosure did not have any effect ($RR_{\text{lion, leopard, hyena depredation}} = 1.02$, $RR_{\text{hyena depredation}} = 0.99$, with increasing transparency) on the risk of carnivore depredation[23]. However, fortified or improved enclosures were effective in systems with lions[34] *(Panthera leo)* ($RR_{\text{lion depredation}} = 0.12$, Fig. 1) and appear to reduce the impact by spotted hyenas[35] (see Supplementary Table S1).

Guarding. Three studies fulfilled our requirements for evaluating the effectiveness of guarding. Estimates of RR suggest that herders reduced the severity of wolf *(Canis lupus)* attacks on sheep in Greece[36] ($RR_{\text{wolf depredation}} = 0.21$, Fig. 1). The risk of coyote (*Canis latrans*), puma, and black bear (*Ursus americanus*) predation was also lower in sheep herds where human herders were present[24] ($RR_{\text{coyote, puma, black bear depredation}} = 0.49$), and a larger number of men present in a boma had a small negative effect on the depredation risk ($RR_{\text{lion, leopard, sp. hyena depredation}} = 0.87$) by lions, leopards, and spotted hyenas[23] (Fig. 1).

Livestock guarding dogs. Five studies met our requirements for evaluating the effectiveness of livestock guarding dogs. Four studies suggest a lower risk of sheep predation where guarding dogs were present[13,23,25,32].

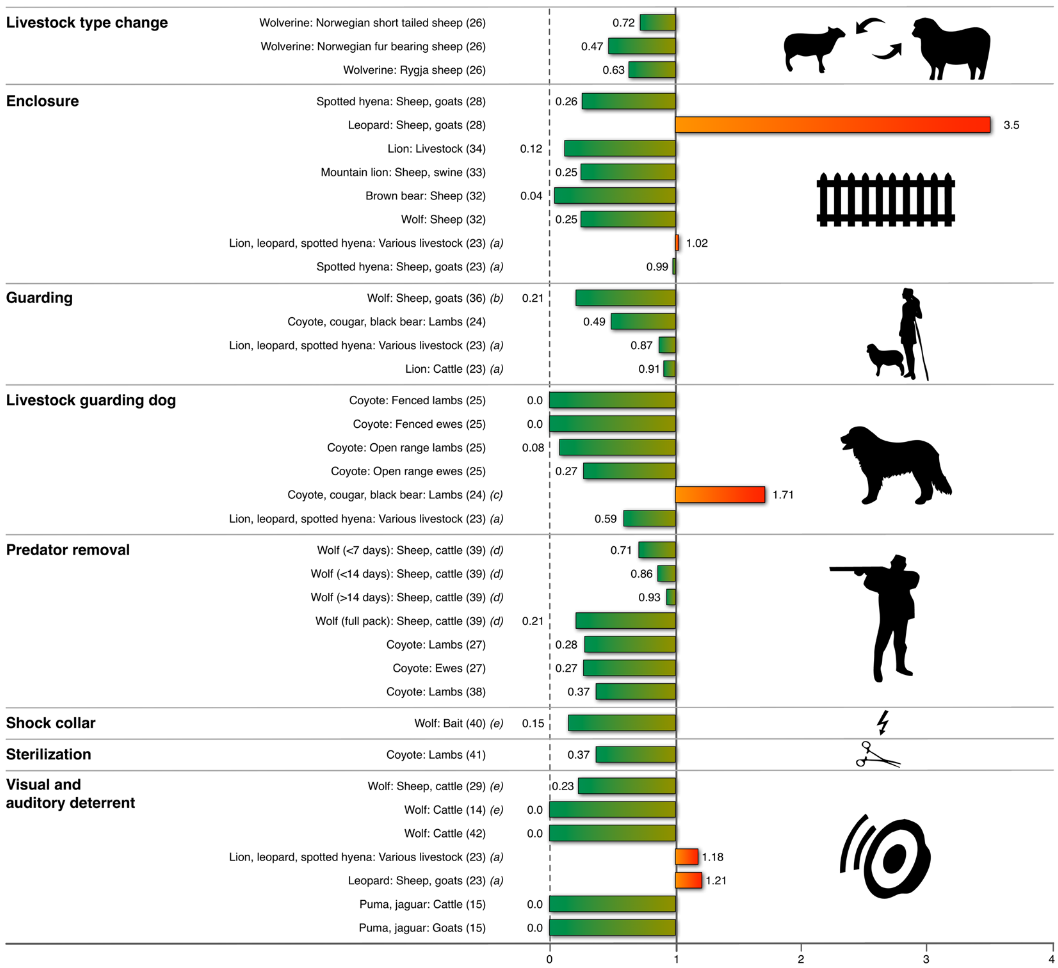

Figure 1. The intervention effectiveness described as relative risk (RR) for each study. RR = 1 suggests no difference in the risk of carnivore attack between treatment and control groups. RR > 1 suggests there is a higher risk of carnivore attack in the treatment group, and the value can be infinitely large. RR < 1 suggests that there is a lower risk of carnivore attack in the treatment group, and the minimum possible value is 0 (no attack in the treatment group). Each row in the figure represents a study or sub-study of an intervention in a certain setting, with the carnivore species and type of livestock described in the figure. Reference to the original publication is written in brackets. For more information of each study please refer to Supplementary Table S1. Additional information for particular studies: (**a**) For Woodroffe *et al.*[23], odds-ratios were converted to RR using an online Odds Ratio to Risk Ratio calculator[64]. (**b**) Iliopoulos *et al.*[36] measure severity of the wolf attack once it has occurred. (**c**) Palmer *et al.*[24] b state that the treatment herd was divided into two bands, and all losses occurred in the band without the livestock guarding dog. (**d**) For Bradley *et al.*[39] we report hazard ratio HR (1850 days) from the original study. (**e**) Hawley *et al.*[40], Davidson-Nelson & Gehring[29], and Lance *et al.*[14] measure trespass rate into baited areas instead of livestock losses.

In one study[23], the dogs were not typical livestock guarding dog breeds, but an increasing number of dogs nevertheless reduced the total risk of carnivore attacks ($RR_{\text{lion, leopard, sp. hyena depredation}} = 0.59$, Fig. 1) likely by alerting people[23]. In four other studies, dogs were guarding breeds that could alert human herders, but may also have been able to deter attacks, or attack the carnivore themselves. Guarding dogs present in sheep herds reduced the risk of depredation of sheep, particularly where herds were fenced ($RR_{\text{coyote depredation}} = 0$ for lambs, $RR_{\text{coyote depredation}} = 0$ for ewes), but also on open range pastures ($RR_{\text{coyote depredation}} = 0.08$ for lambs and $RR_{\text{coyote depredation}} = 0.27$ for ewes, Fig. 1)[25]. One study had increasing depredation losses to coyotes, cougars, and black bears on the herd level ($RR_{\text{coyote, cougar, black bear depredation}} = 1.7$) when the livestock guarding dog was present[24] (Fig. 1). However, this result was likely due to the insufficient statistical unit (herd) used to measure the effect, as only part of the unit was guarded by a dog and the flock of sheep in which the dog was kept, suffered no losses to carnivores[24].

Predator removal. Removal of carnivores can be accomplished through translocation or by various methods of lethal control. Lethal control can target all individuals in an area (i.e., proactive culling)[3], using methods such as trapping[37] or aerial shooting[38]. Alternatively, lethal control might target specific individuals within a population that pose a higher threat to livestock than the average individual - i.e. "problem individuals"[27,37], through selective trapping, poisoning, or hunting. Four studies met our criteria for evaluating the effect of predator removal, all on coyotes and wolves[27, 37–39]. Only one considered cases of translocation in their dataset[39]. The largest decrease in risk of livestock depredation ($RR_{coyote\ depredation} = 0.27$ for ewe predation and $RR_{coyote\ depredation} = 0.28$ for lamb predation[27]) was shown in studies where adult or breeding canids were selectively removed[27, 37] (Fig. 1). A third study found partial wolf pack removals had a smaller negative effect (Hazards Ratio $HR_{wolf\ depredation} = 0.71$) on the risk of recurring depredations than full pack removal ($HR_{wolf\ depredation} = 0.21$) by lethal control or successful translocation over a period of 1,850 days[39]. An important time aspect of successful removal actions to prevent recurring depredation events was identified: if partial pack removal was accomplished within 7 days there was a slight negative effect on the probability of recurring depredations ($HR_{wolf\ depredation} = 0.71$), after 7 days the effect was reduced ($HR_{wolf\ depredation} = 0.86$), and 14 days after the first depredation event no effect remained ($HR_{wolf\ depredation} = 0.99$), when compared to no wolf removal at all (Fig. 1)[39]. One study of non-selective proactive culling in predefined areas, found that the intervention could still reduce the risk of coyote depredation[38] ($RR_{coyote\ depredation} = 0.37$), although the effect appears smaller than for selective removal of individuals (Fig. 1). To achieve this effect with proactive culling before the grazing season, the total number of coyotes removed in the treatment pastures was more than twice (5.7 ± 1.1) the number removed in the control pastures where coyote removal only occurred during the grazing season (2.0 ± 1.0)[38].

Shock collar. Wolves treated with an electric shock whilst approaching a baited site altered their behaviour and reduced their visitation rate to the bait site during the treatment period, in comparison to control wolves that carried a collar without an electric shock devise ($RR_{wolf\ trespass} = 0.15$, Fig. 1)[40]. Importantly, conditioning against bait sites was not clearly observed once shocking ceased[40].

Sterilization. One study evaluated the effect of sterilization on modifying coyote predatory behaviour[41]. Sterilization of coyote packs eliminates the need for coyotes to provide food for pups. The study found that, in comparison to control coyote packs, the intervention led to a reduction in lamb losses ($RR_{coyote\ depredation} = 0.37$, Fig. 1), but it did not eliminate lamb losses entirely in territories where coyote packs were sterilized[41].

Visual & Auditory deterrents. Three studies evaluated the effect of fladry (i.e., a rope or line running around an enclosure and from which strips of fabric are suspended) on manipulating carnivore movement in a way that might be expected to reduce livestock depredation. A field trial observed that livestock depredation by wolves ceased in treatment pastures, whilst it continued in control fields ($RR_{wolf\ depredation} = 0$, Fig. 1)[42] and another trial found that wolves did not trespass into fladry pastures whereas they did trespass into control fields ($RR_{wolf\ trespass} = 0.23$, Fig. 1)[29]. On the contrary, coyotes trespassed into treatment fields whilst they did not trespass into control fields, suggesting no effect in reducing coyote trespassing[29]. An electrified fladry was tested during a 3-week trial, and wolves did not trespass in treatment fields but did trespass in control fields ($RR_{wolf\ trespass} = 0$, Fig. 1)[14]. The same study[14] detected no effect on livestock losses. Regarding fladry, the timespan during which the intervention can remain effective should be considered. For example, Musiani *et al.*[42] observed an effect for 60 days, after which wolves trespassed into a treatment pasture and killed livestock again, suggesting that the intervention may have an effect whilst it remains a novelty to the wolves. Other visual deterrents (e.g., hung clothes), combined with auditory deterrents, were found to repel depredation by large neotropical felids[15]. However, visual interventions could potentially also function as an attractant. An evaluation of the effect of the number of scarecrows on an enclosure (thorn-bush boma), to reduce livestock depredations, found that a higher number of scarecrows was associated with a higher number of attacks ($RR_{lion,\ leopard,\ hyena\ depredation} = 1.18$, $RR_{leopard\ depredation} = 1.21$, Fig. 1)[23].

Discussion

After reviewing 562 scientific publications addressing livestock depredations by large carnivores from 1990 to 2016, our study reveals a worrying result with substantial implications for large carnivore management: there are not many scientific publications with evidence of effectiveness for any intervention intended to prevent livestock depredation by large carnivores ($n = 21$). These results are in line with the result of Miller *et al.*[17] and Treves *et al.*[18], albeit using a different literature search method and analytical approach, reinforcing the notion that there is very little scientifically published material on the topic, regardless of the literature search methodology. The results presented here could, and should, make us question the presumed effectiveness of widely recognized interventions. Interestingly, some well-known and broadly recommended interventions completely lack scientifically published evaluations, such as carnivore deterring electric fences[43]. But evaluations based on scientifically-sound study designs are also needed to quantify the efficacy of virtually all other interventions.

Nevertheless, the final set of selected studies allows some discussion about the current state of knowledge of intervention effectiveness. Unsurprisingly, the effect of interventions is context dependent and appears to vary with how well the actual problem is targeted. However, identifying the problem locally is rarely easy – the problem could be carnivores of various species, all individuals of a certain species, or even certain individuals within a species[44]. Other carnivore species, or individuals, may be completely unaffected by a specific intervention[28], or potentially even be attracted to it[23].

For example, livestock enclosures generally appear to be an effective intervention for protecting livestock from carnivores, but only when the problem species or individual is successfully targeted. In certain cases, the enclosure construction may facilitate entrance by a certain carnivore which may exacerbate livestock losses, likely because livestock are unable to escape the predator. In situations where the carnivore guild is diverse, enclosure constructions may be difficult to design to target multiple species, leading to a reduced total effect of the intervention[23, 28]. Nevertheless, where the problem species/individuals are known and can be targeted with suitable enclosure construction, this intervention has great potential for protecting livestock, similarly if the intervention is applied during night-time and the targeted carnivores are mostly nocturnal[32, 33]. Enclosures can likely be improved to exclude multiple carnivores if their biology and behaviour is considered during construction.

Livestock guarding, either by humans or by livestock guarding dogs have been used to protect livestock from large carnivores for millennia[45]. These types of interventions appear to be effective measures for reducing livestock losses (Fig. 1). However, there are factors to take into account when implementing these interventions. For example, there are investments (e.g. acquisition of guarding dogs), running costs (e.g. salaries for shepherds or food for guarding dogs), and - in the case of guarding dogs - dog handling legislation associated with their use. Therefore, operators will need to calculate if costs are covered by a reduction of livestock losses for their system, or if there are other benefits that make up the difference. The effect of the livestock guarding dogs appears greater when lambs were fenced in comparison to open range husbandry[25]. Thus, it seems that this intervention could work well in areas where the likelihood of a carnivore attack is high, and where livestock are confined in a way that allows a dog to supervise the flock without straying, particularly at night.

In the past, large-scale predator removal programmes, supported by carnivore eradication policies, brought many carnivore species to regional or national extinction, at which point livestock depredation would cease. In this regard, complete carnivore removal could be considered effective at eliminating livestock depredation. However, carnivores are now more highly valued by society and most large carnivore populations currently benefit from some levels of legal protection that precludes their unregulated killing. For example, the European Habitats Directive 92/43/EEC permits derogating to the strict protection regime for bears, lynx, wolves, and wolverines only to cases where there is no satisfactory alternative to prevent serious damage, in particular to livestock (Article 16.1)[46]. Carnivore removal remains a controversial intervention, and lethal interventions are becoming less popular in society. Nevertheless, where no satisfactory alternative is found to prevent damage on livestock, some predator removal is still used as an intervention. Unselective predator removal may reduce livestock losses[38] unless the removed individuals are instantly replaced by new individuals, or represent a part of the population that does not kill livestock. In this case predator removal may be completely inefficient[37, 39]. On the contrary, where problem individuals can be identified and specifically targeted, this intervention may have greater potential[27, 37]. A recent publication showed that the removal of entire wolf packs reduced the occurrence of recurring depredations, whereas partial pack removal was only marginally more efficient than no removal[39]. Although the social status of removed individuals is not reported, a full pack removal - where all individuals are essentially removed - would include potential "problem individuals", whereas they could have been missed in a partial pack removal. The timing of predator removal can also influence the effectiveness of the intervention[39, 47], either because most recurring depredation events occur within a limited time, or due to different movement patterns in the carnivore population. If culling occurs before the dispersal season, it may be more likely that new depredating individuals reclaim an area, than if culling is made after the dispersal season[48, 49]. Additionally, the population and social structure of the target species can influence the effectiveness and suitability of lethal interventions. For instance, predator removal from source populations may be more effective in reducing livestock depredation compared to removals in sink populations (removed individuals in sink populations may be replaced by new individuals quicker). However, removal of predators from source populations may also be more detrimental to the carnivore populations[50]. Controversy exists about the potential side effects of lethal control, such as disruption in carnivore social structure that could lead to increased immigration causing further livestock depredation[47, 51–53].

Visual and auditory deterrents represent ways of non-lethal control that propose alternatives to lethal control. Fladry is a historically utilized visual deterrent that has received some attention in research. Cloth flags hung from a rope were used historically to steer wolves during hunts, and now fladry is used to repel wolves from areas with livestock. Sample sizes are limited, but studies indicate that fladry can manipulate wolf movement in a way that is expected[14, 29], or observed[42], to reduce livestock losses. Electrical fladry may have additional repelling properties, but is costlier and needs more maintenance to be fully functional[14]. The need for maintenance also means that fladry lines are frequented by humans[29, 42], likely adding additional human scent and presence to the area. An interesting prospect for future studies would be to disentangle this effect from the effect of the cloth line itself. Over time, as the fladry becomes a familiar feature in the wolves' environment[42], or due to mechanical failures, e.g., wear and tear, or entanglement, the effectiveness of the intervention decrease[54]. The application time of an intervention is therefore important to address. Indeed, some of the reviewed studies were of limited duration[14, 40, 41], and many more lack a clear description of the timeframe during which interventions were implemented. Future studies would benefit from considering closely the study length, to better evaluate the length of time that intervention implementation is expected to reduce losses.

Several interventions – *change livestock type*, *shock collar*, and *sterilization* - have only been properly evaluated in one single study. Generalizing the effects of these interventions on livestock losses is, therefore, practically impossible. Choosing a type of livestock species or breed that is less prone to predation may, in some situations, reduce livestock losses[26]. Nevertheless, unless we gain knowledge about what types of livestock are more resilient to particular predators, livestock owners are at risk of choosing a type of livestock that is actually more, or equally, prone to depredation. In this case the intervention could be useless or even counterproductive. It may thus be better to refrain from using an expensive or effort intense intervention like changing livestock type, until more evaluations are made. Meanwhile, practitioners can choose among those interventions for which there is at least some evidence available.

Policy makers and practitioners should also give thorough consideration to the feasibility of interventions, before advocating its use. For instance, shock collars can potentially train wolves to avoid livestock herds[40], but it may not be financially or logistically feasible to collar all carnivores in an area, and uncollared individuals could still kill livestock. In such situations, managers and livestock owners are likely better off considering other interventions. Likewise, sterilizing reproducing animals may reduce livestock losses[41], but this intervention may be very expensive, ethically unacceptable, and counterproductive in achieving conservation goals.

We suggest that feasible interventions, i. e. low-cost interventions that build on existing technology and can be easily implemented in multiple contexts, should be prioritized for scientific evaluation. This could be achieved by testing the effects of different versions or applications of already existing interventions such as guarding or enclosures. Scientific evaluations of intervention effectiveness are needed to improve, and increase the trustworthiness of, large carnivore management. Whilst carnivore management relies heavily on interventions to protect livestock from carnivore attack, the scientific evidence supporting intervention effectiveness is sparse, highlighting the need for further investigations to provide scientifically-sound information on what interventions work. It is questionable if the contemporary scientific evidence is even solid enough to allow generalized assumptions about the effectiveness of the presented interventions, not least as the few studies which have attempted to evaluate intervention effectiveness deal with different systems and carnivore species. As scientific evaluations are generally lacking, it is likely that the choice of interventions is often based on expert opinion, rather than evidence[2, 8, 43]. The time and money spent on interventions could likely be used more efficiently, if there was evidence available to guide management choices.

While in a long term perspective it would be beneficial to know the effect of interventions before investing in them, the short term goals for farmers, managers and researchers may not be in favour for these kinds of studies being conducted in the near future. Attempts to increase the involvement of these actors, contributing together to evidence-based approaches, may be one way to alter the odds in a favourable direction. We are not suggesting that farmers or managers should do nothing until evidence is available, but merely encourage these actors to promote collaborative approaches, and work together in order to increase the proportion of studies aiming to quantify the effect of interventions. Within systems that aim for adaptive management of large carnivores, the use of interventions should preferably be applied in a way that allows scientific evaluation of intervention effectiveness[55]. Special attention should focus to match comparable treatment and control samples. A limitation in this regard has been that interventions often are allocated to areas or situations where the risk of predation is higher whilst the controls are allocated to areas or situations where the risk is lower, thus the pair does not match.

Although in this study we have reviewed the existing scientific evidence using the broad scope of the Zoological Record database, we are unable to answer the question "What works?" with regards to intervention effectiveness to prevent carnivore attacks on livestock. This inability is not caused by flaws in our methodology, but by a lack of robust studies able to identify reliable answers, which some have suggested also applies to other large carnivore questions[55]. It is possible that some studies were overseen in this literature review as we limited our search to one database and excluded grey literature and literature published in languages other than English. We chose to exclude grey literature because the scientific contribution was an important focus of this review, and we believe that this exclusion criterion did not eliminate a large number of studies. While including only peer-reviewed papers we may be at risk of some publication bias, as it may be harder to publish studies when no effect of interventions was found, peer review provides a quality control for the included studies, reducing the risk of incorrect conclusions in our review. We did not set any geographical limitations to our literature search, but the 21 studies fulfilling our criteria were limited to the African ($n=4$), European ($n=3$), North American ($n=13$), and South American ($n=1$) continents, thereby limiting the study species to large carnivores present in these ranges (see Supplementary Table S1).

Ultimately, the field needs to move toward an evidence-based practice informed by regular gold standard systematic reviews – with published peer-reviewed protocols and replicable searches for scientific, as well as grey, literature in multiple relevant databases and websites - to turn the management of large carnivores into a cost effective and trustworthy practice[22]. We also suggest that future large carnivore management adopts the basic principles of an adaptive approach[56] and plan interventions to allow evaluations of effect and causality. We hope researchers embrace the challenge to improve study designs and move towards solid evaluations of management interventions.

We fully acknowledge the difficulties facing research projects studying large carnivores and livestock and that it is far from always possible to perform randomized controlled trials. Nevertheless, we suggest the field of large carnivore management follows the lead of medical sciences and conservation practices, and aim to produce evidence of the highest possible quality[57]. If we continue to do research in the way we have so far been doing, many more papers will be published in the coming years, but likely providing very little reliable knowledge on the effectiveness of interventions that cost farmers and tax payers vast amounts of money, as well as the lives of livestock and carnivores around the globe.

Methods

Literature review. To be included in this literature review, studies had to *i)* be written in English and published in a peer-reviewed scientific journal; *ii)* include an empirical study of wild (i.e., not captive) carnivores; *iii)* include a quantitative evaluation of interventions to prevent/reduce depredation of livestock (excluding apiaries); *iv)* include a description of the methods used to implement the intervention (treatment) and of a study design sufficient for replication; and *v)* include a matched control to which the treatment was compared.

We compiled a database from the *Zoological Record* (http://wokinfo.com/products_tools/specialized/zr/) containing publications between 1 January, 1990 and 16 June, 2016. The search was made with the subject descriptors "*Carnivora OR Canidae OR Felidae OR Hyaenidae OR Mustelidae OR Procyonidae OR Ursidae OR Viverridae OR Viverridae*". In total, we retrieved 48,894 titles. The titles and abstracts of these publications were imported to EndNote X7.0.2 (Thomson Reuters, New York, United States) and screened by the following search string: "*depredation OR stock OR poultry OR damage OR mitigation OR conflict OR control OR cull OR cow OR bull OR calf OR calves OR chicken OR hen OR ewe OR lamb OR pet OR cat OR hound OR pony OR ponies OR mule OR reindeer OR llama OR yak OR buffalo OR livestock OR cattle OR sheep OR goat OR horse OR pig OR dog OR attack OR camel OR donkey*". With this screening, we were left with 27,781 publications.

We manually screened the remaining publications to identify studies written in English that dealt with depredation of domestic animals (livestock and pets) by terrestrial mammalian large carnivores. Large carnivores were defined as species with an average body mass of >15 kg. We included studies of the 28 species listed by Ripple *et al.*[58] as well as coyotes and wolverines. These species were included because their body size can be larger than 15 kg[59,60] and conflicts associated with livestock depredation occur in different parts of their worldwide range[61,62]. After the first manual screening, two co-authors (AE and MT) read the 562 remaining publications in full. During the full text screening, we identified whether studies were correlational, quasi-experimental, or experimental. Experimental studies should include a randomized case-control study design, whereas studies were considered quasi-experimental if a case-control study design was used, but not assigned randomly. We also identified which studies included a quantitative measure of the effectiveness of an intervention. More precisely, the effectiveness of the intervention should be measured as the number of livestock killed, the number of attacks on livestock units, or the ability of the intervention to manipulate carnivore behaviour/movement in a way that is expected to reduce exposure of livestock to carnivore predation, e.g., by preventing trespasses into a baited exclusion area. We refer to all these instances as depredation events. A panel of two additional co-authors (JF and JVLB) additionally read all studies where intervention effectiveness was quantitatively evaluated, but where some uncertainty remained about the classification (correlational, quasi-experimental, or experimental), after which we collectively determined the classification. Only publications that included a quantitative measure of the effectiveness of an intervention and had an experimental or quasi-experimental study design, were included (i.e., all correlational studies were excluded). At this stage, we also removed one publication that had not gone through a scientific peer-review process. In the end, 21 scientific papers remained, describing 34 evaluations of intervention effectiveness.

Grouping interventions. We defined eight intervention groups: Change of livestock type, Enclosure, Guarding, Livestock guarding dog, Predator removal, Shock collar, Sterilization of carnivore, and Visual & Auditory deterrents. We aimed to group the interventions as specifically as possible to aid in the interpretation of between-intervention effectiveness. An even more specific grouping (e.g., trapping, shooting, poisoning, translocation instead of "predator removal") might have revealed more detailed information of between-intervention effectiveness, but was not possible due to the low number of studies that met our criteria (see Results). A broader categorization of interventions would involve a risk that the effect of one specific intervention could be masked by the effect of other interventions within the same category. Two co-authors (AE and JF) qualitatively summarized the results of the studies within each intervention group in the synopses presented in the Results section.

Data analyses. To allow a quantitative comparison of effectiveness between interventions, we calculated the relative risk (or risk ratio, RR) for carnivore depredation in treatment vs. control groups for each study[63]. For studies that measured the effect of an intervention in preventing trespasses, we used the number of incursions instead of the number of depredated animals. With the RR, we obtained an estimate for the effectiveness of the intervention (treatment) in comparison to using no intervention (control) in the same setting[63]. The relative risk is defined as the ratio between the probability of depredation by large carnivores in the treatment group and the probability of livestock depredation by large carnivores in the control group:

$$\text{Relative Risk(RR)} = \frac{a/(a+b)}{c/(c+d)} \tag{1}$$

where a is the number of depredated animals/units in the treatment group, b is the number of unharmed animals/units in the treatment group, c is the number of depredated animals/units in the control group, and d is the number of unharmed animals/units in the control group. In cases where there is no difference in the risk of depredation between the treatment and the control group, the relative risk is 1. When $\text{RR} > 1$, the risk of depredation is more likely to occur in the treatment group (with larger values of RR indicating a counter-productive intervention), and for $\text{RR} < 1$ depredation risk is higher in the control group (with values of RR indicating a greater intervention effectiveness as they get close to 0).

We took slightly different approaches to calculate RR, depending on the statistical units or reported statistics in the various studies. When possible, we used the mean number of animals in treatment and control herds, as reported in the original studies ($n = 1$). In studies where true numbers of herd size and depredation loss were reported for several herds separately, we first calculated the average herd size and the average numbers of depredated animals in treatment and control groups, and used these means to calculate RR ($n = 11$). For studies that reported the number of livestock units that suffered depredation and the number of livestock units that were unharmed, we used the number of livestock units for our calculation of RR ($n = 2$). In two cases, odds-ratios (OR) and hazards-ratios (HR) were reported in the original papers. Odds-ratios were converted to RR using an online odds ratio to risk ratio calculator[64], whereas we report the HR. For five papers that did not report herd sizes, we contacted the corresponding authors by email with a request for that data. All authors kindly responded to the request and we were provided data from two papers[32,36]. All studies considered in this review are included in Supplementary Table S1, and all studies with calculated/converted RR ($n = 17$) and reported HR ($n = 1$) are presented in Fig. 1.

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Author Contributions

A.E. participated in the planning of the project, did part of the publication review process, made the data analysis and wrote the main manuscript text. J.V.L.-B. participated in the planning of the project and in the review process and worked on the main manuscript text. M.T. did the main part of the literature screening and a large part of the publication review process, and reviewed the manuscript text. G.C. participated in the planning of the project, made the initial literature search, created the figure, and reviewed the manuscript text. J.F. planned the project, participated in the publication review process, and worked on the manuscript text.

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Review

Evaluating the efficacy of predator removal in a conflict-prone world

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ABSTRACT

Predators shape ecosystem structure and function through their direct and indirect effects on prey, which permeate through ecological communities. Predators are often perceived as competitors or threats to human values or well-being. This conflict has persisted for centuries, often resulting in predator removal (i.e. killing) via targeted culling, trapping, poisoning, and/or public hunts. Predator removal persists as a management strategy but requires scientific evaluation to assess the impacts of these actions, and to develop a way forward in a world where human-predator conflict may intensify due to predator reintroduction and rewilding, alongside an expanding human population. We reviewed literature investigating predator removal and focused on identifying instances of successes and failures. We found that predator removal was generally intended to protect domestic animals from depredation, to preserve prey species, or to mitigate risks of direct human conflict, corresponding to being conducted in farmland, wild land, or urban areas. Because of the different motivations for predator removal, there was no consistent definition of what success entailed so we developed one with which to assess studies we reviewed. Research tended to be retrospective and correlative and there were few controlled experimental approaches that evaluated whether predator removal met our definition of success, making formal meta-analysis impossible. Predator removal appeared to only be effective for the short-term, failing in the absence of sustained predator suppression. This means predator removal was typically an ineffective and costly approach to conflicts between humans and predators. Management must consider the role of the predator within the ecosystem and the potential consequences of removal on competitors and prey. Simulations or models can be generated to predict responses prior to removing predators. We also suggest that alternatives to predator removal be further developed and researched. Ultimately, humans must coexist with predators and learning how best to do so may resolve many conflicts.

1. Introduction

Predators can influence ecosystems through top-down control of the distribution and abundance of other species (Estes et al., 2011; Mills et al., 1993; Newsome et al., 2017; Pace et al., 1999). The loss of predators can therefore have profound ecological effects in certain contexts, including disease outbreaks, biodiversity loss, and ecosystem state changes (Myers et al., 2007; Ripple et al., 2014). There is evidence to suggest that ecological communities can exhibit dramatic shifts following the loss of predators (Crooks and Soulé, 1999; Pech et al., 1992; Ritchie and Johnson, 2009; Wallach et al., 2010), including changes at other trophic levels (Anthony et al., 2008; Atwood et al., 2015; Suraci et al., 2016). Although predators occur among diverse animal taxa (e.g., arthropods, molluscs, teleosts, raptors, canids, mustelids, etc.), vertebrate predators frequently conflict with humans, and many species are threatened (Ripple et al., 2014); they are therefore the focus of this paper.

Many predatory vertebrates are vulnerable to disturbances because they generally have slower life histories, higher investment in parental care, lower abundances, and patchy distributions (Purvis et al., 2000). Yet, predators are challenged by a perception of being a threat to human interests or safety. Indeed, predators can be considered hazardous to domesticated animals (Gusset et al., 2009; Mishra, 1997; Oli et al., 1994), prey species of economic importance (Dalla Rosa and Secchi, 2007; Henschel et al., 2011; Weise and Harvey, 2005), or human safety via direct conflict (Dickman, 2010; Gore et al., 2005; Löe and Röskaft, 2004; Penteriani et al., 2016). Consequently, predators are often negatively perceived and persecution of vertebrate predators has a long a history (Bergstrom et al., 2014; Kruuk, 2002; Reynolds and Tapper, 1996; Treves and Naughton-Treves, 1999). Competition with

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predators yielded many institutionalized campaigns against them dating back to ancient Greece and Rome, a trend that pervaded through medieval Europe and was exported to North America with emigrants in the 1700s (Reynolds and Tapper, 1996 and references therein). Today, state, regional, and agency-led programs to systematically control predator populations exist. Predator removal is carried out systematically via a number of methods and across various geographic scales (Bergstrom et al., 2014; Reynolds and Tapper, 1996), including poison baiting, trapping, hunting, and culling or via bounty or reward systems in public hunting or fishing events, but may also be more haphazard as retaliation for encroachment or interaction with humans or their property (e.g., farmer killing a wolf encroaching on their herds; e.g. Bergstrom et al., 2014; Treves and Karanth, 2003).

The significance of predators in ecosystems is well established yet their removal remains a component of the management toolbox. Owing to a lack of clarity pertaining to how and when removal can be expected to be successful, it may be difficult for management agencies to decide whether to proceed with predator removal when confronted with a problem. Furthermore, there is mounting opposition from advocacy groups (especially animal rights) and conservation-aware citizens that provide social inertia and pressure on animal control (van Eeden et al., 2017), which may complicate and influence decision-making (see Wallach et al., 2015). The science of predator removal therefore could benefit from an objective evaluation to identify successes and failures to both inform decision-making and identify lingering research gaps across multiple taxa (Treves et al., 2016; Eklund et al., 2017). Syntheses of this topic have focused on using meta-analysis, particularly for nesting birds (Côté and Sutherland, 1997; Smith et al., 2010, 2011), but it is challenging to apply such an approach across taxa and research paradigms (i.e., motivations). In this review, we evaluated these two competing hypotheses by considering of the available evidence for predator removal to determine whether predator removal is successful for wildlife conservation and management. We reviewed relevant literature and evaluated outcomes. In doing so, we propose a definition of success that can be applied to predator removal programs and we provide examples of success and failure in predator removals based on the following motivations 1) protection of domestic species, 2) preservation of prey species (e.g. economically important species or species at risk), and 3) mitigating risks of direct human-wildlife conflict. We conclude by considering evidence for the costs of failure in predator removal and a discussion of alternatives to predator removal. Although there are social and economic motivations associated with predator removal (Reynolds and Tapper, 1996; Engeman et al., 2002; Eklund et al., 2017; Swan et al., 2017), we focus on the ecological motivations aiming to synthesize perspectives on this practice. In this context, we refer to removal interchangeably with killing or lethal control. Removal may also refer to translocation, however, translocating predators has generally been demonstrated as ineffective for reducing conflicts (Athreya et al., 2011; Linnell et al., 1997; but see Hazin and Afonso, 2014). We focus on examples of aquatic and terrestrial vertebrate predators and ecosystems that include urban and rural areas. Moreover, we restrict the scope of this review to native predators. Invasive species are a global threat to biodiversity (Doherty et al., 2016) and the problems associated with biological invasions, although not necessarily unique or distinct from the problems that create nuisance predator conflict, are sufficiently different from a conservation and management perspective (see Doherty and Ritchie, 2016). Specifically, we incorporated evidence from published and gray literature on a variety of predatory taxa and from studies with varied predator removal motivations.

2. Approach

Based on preliminary searches and our perceptions regarding the quality of the evidence base (i.e., most studies had replication or included appropriate controls) we opted to conduct a qualitative literature review rather than a systematic review. Because the scope of our paper was broad, we used general search terms of the title, keywords, and abstract of papers in the Scopus search engine: "predator remov*", "cull", and "predator control" to identify relevant literature (asterisks are wildcards in the Scopus search engine). Reference lists in identified literature were consulted for additional resources and searches were repeated in Google Scholar. Articles were appraised at the title and then abstract level for inclusion in a synthetic table. Referring to our definition of success (see below), we sorted literature into successful and failed applications of predator removal and by the objective of the study in removing predators. All searches, filtering and analysis were conducted by the same individual (RJL) following input from co-authors. Bibliometric analyses were conducted in R (R Core Team, 2017). Figure plotted using the ggplot2 package (Wickham, 2009). Included studies were stored in a table (Supplementary material) with the predator species, motivation for removal, study duration, experimental method, our evaluation of success or failure (or equivocality), the removal method, a description of the study, and a citation (if not included in main text).

2.1. Defining successful predator removal

Success is a difficult outcome to define in predator removal because the motivations may be variable and idiosyncratic. Although we define success in the context of ecological responses, we acknowledge that successful predator removal must also consider the socioeconomic dimensions. For governments, the decision to implement predator removal may be a balance between satisfying demands of constituents for safety or prosperity against national or international agreements to protect species and economic externalities associated with wildlife, particularly ecological integrity. Nonetheless, we approach it from a conservation perspective insofar as removal must not cause long-term change or damage to the ecosystem while demonstrably benefiting the prey species, be they domestic animals (e.g. reduced rates of depredation), species at risk (e.g. increased local abundance or population growth rate) or of economic concern (e.g. increased harvest yield), or humans (e.g. reduced conflict or fear from predators). From an ecological and management perspective, we propose that successful predator removal would *reduce predator population to a size (or demographic state) that would not negatively impact the persistence of that population or its competitive status relative to mesopredators, but still provide demonstrable benefits to the prey species following predator removal* (Table 2).

Correlative methods used to evaluate success broadly match population trends of predator and prey species and ascribe outcomes (in terms of predator or prey densities) to the removal. Correlative approaches may lack the power to identify mechanisms (at least in the short term) driving population dynamics (Grubbs et al., 2016; Marcström et al., 1988) but can still provide insight into processes underlying prey population dynamics, particularly where experiments are infeasible. This can be observed in open marine systems where marine mammal culling programs may be tested by measuring correlations with fishery yields (Bax, 1998; Morissette et al., 2012). Shortcomings of retrospective analyses and correlational studies render it difficult to identify evidence supporting any positive effects accrued from predator removal, particularly in the context of different problems that arise where predator removal is being considered as a management strategy.

Experimental approaches to predator removal have more power to detect main effects on livestock depredation or species recovery. Controlled experiments using reference sites may be necessary but before-after-control-impact (BACI) studies can be useful to relate demographic trends to predator removal; however, BACI cannot account for changes to the environment that occur over time (e.g. Hervieux et al., 2014). Marcström et al. (1988) monitored grouse (*Bonasa bonasia*, *Lagopus lagopus*, *Tetrao tetri*) and capercaillie (*Tetrao urogallus*) populations across eight years, the first four with fox (*Vulpes vulpes*) and marten (*Martes martes*) removal followed by four years without killing. Although removal improved nesting success and increased adult density over time, the authors still cautioned that factors other than predator removal could have stimulated the increases. Simulations can be useful, such as Martin et al. (2010), in which the number of removed raccoons (*Procyon lotor*) necessary to achieve oystercatcher productivity (*Haematopus palliatus*) was simulated, suggesting that the specific number targeted should depend on the density of raccoons. Ernest et al. (2002) similarly used simulation to calculate the number of mountain lion removals necessary to reduce extinction risk of bighorn sheep (*Ovis canadensis*). Such frameworks are one solution for testing the efficacy of predator removal programs prior to implementation.

Attributing predator removal to livestock depredation, species recovery, or direct conflict with humans is complicated when the measurement of outcomes is restricted to relatively short intervals after predator removal. The period immediately following the action of predator removal is the period most likely to indicate a reduction in predator density and conflict and an increase in prey density, but this may decrease at longer post-treatment intervals (e.g. Engeman et al., 2006; Magella and Brousseau, 2001; Sagør et al., 1997; van Eeden et al., 2018). Short-term increases to nesting success or juvenile survival fail to consider density-dependence that manifests in the longer-term and cannot demonstrate success of predator removal when there is no demonstrated benefit to the population in subsequent years. Several studies observed increased nesting success of ducks following predator removal (Garrettson and Rohwer, 2001; Pearse and Ratti, 2004; Pieron and Rohwer, 2010), but a longer-term study conducted by Pieron et al. (2013) found that benefits to nesting did not carry over to the breeding population and therefore the latter studies provided no evidence to support predator removal (see meta-analysis by Côté and Sutherland, 1997). Although removal must generally be sustained (e.g. seasonally or annually) for benefits to be realized, success must be demonstrable and persistent over time (Bergstrom et al., 2014; van Eeden et al., 2018). Moreover, the benefits must outweigh the costs (Chessnes et al., 1968). A lack of longer term monitoring to determine whether predator removal was effective limits the power to interpret whether it was a successful intervention (van Eeden et al., 2018).

3. Synthesis

Our searches identified 141 empirical studies in which predator removal was studied by haphazardly culling predators with traps, guns, or poisons ($N = 87$), selectively removing predators ($N = 10$), controlled removal (i.e. a pre-specified number; $N = 21$), observing a natural decrease ($N = 1$), or in a simulation ($N = 10$). Studies were conducted on data from 1 to 78 years (mean ± SD = 9 ± 12 years). Most studies ($N = 104$) were conducted to evaluate whether predator removal could improve prey populations, followed by studies determined to evaluate impacts on domestic animals ($N = 28$) and direct interactions with humans ($N = 8$).

We evaluated a large number of these studies ($N = 37$) to have equivocal results, for example owing to a lack of statistical analysis, poor control to detect main effects, or because the study did not include sufficient information with which to make a determination about success (Fig. 1). Frequently, this arose because predator removal resulted in increased breeding success without evidence that this contributed to subsequent increases in the population. Although the scope of a study may have intentionally been focused on briefer time scales or questions, for our purposes and based on our definition of success we could not describe such results as indicative of success. Most studies we evaluated we determined to have failed ($N = 67$) owing to direct evidence that predator removal had either not succeeded in limiting the predator population or had no statistical demonstration of success in reducing livestock losses, increasing prey densities, or mitigating direct conflict with humans (Fig. 1). Studies that were successful ($N = 36$) demonstrated that predators were agents of additive mortality and that their removal resulted in subsequent increases in prey.

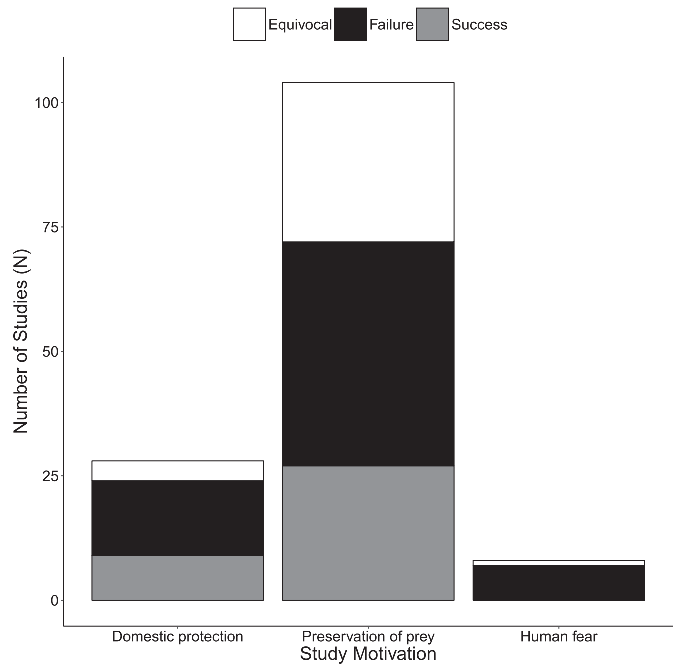

Fig. 1. Bibliometric summary of studies reviewed in this paper based on the three motivations for predator removal and the outcome. Studies are summarized in Table S1. Success was evaluated based on the definition in Table 2. Shading indicates our evaluation of the study as representing a success, failure, or equivocal outcome. Equivocality was ascribed for studies with inconclusive study design to determine success based on our definition.

An important caveat of this bibliometric approach is that studies that were deemed to be successful or failed may have been so because of some idiosyncrasy in the sampling protocol that could not be expected to be consistent among studies. When measurements were made (e.g. when prey abundance was measured) and when interventions were undertaken (e.g. what season the predator was removed in) could influence the outcome and the determination of success or failure. Successes or failures could also emerge consistently for similar taxa that were overrepresented in the literature, a limitation of the vote-counting approach that we used to present percentages.

The numbers presented in this bibliometric analysis are intended only to represent relevant information and a summary of published literature and are not intended to provide evidence for or against predator removal without further context. Below, we discuss factors associated with success and failure in predator removal with the objective of introducing more context, nuance, and interpretation of the literature covered in our bibliometric review along with other research focused on the relationship between predators and their prey in an effort to address the question of predator removal from a conservation perspective.

4. Factors contributing to success and failure

4.1. Resulting in success

A prevailing hypothesis is that predator removal can be implemented to achieve wildlife management objectives. We predicted that predator removal would be successful in some contexts, specifically, when implemented as a solution for short-term conservation challenges in which the return or replacement of the predator population in the long-term is not necessarily relevant to success (see Table 1).

Table 1
In our literature review we identified outcomes of experiments that yielded success or failure given three motivations for removing predators and based on our definition of success (see Table 2). Here we review 11 of the common outcomes of predator removal, two of which we considered to be successful and nine of which we considered to result in failure. Rows are populated with examples of the predator removed and references to literature demonstrating the given outcome for different motivations. References without species are those that removed multiple predators or predators were generalized (e.g. in a simulation). Refer to Table S1 for a comprehensive review of the appraised literature.

Outcome	Evaluation	Protection of domestic animals	Preservation of prey	Mitigating risks of direct conflict
Removal of "problem predators" known to instigate conflict reduces future conflict	Success	• *Canis latrans*: Blejwas et al., 2002, Till and Knowlton, 1983 • *Canis lupus*: Blejwas et al., 2002 • *Lynx lynx*: Stahl et al., 2001	• *Arctocephalus pusillus*: Makhado et al., 2009 • *Larus michahellis*: Sanz-Aguilar et al., 2009	
Measurable reduction in conflict or improvement in prey demographics while maintaining predators in the ecosystem	Success	• *Canis lupus*: Bradley et al., 2015 • *Lynx lynx*: Herfindal et al., 2005 • *Pogonias cromis*: George et al., 2008	• *Arctocephalus pusillus*: Weller et al., 2016 • Fletcher et al., 2010 • *Canis latrans*: Smith et al., 1986; Reynolds et al., 2010 • *Canis lupus*: Bjorge and Gunson, 1985, Boertje et al., 1996, Gasaway et al., 1983, Hayes et al., 2003, Hervieux et al., 2014, Keech et al., 2011, Potvin et al., 1992	
Conspecifics immigrate, replace predators, and conflict persists	Failure	• *Canis lupus* (Wielgus and Peebles, 2014 [refuted by Poudyal et al., 2016], Fernández-Gil et al., 2016 • *Caracal caracal* Bailey and Conradie, 2013; Conradie and Piesse, 2013) • *Puma concolor* (Peebles et al., 2013), *Canis dingo* (Allen, 2014, 2015)	• Greentree et al., 2000	
Removal was equally or less effective than non-lethal alternatives	Failure	• McManus et al., 2015, Palmer et al., 2005		
Predator population becomes imperilled by introgression with congeneric species or suffers depensation due to superadditive mortality	Failure	• *Canis lupus*: Chapron and Treves, 2016 • *Puma concolor*: Cooley et al., 2009 • *Ursus arctos*: Swenson et al., 1997		
Unknown mechanisms of compensatory mortality reveal short-term success (e.g. improved nesting or hatching rates) but no evidence of increases in abundance or density in subsequent years	Failure		• Amundson et al., 2013, Ellis-Felege et al., 2012, Littlefield and Cornely, 1997	
Conspecifics increase reproductive rate, predator population increases, and conflict persists	Failure		• *Canis latrans*: Knowlton, 1972	
Density independent conflict yields no benefits of removing predators on the incidence of conflict	Failure		• *Canis lupus*: Eggemann, 2015 • *Epinephelus itajara*: Frias-Torres, 2013 • *Larus cachinnans*: Bosch, 1996	• *Puma concolor*: Peebles et al., 2013, Teichman et al., 2016 • *Ursus americanus*: Obbard et al., 2014 • *Ursus arctos*: Artelle et al., 2016 • *Ursus thibetanus*; Huygens et al., 2004 • Wetherbee et al., 1994
Disappearance of a predator releases a mesopredator from competition, which maintains depredation	Failure		• *Canis dingo*: Wallach et al., 2010 • *Canis lupus*: Rutledge et al., 2012 • *Corvus brachyrhynchos*: Clark et al., 1995 • *Lynx lynx*: Palomares et al., 1995; Bodey et al., 2011, Prugh and Arthur, 2015	
Loss of predators deregulates pathogens within populations, resulting in increased disease-related mortality of prey	Failure		• Packer et al., 2003	

Table 2
Proposed maxims of predator removal summarizing important findings about successful applications of predator removal for management.

1	Predator removal is an interdisciplinary topic necessitating consideration of ecological, economic, sociological, political, and other dimensions.
2	Failure to consider ecological issues when initiating predator removal can harm the ecosystem.
3	From an ecological perspective, successful predator removal would reduce predator population to a size (or demographic state) that would not negatively impact the persistence of that population or its competitive status relative to mesopredators, but still provide demonstrable benefits to the prey species following predator removal.
4	The functional response of the predator is essential to consider because it influences the rate of depredation of prey species.
5	Targeted removal of problem individuals may be an effective application of predator removal (Swan et al., 2017), as opposed to indiscriminate or retaliatory killing, but it is logistically difficult to confidently identify culprit predators (Stahl et al., 2002).
6	Killing predators seems to generally result in an increase of local depredation of livestock resulting from demographic compensation via increased birth rates of predators (Knowlton, 1972), immigration (Sagør et al., 1997), or release of mesopredators/invasive species (Wallach et al., 2015).
7	Among humans, there are broad demographic differences in attitudes towards predators, with support for predator removal generally from older and more rural individuals (Andersone and Ozoliņš, 2004; Lüchtrath and Schraml, 2015).
8	Justifiable objectives for removal, especially the number to be removed, are necessary in planning predator removal rather than haphazard killing. Understanding the demographics and population dynamics of the predator is therefore essential. Adaptive management approaches can be applied to attempt sustainable removal that does not imperil the predator population (e.g. Martin et al., 2010).
9	Whether predator removal is actually effective at reducing conflicts or satisfying human attitudes towards predators is essential to its overall success as a management practice but the evidence for it is either equivocal or deficient.
10	There are increasing examples of non-lethal alternatives to predator removal, although many require scientific validation (Ogada et al., 2003; Okemwa, 2015).
11	Evidence that conflicts are mechanistically linked to depredation is important before beginning predator removal, along with evidence that predator removal will resolve the conflict, which can be tested via simulation (e.g. Morissette et al., 2012).
12	Coexistence with predators is possible and the most sensible way forward, but interdisciplinary research is necessary to continue to refine understanding of the human dimensions of predator removal (Carter and Linnell, 2016; Johnson and Wallach, In Press; Woodroffe et al., 2005).

4.1.1. Protection of domestic animals

When predators encroach on property or property development intersects with predator ranges, the presence of predators can become problematic if they threaten production animals (e.g. farms, ranch land, aquaculture facilities). Apex predators such as sharks, wolves (*Canis lupus*), dingoes (*Canis dingo*), lions (*Panthera leo*), tigers (*Panthera tigris*), cougars (*Puma concolor*), jaguars (*Panthera onca*), and leopards (*Panthera pardus*), for example, can affect the livelihoods of pastoralists, but so too can mesopredators (Davis et al., 2015) such as coyotes (*Canis latrans*), jackals (*Canis* spp.), crows (*Corvus corax*) and red foxes (*Vulpes vulpes*).

Predator removal can acutely reduce conflict when known predators are dispatched, but removals must often be of sufficient frequency or magnitude that they actually affect the population size or structure of the predator such that immigration does not compensate for removal (Bjorge and Gunson, 1985; Herfindal et al., 2005; Landa et al., 1999). For example, Bradley et al. (2015) found that wolf removal was successful at reducing livestock depredation if the entire pack was eliminated. Wagner and Conover (1999) killed coyotes and found that pastures with removal experienced slower rates of lamb depredation following removal (but see Treves et al., 2016 Supplementary material). In some cases, the success of predator removal is highly concentrated and neighbouring areas will suffer increased pressure; this may be a success on a small spatial or temporal scale but in general it would not achieve the desired outcomes (Santiago-Avila et al., 2018). Whether predators are actively targeting livestock or are encountering them opportunistically can affect success of the removal program. Odden et al. (2013) suggested that increased sheep production and decreased roe deer (*Capreolus capreolus*) density triggered a shift by lynx (*Lynx lynx*) towards sheep depredation, a type III functional response (i.e. preferentially targeting abundant species) that supports either lynx removal or roe deer conservation/supplementation. Moreover, there are different patterns of depredation for male and female animals. Males are generally more frequent livestock predators than females among solitary species, requiring selective removal to be successful (Felids: Odden et al., 2002; polar bear *Ursus maritimus*: Stenhouse et al., 1988).

Individuals within a population can differ in their propensity to depredate livestock for many reasons. Selective removal of individuals known to depredate livestock could be most effective in reducing future problems than haphazard culling (e.g. Woodroffe and Frank, 2005), the challenge being to accurately identify the offending individuals (Stahl et al., 2002; Swan et al., 2017). In our bibliometric review, we ascribed success to 40% of selective removals ($N = 10$) and only 19% in which predators were non-selectively removed by haphazard culling ($N = 87$) or public hunts ($N = 11$). Blejwas et al. (2002) found that only selective removal of coyotes following depredation events reduced subsequent depredations and not pre-emptive or non-selective removal. Some predators socially transmit knowledge that livestock are prey (e.g. to offspring; Mondolfi and Hoogesteijn, 1986) and systematic removal of known predators could instill wariness in predators by "hunting for fear" (Cromsigt et al., 2013) or social transmission of risk (e.g. invasive lionfish; Côté et al., 2014). In spite of a long history of predator persecution, we did not identify examples that support this, suggesting more research is needed to address this question.

4.1.2. Preservation of prey species

When prey species or populations are declining in abundance, there may be added pressure for managers to take remedial action (Lessard et al., 2005; Reynolds and Tapper, 1996). This is particularly true of economically important species that are hunted or fished or those that are at risk of extinction. Most examples of success were from studies aiming to preserve prey, although not on a relative basis as only 26% were deemed to be successful.

Many of the most important terrestrial game species are herbivores whose populations may be moderated by depredation. Removing predators can release prey species from predation and, so long as mortality from those predators is additive and not compensatory, the prey species could increase in following years and re-establish a higher abundance. The most successful examples of preserving prey by removing predators emerge from studies of predator removal in northern ecosystems with fewer trophic linkages and more direct influences of predators. Jarnemo and Liberg (2005) correlated roe deer (*Capreolus capreolus*) population growth to a disease outbreak that reduced red fox density and released the deer from predation. Moose and caribou (*Rangifer tarandus caribou*) survival has also improved following removal of wolves as demonstrated by several studies observing increases in prey abundance (Boertje et al., 1996; Hayes et al., 2003; Gasaway et al., 1983; Keech et al., 2011).

Prey species suffering from depensation may specifically benefit from predator release (e.g. Liermann and Hilborn, 2001; Stephens and Sutherland, 1999). For example, cormorant (*Phalacrocorax auritus*) culling preceded yellow perch (*Perca flavescens*) abundance increases in Lake Huron, suggesting that removing the predators assisted in rebound of its prey. Although human intervention is generally the mechanism for small population size of prey species, the added pressure of predation can still be linked to depensation (Gascoigne and Lipcius, 2004; Kramer and Drake, 2010; Liermann and Hilborn, 2001). Juveniles of species at risk such as marine turtles (Gascoigne and Lipcius, 2004) and salmon (*Oncorhynchus* sp.; Wood, 1987) that rely on safety in numbers to saturate predators during migration can undergo rapid declines from depredation (Hervieux et al., 2014; Liermann and Hilborn, 2001) and predator removal may facilitate increased juvenile survival and recruitment to such populations (Engeman et al., 2006; Pichegru, 2013; Hervieux et al., 2014; Makhado et al., 2009). However, it is not universally effective and alternate actions may have higher success than predator removal (Ratnaswamy et al., 1997). Improving juvenile survival may be a relevant management outcome for some species, but it does not necessarily improve population growth rate or abundance when there is density dependent or otherwise compensatory mortality and therefore studies that only observed increased egg hatching or juvenile densities were evaluated as equivocal without longer-term investigation (see Pieron et al., 2013). Considering generalist predators that consume fewer prey at smaller prey densities (type III functional response characterized by a logistic-type relationship between prey density and prey consumption, in which depredation is low until prey achieve a relatively high density and predators begin targeting that species), predator removal will not likely have a considerable effect because they would more likely switch to alternative prey instead of expending energy pursuing the rarer prey species (Murdoch, 1969; e.g. Middlemas et al., 2006). Specialization may also occur within species, in which cases the selective removal of specialized individuals can be an effective application of predator removal to release prey from depredation pressure (Sanz-Aguilar et al., 2009). Although predator removal may be effective when problems arise because of specialization, removal is not necessarily the most effective management option; alternatives such as exclosures may be more effective for reducing depredation and recovery of species at risk and should be tested (Rimmer and Deblinger, 1990; Reynolds and Tapper, 1996; Smith et al., 2011; Stringham and Robinson, 2015). However, the logistics of fencing off entire areas (e.g. breeding sites) to exclude predators are questionable and the long-term consequences can also be destructive (Hayward and Kerley, 2009).

4.1.3. *Mitigating risks of direct human-wildlife conflict*

Direct human-wildlife conflict has stimulated efforts to kill predators after attacks or a pre-emptive strike against future conflict (Gallagher, 2016). Few examples in the literature were identified that studied predator removal for relieving direct conflict between humans and predators ($N = 8$), with no examples of success. Fukuda et al. (2014) was determined to provide equivocal evidence for predator removal because it lacked proper control. However, they provide a salient example for future research in which predators that attack humans may learn to target them, in which case removing individual animals that have attacked humans could reduce future conflict (e.g. saltwater crocodiles *Crocodylus porosus*). There is a threat of animals habituating to humans, which may lead to more direct conflict in subsequent years and require removal of problem individuals (Linnell and Alleau, 2016). Predators infected with rabies or other diseases that increase conflict may also require lethal control (Linnell and Alleau, 2016). However, there is limited evidence that targeted killing of animals that have a history of interacting with humans reduces future conflicts, probably because such events are rare to observe, precluding experimentation or analysis (Swan et al., 2017).

4.2. *Resulting in failure*

The prevailing alternate hypothesis that we tested in conducting this literature review was that predator removal is not an effective tool for conservation or management of ecosystems. We reviewed the literature to identify research that described experiences or experiments with predator removal that have yielded perverse impacts on the ecosystem, or failure to achieve the desired objectives, which were different depending on the motivation for predator removal. Thus, we have divided this section into the familiar subheadings based on those motivations (see Table 1).

4.2.1. *Protection of domestic animals*

Protecting domestic animals by removing predators should reduce the rate of depredation on those domestic animals (Eklund et al., 2017). However, predator removal efforts fail when depredation rates do not respond to culling because the predator population compensates or is replaced by another predator. When there are multiple predatory species, Kissui (2008) found that pastoralists had difficulty identifying which species was responsible for livestock depredation and that higher visibility of lions during daytime caused them to be incorrectly accused. Targeted killing of leopards and caracals (*Caracal caracal*; Bailey and Conradie, 2013; Conradie and Piesse, 2013), cougars (Peebles et al., 2013), dingoes (Allen, 2014, 2015), and wolves (Wielgus and Peebles, 2014 [refuted by Poudyal et al., 2016]; Fernández-Gil et al., 2016) designed to reduce livestock depredation actually increased depredation in subsequent years (but see Bradley et al., 2015). Removal of adults may have triggered compensation via rapid replacement by immigrants in open systems (e.g. Baker and Harris, 2006; Bjorge and Gunson, 1985; Lieury et al., 2015; Sagør et al., 1997), enhanced local juvenile survival (Kemp, 1976; Peebles et al., 2013), or increased reproductive rates (Knowlton, 1972; Pitt et al., 2001). These demographic responses maintain or increase the number of local predators, stabilize the probability of further conflict, and represent distinct failures (Boyce et al., 1999; Sacks et al., 1999). Demographic responses of predators to culling may therefore render predator removal largely ineffective unless removal is so extensive that it alters predator demography on a broad scale, perhaps to impose an alternative stable state (Greentree et al., 2000; Herfindal et al., 2005). Removal can imperil the predators by accelerating their population declines if mortality is additive (or even super-additive; Creel and Rotella, 2010), for example when it instigates increased poaching (Chapron and Treves, 2016) or infanticidal behaviour (e.g. cougar: Cooley et al., 2009; grizzly bear *Ursus arctos*: Swenson et al., 1997; lion: Packer et al., 2009). Removal can also isolate remaining individuals, resulting in increased dependence on livestock in the absence of a group that would otherwise target wild prey (Bjorge and Gunson, 1985) or result in hybridization and degradation of genetic integrity (Rutledge et al., 2012). Short of predator eradication, removal generally does not protect domestic animals in the long-term.

Extensive removal of predators or eradication of top predators can also release subordinate species from competition (i.e. mesopredator release; Crooks and Soulé, 1999). Mesopredators can be of equal or greater possible or perceived threat to livestock and may be invasive species that become difficult to remove (Gross, 2008; Wallach et al., 2010), with cascading changes at other trophic levels (Hebblewhite et al., 2005; McPeek, 1998; McClanahan and Muthiga, 1988; Ritchie and Johnson, 2009). Mesopredator release can undermine predator removal and sustain depredation of domestic animals. In some cases, multiple mesopredators replace one extirpated top predator, complicating further control efforts.

4.2.2. *Preservation of prey species*

Removing predators theoretically reduces the extent to which prey species are removed from a population (e.g. Weller et al., 2016) given an assumption that predation contributes to additive and not compensatory mortality of the prey species, and therefore removal of the predators will directly contribute to an increase in prey (e.g. Flaaten, 1988). Evidently, this presupposes negligible effects of bottom-up processes (see Grange and Duncan, 2006; Elmhagen and Rushton, 2007), that the prey would not be limited by density-dependent resource limitation, and that prey is limited by a specific predator (Frias-Torres, 2013; Parker, 1984). However, most acknowledge both forms of regulation are simultaneously important in ecosystems and the relative importance of top down vs. bottom up control can shift in relation to productivity (Oksanen et al., 1981). Despite repeated efforts to connect predation to declines of economically important fishes, evidence for such a relationship is tenuous (Anon., 1986; Trzcinski et al., 2006). Eggemann (2015) also suggested that wolf depredation of moose (*Alces alces*) is density independent, meaning that reduced pack size could not succeed to increase moose escapement availability to hunters (also Kauhala et al., 2000). Similarly, Serrouya et al. (2017) showed that removing moose was effective for recovery of caribou in British Columbia because of apparent competition between wolves and caribou; although moose removal was not compared to wolf removal, this shows how predators can be incorrectly persecuted if alternative solutions to maintaining prey densities are not explored. Using Ecopath with Ecosim for mass balanced simulation based on foraging arena theory, Morissette et al. (2012) tested whether marine mammal removal would increase fishery yield and suggested that it would more likely lead to reductions than increases because of limited actual competition between fisheries and whales (see also Gerber et al., 2009). Yodzis (1998) also predicted a decline of fisheries yields during cape fur seal (*Arctocephalus pusillus*) culling programs that were proposed to increase yields. Lessard et al. (2005) simulated seal removal and predicted an increase in Pacific salmon smolt survival but suggested that it might increase predatory fish populations, which would replace the seals in depredating the smolts; generalist predators such as seals often regulate multiple populations within a community and removing these predators can lead to disequilibrium in the ecosystem.

The trophic position of the predator contributes to its functional response to changes in prey, an important factor when considering removal (Bowen and Lidgard, 2013). Removing a mesopredator will most likely yield compensatory depredation by other mesopredators (Clark et al., 1995). Elimination of a top predator could release herbivores from control, resulting in extensive damage to landscapes and changes to habitat suitability that cause shifts in the community (Bertness et al., 2014; Ripple and Beschta, 2006). Hunters can compensate for predation mortality but will generally remove highly fit phenotypes (Allendorf and Hard, 2009) whereas predators target weak or diseased prey (Genovart et al., 2010; Krumm et al., 2010; Quinn and Cresswell, 2004); loss of predators can then proliferate disease within prey populations (Packer et al., 2003) and can spill over to infect domestic animals (Cross et al., 2007). Even when removal is successful in the short-term, compensatory processes may regulate predator populations such that removal is ineffective in the long-term (e.g. Donehower et al., 2007). Long-term studies or simulation models are necessary to detect effects of predator removal on prey (see Costa et al., 2017).

4.2.3. Mitigating risks of direct human-wildlife conflict

Predatory animals are often perceived as threats to human safety in spite of infrequent interactions and small odds of actual conflict relative to many other habitual activities such as driving cars (Slovic, 1987). According to the social amplification of risk framework, empirically rare events contribute disproportionally to concern among the public and lead to economically, socially, or ecologically illogical responses (e.g. fear of flying; Kasperson et al., 1988). This framework could be applicable to human-wildlife conflict if the perceived risk of direct attack on humans is higher than the actual risk. Sharks are often victimized by social amplification of risk, which has resulted in publicized and prominent state-sponsored programs that aim to cull sharks near beaches (e.g. Wetherbee et al., 1994; Gallagher, 2016). The major failure of shark culling programs, however, has been exemplified by a lack of evidence that it actually decreases attacks (Wetherbee et al., 1994), arising in part because many large predatory shark species are migratory and therefore there is a low probability that locally-based actions will be effective once they cease and sharks from surrounding and more distant areas move into these managed areas continually (Holland et al., 1999). Gray and Gray (2017) found limited support among patrons for lethal control of sharks. Correspondingly, we found no research asking whether culling programs actually affected the perception of risk by patrons; safety is difficult to guarantee, and a perception of safety may encourage reckless behaviour (e.g. ignoring key risk factors associated with shark attack) that increases the likelihood of negative encounters with sharks (e.g. swimming offshore). The legacies of such efforts could instead just be negative public perception of the animals, increased fear, and impoverished conservation status of the targeted species. Perversely, Teichman et al. (2016) found that human-cougar conflict was higher in areas of cougar trophy hunting yet Gilbert et al. (2017) suggested that economic value of cougar populations exceeds the costs because they control deer populations that cause costly collisions with vehicles. Skonhoft (2006) discussed this in terms of Scandinavian wolves, suggesting there is an equilibrium possible between the economic losses of lucrative moose depredated by wolves (*Alces alces*) and gains in terms of reduced vehicle-moose collisions and damage to foliage caused by moose browsing in the winter, emphasizing the value of maintaining predators and the costs of predator removal.

There was no direct evidence that removing predators changes outcomes for human-wildlife conflict. Obbard et al. (2014) found no influence of black bear removal on future conflict with humans and Artelle et al. (2016) perversely observed that removal of grizzly bears was followed by no difference in future conflicts rather than a reduction. Although data in Artelle et al. (2016) do not suggest causality, it does indicate that removal was not successful at mitigating conflicts. Treves et al. (2010) further suggested that the number of black bears (*Ursus americanus*) killed by hunters did not reduce, and was actually correlated with increases in, reports of conflict in subsequent years (although it is relevant to note that complaints were not necessarily related to predatory activity of bears, but also property damage). Apparently, overlap between humans and black bears increases during poor years when urban resources aggregate the animals, meaning that removal of predators in these years has disproportionately high impact on the population (Baruch-Mordo et al., 2014).

5. Discussion

We evaluated the two opposing hypotheses considering the (a) success or (b) failure of predator removal as in the conservation and management of ecosystems. We selected a qualitative approach to testing these hypotheses by searching for published evidence of success and failure. We identified examples of success but ultimately found much more consistent evidence for failure (Table 1). Evidence that removing predators achieved conservation-sound outcomes was context-specific (see Section 4.1). Removing predators presumes that ecosystem-level responses are predictable (Ramsey and Norbury, 2009), yet theoretical and empirical evidence often suggests the contrary (Bax, 1998; Ruscoe et al., 2011; Yodzis, 2000). An exception may exist in ecosystems where predators and prey are very closely linked (e.g. northern terrestrial ecosystems) or the prey are suffering from depensatory population declines associated with depredation by predators with a type II functional response. Although predators can influence ecosystems (Holt et al., 2008; Nelson et al., 2004), other factors can make the ultimate response of an ecosystem unpredictable, even with rigorous scientific evaluation. The full range of complexity at the ecosystem scale is poorly understood, especially as it pertains to processes such as parasites in ecosystem dynamics (Roche et al., 2012). This was observed consistently in study designs, which were often either short in duration or lacking in control, rendering it difficult to avoid type I error.

Many governments are responsible for establishing and maintaining protected areas, zoning property (for agriculture or developing buffers), and formulating wildlife management regulations (Rands et al., 2010; Treves et al., 2017). Strong policy based on available evidence can contribute to effective conservation of predators in many ways, including the establishment of suitable regulations and protected areas (Linnell et al., 2001). However, predators are important components of the landscape not just in designated areas but also in areas of human use (Dorresteijn et al., 2015; Gilbert et al., 2017; Kuijper et al., 2016; López-Bao et al., 2017). When conflicts arise, retaliatory killing by local stakeholders may be understandable but can undermine conservation efforts for both predators and the broader ecosystem. It is important to accurately document the movements and actions of depredating species and maintain records of conflicts to determine the appropriate course of action and to advance the science of predator conflict to develop resolutions. In its present form, our findings suggest that success in predator removal is highly contextual and should not be assumed by management without rigorous testing.

5.1. Alternative actions for managing human-predator conflict

Human-predator conflict challenges managers because depredation can be damaging to some livelihoods and traumatic for individuals (e.g. pastoralists, aquaculturists, fishers; Butler, 2000; Graham et al., 2005; Mishra, 1997; Patterson et al., 2004). Attitudes of retaliation (Holmern et al., 2007; Kissui, 2008; Thorn et al., 2012) are understandable, even though conflicts tend to be isolated incidents (Cozza et al., 1996; Chavez and Gese, 2005). Economic losses to depredation are, however, generally less than those attributable to other sources of mortality such as disease (Breck and Meier, 2004; Frank, 1998; Mazzolli et al., 2002; Mizutani, 1999; Kissui, 2008; Rasmussen, 1999). Livestock often comprises smaller components of the diet of predators than assumed by some pastoralists (Allen, 2015; Boast et al., 2016; Davis et al., 2015). Kaltenborn and Brainerd (2016) suggested that restoration of predators to large population sizes and then opening recreational hunting seasons could be a more effective alternative to balance socioeconomic objectives. However, sustainable harvest limits are incalculable without demographic data (Packer et al., 2009; Treves, 2009). Moreover, human harvests tend not to be non-selective for problem predators (Sunde et al., 1998) or can undermine conservation (Creel and Rotella, 2010). Where livestock comprise a more important food source for predators, conservation or restoration of native prey sources could mitigate losses (Meriggi and Lovari, 1996; Odden et al., 2013). Husbandry practices can alternatively reduce conflict with wildlife without ecological issues or social controversy (e.g. Jackson and Wangchuk, 2004; Johnson and Wallach, 2016). Fencing is already used by pastoralists (Hayward and Kerley, 2009) with variable success (Eklund et al., 2017). Birthing of calves during a short period may facilitate predator satiation, reducing depredation on farms (Palmeira et al., 2008). Calves can also be kept centralized and away from edges (Palmeira et al., 2008). Deterrent devices (e.g. fladry) also hold promise for reducing depredation (Ogada et al., 2003; Okemwa, 2015), evidenced by a 93–97% reduction in depredation of aquaculture sites using a non-lethal deterrent by seals (Götz and Janik, 2016). In scientific study, predator removal should be tested against realistic alternatives because in some cases deterrents are just as effective (Harper et al., 2008; Ratnaswamy et al., 1997) and may be more economical (McManus et al., 2015). When conflicts do arise, the costs can be offset with subsidies (Bulte and Rondeau, 2005; Dickman et al., 2011; Mishra et al., 2003). Challengingly, some governments do not have the resources to support conservation initiatives or compensate farmers for losses and in others, the systems are not developed to properly address the problems (Chen et al., 2015). In developing countries, this leads to continued persecution of predators, maybe out of bare necessity to maintain herds in some cases (Dar et al., 2009), but improved education and validation of effective alternatives hold promise for resolving conflict.

When prey species decline, hunters may support and lobby for predator removal (Franzmann, 1993) and conservation movements may support protection of species at risk by controlling their predators. Species persistence is considered a priority of conservation science and is often nested within the laws of regional and national management plans. Predator removal may appear to be a logical solution for maintaining adult populations and increasing juvenile survival during species declines; however, our results clearly show that studies are needed to demonstrate this (Oro and Martínez-Abraín, 2007). Deterrents or barriers can reduce predator access to endangered species and may be more effective and economical in many scenarios (Shivik, 2006; Smith et al., 2011; Yurk and Trites, 2000). Emerging solutions that use sensory modalities to mitigate predation can also yield promising results, for example, Neves et al. (2006) tested taste aversion methods of reducing nest predation of endangered roseate tern (*Sterna dougallii*). Guardian animals have also shown promise for livestock (Meadows and Knowlton, 2000; Smith et al., 2000; van Bommel and Johnson, 2012) and species at risk (King et al., 2015).

The willingness to pay for hunting/fishing for large predators may be high, species of recreational importance tend to have higher acceptability and be better conserved, and illegal hunting can undermine ecological, economic, and sociological objectives of wildlife management. Therefore, managed hunts or fisheries targeting predators have been proposed as a solution to reduce poaching, maintain stable predator populations, fund conservation initiatives, and increase acceptability of some predators (Creel et al., 2016; Gallagher et al., 2016; Kaltenborn and Brainerd, 2016; Lindsey et al., 2007). Sportspeople and guides can keep watch for illegal activity, particularly in remote areas and activity can also reduce predator activity (Harper et al., 2008). The result of such managed hunts would, however, probably result in random, rather than targeted, removal that would not likely have any effect on the rate of predator conflict (Packer et al., 2009; Treves, 2009) unless it can be confidently applied to maintain a smaller predator population without resulting in depensation.

5.2. Social and economic costs of failure

Killing by people is the largest threat to the conservation of many predators (Kissui, 2008; Ripple et al., 2016; Woodroffe and Ginsberg, 1998). In spite of the problems with implementing predator removal for management, human-wildlife conflict persists (Treves and Karanth, 2003) and predator persecution and removal will likely continue, particularly when there is direct conflict between a human and a predator. Research on human attitudes towards predators is plentiful, relating demographics to perception of predators (e.g. nationality, gender, age; Andersone and Ozoliņš, 2004; Lüchtrath and Schraml, 2015). Attitudes towards predators depend greatly on exposure and experience as well as cultural values towards wildlife, for example, rural people tended to favour control more than urban dwellers (Andersone and Ozoliņš, 2004).

It should be possible to quantify the carrying capacities and demographics of predators to maintain a smaller population of predators to limit conflicts, although in general we found that this is likely only possible via continued intervention (e.g. Landa et al., 1999). Careful calculation and monitoring would be essential for this because of unanticipated changes in demography arising from human-induced mortality and the potential for additive or super-additive (rather than compensatory) mortality following intervention that imperils the predators (Creel and Rotella, 2010). Indeed, Bradley et al. (2015) found that partial wolf pack removal was effective for mitigating livestock depredation while maintaining wolves in the northern Rocky Mountains; however, more rigorous methods can be implemented to calculate removal targets. The justification for predator removal targets and how they are defined is often weak and idiosyncratic. Strategies can include controlled removal with a stated goal (e.g. 50% reduction), haphazard culling (e.g. opportunistic removal), or selective removal (e.g. removing problem individuals), with variation in the expected outcomes. In Western Australia, the social licence and evidence for culling has recently been questioned (Legge et al., 2017). Simulation to determine the optimal number of predators to be removed to achieve conservation objectives can assist with validating predator removal prior to implementation (Ernest et al., 2002; Martin et al., 2010).

Modern management of predator conflicts must include stakeholders (Breitenmoser, 1998) and consider predators in an ecosystem context rather than as individual species in conflict with humans. There is limited evidence that retaliation against a species or pre-emptive culling decreases conflicts or generates a sense of security in landscapes where predators exist. This points to a failure to consider the broader-scale processes that regulate predator populations and ecosystems (Berlow, 1999) as well as a lack of understanding of the human dimension of attitudes towards wildlife that promote negative perceptions. Moreover, it ignores the positive impacts of predators and intact ecosystems by regulating herbivores, mesopredators, and disease. Predator removal can also disconnect public perceptions of nature by acclimating people to manipulated and arguably depauperate ecosystems (Wallach et al., 2015), an outcome that can shift baselines and reduce support for conservation initiatives (Chapron and Treves, 2016). This problem is exemplified in coyote control, where Berger (2006) calculated a long-term expenditure of over a billion dollars for coyote removal programs in the United States that were intended to improve the sheep farming industry and wool production had no measurable benefits across 78 years of data.

5.3. Study context and future research directions

Evaluating the contribution of predators and the success of predator removal to conservation efforts has been attempted elsewhere in the ecological literature. Meta-analysis is well suited to this problem because it reduces type II error (compared to vote-counting approaches) and weights studies by their sample size; however, it can be overly influenced by few studies with large sample sizes. Whereas meta-analysis is suited to analyzing studies with similar intervention, endpoint, and subjects (Eysenck, 1994; see Smith et al., 2010, 2011 for effective examples of this), it is constraining for broad topics such as predator removal (Haddaway et al., 2015), which is conducted for many different reasons on a variety of taxa, making it difficult to generate reliable numerical assessments that could be considered relevant across socioecological contexts. Instead, we opted for a qualitative review with bibliometric analyses to reveal successes and failures with appropriate consideration to context. There are lessons to be gained from viewing many different, often disparate predator removal attempts through a common lens and identifying how varying inputs (e.g. motivations, taxa) contribute to outcomes to address future problems that arise. Provided that future studies on this topic address some of the deficiencies in experimental design noted here, there is potential to improve the quality of the evidence base such that meta-analysis within the context of a systematic review should be possible and will help to ensure evidence-based environmental management in the future (Sutherland et al., 2004).

6. Conclusion

Human-wildlife conflict will persist with direct impacts on ecosystems globally. Desire to manage predator populations will therefore continue in spite of growing conservation concern for many predators (and in some cases, recovery of their populations; Curtis et al., 2014).

Our review suggests that the success of predator removal depends on the motivation and design of the effort because of the variability in success identified across studies. More research is needed to determine whether predator removal reduces direct conflict with humans or human fear. However, there was some circumstantial support that removing predators facilitated prey recovery and some evidence that it assisted with protection of domestic animals. Nonetheless, a main takeaway from this review is the inconsistencies and idiosyncrasies of outcomes. Predator control should be pre-empted by research to justify the action and set removal targets, with anticipated outcomes stated and follow-ups planned to evaluate the action. Alternative actions may be equally or more effective and should be studied in parallel when possible. Some studies are not designed to detect main effects of predator removal and are instead retrospective and correlative because predator removal may not always be motivated by conservation (Treves et al., 2016). How the decision to remove predators is arrived at typically remains unclear. Although much can be learned from experimental approaches (e.g. Lieury et al., 2015), they can be costly, ethically controversial, and require the removal of predators for didactic purposes. Simulation approaches or predictive modelling have the potential to become increasingly useful tools prior to implementing removal in order to project whether the predator removal is likely to achieve the desired outcomes (e.g. Martin et al., 2010; Morissette et al., 2012; Yodzis, 1998). However, such efforts need to consider and account for many potentially confounding external variables such as food availability and competition in order to conclude whether predator removal is likely to be successful as well as the potential for immigration compared to compensation (Creel et al., 2015).

6.1. Promoting coexistence

Coexistence with predators is the desired way forward for many (Bergstrom, 2017; Carter and Linnell, 2016; Johnson and Wallach, 2016; Woodroffe et al., 2005), and there are increasing examples that predators can persist even among dense human populations (Chapron et al., 2014; Elliot et al., 2016; Gilbert et al., 2017). Indeed, predators play important ecological roles in rural areas and even in urban regions (Gilbert et al., 2017). We propose that paradigms positing predator persecution as a positive management intervention require reassessment (see also Graham et al., 2005). However, interdisciplinary approaches that consider socio-ecological perspectives (e.g.; Bisi et al., 2007; Elliot et al., In Press; Hill, 2015; Kaltenborn et al., 2006) will be integral for determining how human perceptions, values, and attitudes towards predators are shaped, and how they can be accounted for to meet the needs of humans and predators and minimise conflict in an increasingly crowded landscape.

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RESEARCH ARTICLE

Killing wolves to prevent predation on livestock may protect one farm but harm neighbors

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Data Availability Statement: Certain privacy issues arise from sharing our data. By agreement with the Little River Band of Ottawa Indians, we try to protect the privacy of the livestock owners involved in depredation events. Therefore, we have redacted the precise locational information of complaints and interventions. Hence, the data being included with this submission does not include the precise locations (section information) of each data point. The complete data, with precise location information, can be obtained by written request to: W. Frank Beaver, Director, Natural Resources Department, Little River Band of Ottawa

Abstract

Large carnivores, such as gray wolves, *Canis lupus*, are difficult to protect in mixed-use landscapes because some people perceive them as dangerous and because they sometimes threaten human property and safety. Governments may respond by killing carnivores in an effort to prevent repeated conflicts or threats, although the functional effectiveness of lethal methods has long been questioned. We evaluated two methods of government intervention following independent events of verified wolf predation on domestic animals (depredation) in the Upper Peninsula of Michigan, USA between 1998–2014, at three spatial scales. We evaluated two intervention methods using log-rank tests and conditional Cox recurrent event, gap time models based on retrospective analyses of the following quasi-experimental treatments: (1) selective killing of wolves by trapping near sites of verified depredation, and (2) advice to owners and haphazard use of non-lethal methods without wolf-killing. The government did not randomly assign treatments and used a pseudo-control (no removal of wolves was not a true control), but the federal permission to intervene lethally was granted and rescinded independent of events on the ground. Hazard ratios suggest lethal intervention was associated with an insignificant 27% lower risk of recurrence of events at trapping sites, but offset by an insignificant 22% increase in risk of recurrence at sites up to 5.42 km distant in the same year, compared to the non-lethal treatment. Our results do not support the hypothesis that Michigan's use of lethal intervention after wolf depredations was effective for reducing the future risk of recurrence in the vicinities of trapping sites. Examining only the sites of intervention is incomplete because neighbors near trapping sites may suffer the recurrence of depredations. We propose two new hypotheses for perceived effectiveness of lethal methods: (a) killing predators may be perceived as effective because of the benefits to a small minority of farmers, and (b) if neighbors experience side-effects of lethal intervention such as displaced depredations, they may perceive the problem growing and then demand more lethal intervention rather than detecting problems spreading from the first trapping site. Ethical wildlife management guided by the "best scientific and commercial data available" would suggest suspending the standard method of trapping wolves in favor of non-lethal methods (livestock guarding dogs or fladry) that have been proven effective in preventing livestock losses in Michigan and elsewhere.

Indians, fbeaver@lrboi-nsn.gov, 231-398-2191. Requests should describe the rationale for the data request, as well as what steps will be taken to ensure the privacy of livestock owners involved in depredations remains protected.

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Introduction

Large carnivores, such as gray wolves, *Canis lupus*, are difficult to protect in mixed-use landscapes because some people perceive them as dangerous and because they sometimes threaten human property and safety. Traditionally, governments kill wild animals in an effort to prevent threats to property and safety [1]. However, a recent summary of peer-reviewed studies that employed experimental or quasi-experimental tests of interventions against carnivore attacks on domestic animals in farms raised doubts about the functional effectiveness of lethal methods [2]. Namely, most tests of lethal methods showed no effect or counter-productive effects (higher livestock losses after intervention), and numerous tests contained biases or flaws that preclude reliable inference [2]. Two tests using quasi-experimental designs showed minimal, regional effect of various lethal methods [3] and a strong, local effect of government trapping and aerial shooting [4], respectively. But none provided the highest standard of evidence [2], which are random-assignment experimental tests of an intervention without bias in sampling treatment, measurement, or reporting [5, 6]. Higher standards of evidence were applied to tests of non-lethal methods generally, and two such tests applied the highest standards that also proved effective in preventing predation events on domestic animals (depredation). The two methods were fladry (a visual deterrent effective against wolves only, thus far) and livestock-guarding dogs [7, 8]. A recent controversy over killing wolves in the Northern Rocky Mountains (NRM) illustrates the difficulty of forming scientific consensus on the effectiveness of lethal methods for preventing depredations when standards of evidence are not consistent.

Two teams [4, 9] came to opposite conclusions when analyzing very similar data from the same region and similar period for the Northern Rocky Mountain wolf population. A deeper look suggests that inferences drawn from these quasi-experimental tests are weakened by uncontrolled variables (Box 1).

Box 1

One test included only wolf-killing by aerial gunning and several ground-based methods from 1989–2008 [4], whereas the other included all permitted wolf-killing, including public hunting, from 1987–2012 [9]. The latter of these two analyses found that killing more wolves was followed by more livestock losses the following year, using a negative binomial regression model controlling for multiple variables [9]. However, that test did not account adequately for the time series underlying several variables that increased over time. For example, over time the wolf population increased in size and also spread geographically, thereby exposing more farm animals to depredations. Because the amount of wolf-killing increased over time as (a) recolonizing wolves left the protection of a national park and wild areas, and (b) policy changes introduced wolf-hunting in addition to killing by government agents [4, 10, 11], we should expect the predictors (wolf-killing, livestock exposed, and wolf distribution) to rise over time in parallel with the observed rise in domestic animal losses over time, which would make a statistically significant association spurious if the time trend were not accounted for properly. Another team conducted the same analysis with the same data while accounting for time series trends and statistical misspecifications, and results suggest killing wolves instead led to an increase in attacks on cattle in the same year and fewer attacks the following year, relative to no killing [12]. However, this analysis seems to have eliminated the possibility of an underlying effect of wolf population size and did not consider the geographic spread of wolves, an approach that remains to be validated [12]. Proper statistical control for exposure (encounters between wolves and domestic animals) might require a measure of geographic spread of wolves, not just wolf and domestic animal abundances regionally. The remedy would have required spatial information at scales below that of the region. The authors of the analysis of wolf-killing between 1989–2008 incorporated spatial information, yet did not extend spatial analyses sufficiently, and limited their data to a time period when only government wolf-killing was legally allowed [4]. They found a reduction in risk of recurrence subsequent to wolf-killing within a wolf pack territory. The reductions appeared significant and high in magnitude after an entire pack was killed, and appeared significant but lower in magnitude when only part of a pack was killed, compared with no removal [4]. The analysis was restricted to the affected wolf pack territory, despite the researchers' own work documenting how partial removal of wolves could scatter survivors beyond their original pack range [11, 13]. Therefore, the analysis of risk of recurrence of depredations should have examined neighboring areas and even more distant consequences. The importance of examining livestock loss beyond the edges of wolf pack territories had been noted [14]. We examine the analysis of [4] in greater detail in the Discussion.

We tested the hypothesis that two treatments (lethal and non-lethal intervention) following verified depredations had different effects on the risk of a recurrence (occurrence of a subsequent depredation) at that site and at neighboring sites at two larger geographic scales. We tested that hypothesis because the common justification for lethal interventions worldwide is that eliminating problem individuals, or regional predator reductions, will delay or curtail future losses immediately, and for at least one year until wolves are replaced [15]. We retrospectively examined data collected by state and federal agents in the state of Michigan, USA, from 1998–2014, using methods similar to [4], with two main differences. The first difference was that we examined spatial scales beyond the site of the intervention, so we could detect spill-over effects up to a radius of 16.25 km from the site of the intervention (neighborhood of township scale; see Methods section below). The second difference was that we included 2 distinct interventions: lethal and non-lethal interventions (pseudo-control, see below). Our analysis was retrospective and treatments had not been assigned randomly, thus the highest standard one might achieve would be a silver-standard experiment [2]. With data on the history and locations of events and interventions, we were able to draw stronger inference than a simple comparison of means between interventions. But quasi-experimental tests might be confounded by the effect of time passing (before-and-after) as carnivores, livestock, and people respond to changing conditions and other aspects of the environment change independently.

We had to consider potential bias in treatment. Field agents apparently made subjective judgments about where to implement lethal intervention when that was permitted by the federal government (Table 1 & [16]). Therefore, we had to contend with a pseudo-control as follows: At times, the state agency opted not to kill wolves or opted to offer farmers non-lethal deterrents, and the state advised the complainant on protection of livestock. The latter intervention involved communications and possible deployment of non-lethal deterrents (see below) with unknown characteristics or consistency. We also considered potential measurement errors–that may have been systematic, not random errors–associated with unreported wolf-killing and unreported depredations, both of which occur in neighboring Wisconsin [2, 14], and are believed to occur in Michigan as well [17, 18].

Table 1. Periods for wolf-killing policy signals in WI and MI, derived from Refsnider [16], ESA sec. 4 10(a)(1)(A) and Humane Society of the U.S. et al. v. Jewell (U.S. District Court, D.C., 5 1:13-cv-00186-BAH Document 52, 2014).

Period start (mm/dd/yyyy)	Period end (mm/dd/yyyy)	Federal status	Culling**
4/15/1994	3/31/2003	Listed as endangered	not allowed
4/1/2003	1/30/2005	Down-listed to threatened	allowed
1/31/2005	3/31/2005	Relisted	not allowed
4/1/2005	9/13/2005	Sub-permit for culling issued	allowed
9/14/2005	4/23/2006	Sub-permit rescinded	not allowed
4/24/2006*	7/31/2006	Sub-permit for culling issued	allowed
8/1/2006	3/11/2007	Sub-permit rescinded	not allowed
3/12/2007	9/28/2008	Delisted	allowed
9/29/2008	5/3/2009	Relisted	not allowed
5/4/2009	6/30/2009	Delisted	allowed
7/1/2009	26/1/2012	Relisted	not allowed
1/27/2012	4/14/2012	Delisted	allowed

*States identical except sub-permit issuance on 6 May 2006 to Michigan instead of issuance on 24 April 2006 to Wisconsin [16].

**Killing a wolf that posed a threat to human safety was always allowed under ESA sec. 11(a)(3).

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Selection bias–or the tendency to apply different interventions to different subjects or locations based on some anticipated outcome–can powerfully affect the results of experimental tests [5]. In short, we controlled for spatial variation by comparing an intervention site to itself, but we could not control for the intervenors' subjective decisions. In the Discussion, we identify and discuss potential sources of bias in the dataset provided to us.

Because of the caveats above relating to the strength of inference we might draw from the uncontrolled 'experiment' conducted by the State of Michigan, we regard our conclusions as preliminary in the same way that other recent published studies should be considered, pending gold-standard experiments [4, 9, 12]. These studies offer new inferences and testable hypotheses about the effect of interventions, rather than conclusions about the functional effectiveness of the interventions *per se*.

Materials and methods

Data sources

The State of Michigan continuously monitored complaints about wolves and annually monitored the wolves themselves, across the Upper Peninsula (42,610 km^2). We used the federal government's published reports for Michigan's minimum, late-winter wolf population (https://www.fws.gov/midwest/wolf/aboutwolves/mi_wi_nos.htm), supplemented by Michigan data provided to the Little River Band of Ottawa Indians after their request through a federal Consent Decree. Michigan estimated wolf numbers by snow-track surveys, summer howling, and aerial telemetry of VHF radio-collared wolves primarily [19]. The exception was wolf-year 2012 when Michigan did not census its wolf population, so we interpolated the midpoint of the 2011 and 2013 estimates (Fig 1). Our study spanned wolf-years 1998–2015 (calendar-years 1998–2014); a wolf-year t was 15 April of year t-1 to 14 April of year t.

Michigan provided Wolf Activity Reports with 379 entries. The U.S. Department of Agriculture Wildlife Services (USDA) investigated many of these incidents since 1990 under state contract [20]. Hereafter, we refer to Michigan when referring to government responses to wolf-related complaints, whether by state or USDA field personnel. We discarded 149 entries that consisted of different categories of wolf encounters: observations, perceived threats to

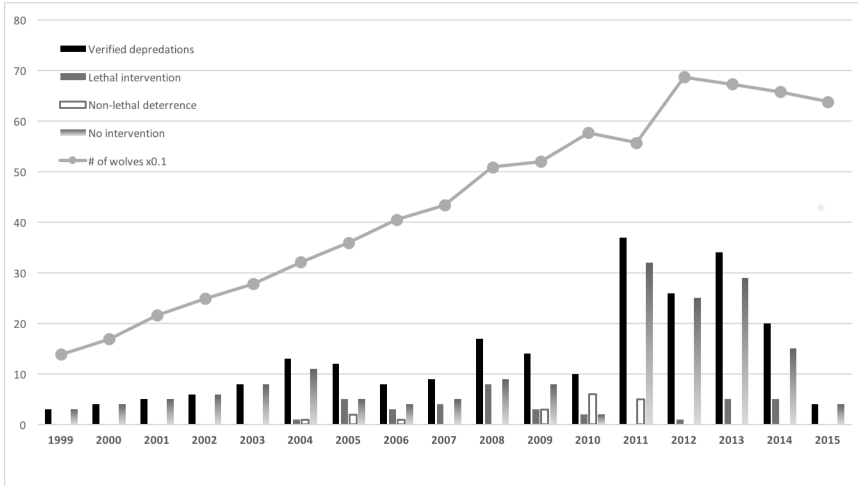

Fig 1. Annual Michigan wolf abundance, verified depredations and interventions. Michigan's annual wolf abundance (divided by 10 to fit the same y-axis as other variables) and two treatments after verified depredations. The x-axis shows wolf-years, which span 15 April of year t-1 to 14 April of year t. Overall n = 230 depredations.

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humans or domestic animals, or wolf interactions with hounds engaged in training or hunting, but which lacked verified depredations on a private property. Discarding perceived threats to humans should prevent the introduction of some biases, because the Wolf Activity Report entries suggested that one complainant's 'threat' was another's 'encounter' that did not result in official complaint, investigation or intervention. Considering the potential biasing effects of perceived threats that did not lead to a complaint (false negatives), perceived threats that were simply observations (false positives), and a complete lack of any such reports before 2002, we felt more secure setting aside all entries lacking depredations and ensuing verification. Also, wolf interactions with hounds occur under very different circumstances than depredations in our region [21–24]. In sum, we retained 230 complaints for screening as described below.

We screened complaints for verification and independence between depredations. During the study period, Michigan verified 499 livestock or farm animals injured or killed by wolves in 230 complaints. Depredations were classified as independent if they occurred on a different date.

Michigan responded in several ways to predation: communication only, provision of non-lethal deterrents, or lethal intervention. Lethal intervention consisted of live-trapping on or near the complainant's property for several days to weeks after a depredation, and if successful, the state shot one or more wolves caught alive in leg-hold traps (n = 98 wolves killed overall, with lethal interventions following depredations in 37 occasions, and resulting in the deaths of 56 wolves in 32 interventions and 0 wolves killed in 5 occasions); in a few cases landowners shot wolves after receiving state permits. We omitted 32 cases in which wolves were killed but were not involved in depredations; only two of which occurred in the same townships (geopolitical mapping area of 36 miles^2 or 92.16 km^2) as lethal intervention during our study. We did not include the public hunting season at the end of 2013 because those removals were not targeted at known complaint sites [25]. Non-lethal deterrence was used primarily when no losses occurred in the Wolf Activity Reports, so most such interventions were excluded by our screening criteria above.

We refer to any intervention that did not lead to wolves dying as non-lethal, which implies only that no wolves were killed, but related actions may have entailed a range of communications with the complainant and other responses, including the provision of non-lethal deterrents in some cases. All interventions included communications with complainants but we had no data to determine if such communications differed between lethal intervention and non-lethal. Non-lethal deterrents included one or more of the following: cracker shells, hazing kits, live-traps, lights, or fencing with various materials, including fladry (a loose flagging hung at regular intervals on fence-lines [26]). We also classified live-trapping (i.e., attempted lethal interventions) that resulted in no wolves killed (n = 5) as 'non-lethal'. Differences in non-lethal methods implemented at different sites could be attributed to costs, judgments by state agents about effectiveness in a given situation, willingness of livestock owners to deploy certain techniques, or other undocumented factors. Because of the small sample of occasions when non-lethal deterrents were deployed after depredations (n = 18), we pooled all interventions that did not lead to wolf-killing as non-lethal, due to insufficient information on whether the deterrents were actually implemented by the farmer.

A true control would have enacted all the same procedures and time spent on the complainant's property without killing wolves, or installing any non-lethal infrastructure. Therefore, we refer to our non-lethal intervention classification as a pseudo-control because it may have included different communications or a judgment by a state agent that lethal intervention was not likely to succeed. However, given that the federal permit for the state to use lethal control was issued and rescinded several times without regard to events on the ground (Table 1), we infer that the two treatments we analyzed were largely selected because of the broader governmental timelines rather than the events at a particular property. Independent decisions about the availability of lethal intervention would reduce the risk of treatment bias [2]. Regardless, this study represents a silver-standard experiment with possible treatment biases that must be considered preliminary and examined carefully (see Discussion).

With the preceding criteria, our primary sample of 230 depredations (or depredation events, by which we mean a verified, independent wolf depredation incident in the Wolf Activity Report) consisted of 32 depredations followed by lethal intervention, and 198 followed by non-lethal intervention.

Analyses

We used geopolitical sections (regular units of 1 mile^2 or 2.56 km^2) as the smallest mapping units, following [27]. Sections can be read from commercially available road atlases. Sometimes more precise locations were also provided, but inspection revealed that many of these were simply the latitude and longitude of the center of the section. Virtually every livestock pasture lay within the borders of a single section. All livestock pastures were on private property of much less than 1 section in area (average farm size was 0.3 miles^2 or 0.68 km^2 in the Upper Peninsula [28]). The state did not record ownership of pastures or the tenure status of complainants. All depredation events are presented in **S1 Data File** with certain personal details, property information, and precise locations redacted for privacy.

We determined the sequence of depredation events by reference to the date of the complaint on the Wolf Activity Reports. We calculated the delay to recurrence as the interval in days to the next event in the same vicinity (2.56 km^2 section or larger geographic unit, see below). If there were no subsequent events in the vicinity that calendar year, we censored that observation of delay to recurrence at 31 December of the same year. Virtually all depredations occurred in the warmer months [20], with most events occurring in the period March-October (90%) and only 3% occurring in November or December, echoing results from Edge et al. [18]. Livestock in the Upper Peninsula are kept within enclosed pastures year-round, usually in small farms, and thus equally available to wolves throughout the year [18, 29]. Therefore, our decision to measure and censor the delay to recurrence within the calendar year provided at least 60 days to detect an effect in 97% of events (recurrence at section scales occurred within a median of 13 days if it occurred the same year). Had we extended the time horizon as in [4], we saw a risk of conflating the recurrence of depredation events by later wolves with the treatment applied to prior wolves.

We also examined if depredations recurred at two larger spatial scales. At the intermediate scale of townships (36 miles^2 or 92.16 km^2), the area used for measuring recurrence approximated half the core area of an average wolf pack territory [30]. At our largest spatial scale, the neighborhood of townships (320 miles^2 or 829.44 km^2) was equivalent to 9 contiguous townships centered on a depredation event and >4 times the average core area of a wolf pack territory [30]. For analyses of risk of recurrence at the township and neighborhood scales, we replaced the fixed geopolitical unit with a square buffer of the same area centered on each depredation event (Fig 2). We detected no difference in the sequence of depredation events for particular areas when using a circular buffer, possibly due to the coordinates for depredation incidents obtained from the Wolf Activity Reports frequently placing the incident in the center of a section, which both buffer shapes contained. The square buffer was preferred based on its consistency with the underlying Public Land Survey System (USGS, https://nationalmap.gov/small_scale/a_plss.html) layer containing the spatial subdivisions we based our three spatial

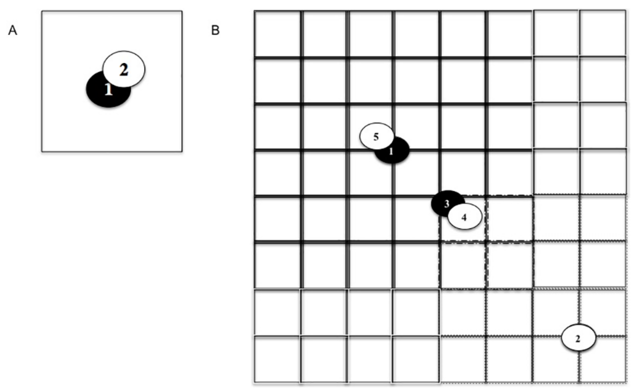

Fig 2. Measuring recurrence between depredation events at multiple spatial scales. Each small rectangle is a section (1 mile^2). Each oval is a single event of verified depredation. A 1 indicates the first of such events in its vicinity and year, and higher numbers are subsequent events in chronological order of occurrence in the same year. The intervention is shown with colored ovals: lethal (black), non-lethal intervention (open); and events within the same section are depicted as overlapping each other partially (1 and 2 in A; 1 and 5 or 3 and 4 in B). **A**: Smallest scale of analysis where the vicinity is limited to the section. Datum 1 stratum 1 measures the number of days between events 1 and 2 with lethal intervention. Because there is no event 3 within the vicinity, datum 1 stratum 2 measures the number of days between event 2 and the end of the calendar year but switches to non-lethal intervention (open oval). **B**: Medium-scale of analysis where rectangles are sections in a township (36 miles^2 centered on event 1). Solid black grid lines indicate buffer around event 1; dotted gray lines indicate buffer around event 2; black dot-dashed lines indicate overlap between buffers. Because event 1 and event 2 are not in the same township-sized buffer, they generate datum 1 and datum 2 with lethal intervention and non-lethal intervention, respectively. Datum 1 stratum 1 measures the number of days between event 1 and event 3. Although event 3 is also within the buffer of event 2 (within black dot-dashed lines), it was assigned to event 1 because it was nearest by Euclidean distance. We did not measure the number of days between events 2 and 3 because event 3 was already used to create datum 1 stratum 1; in this way, we avoided double-counting events. Next, events 3 and 4 are collapsed (treated as a single event) because they occurred in the same section *sequentially*. Because event 3 was followed by lethal intervention (black oval), the resulting single collapsed event was classified as lethal intervention. We then measure datum 1 stratum 2 as the number of days between event 4 and 5, remembering that the collapsed event is classified as lethal even though 4 is followed by non-lethal intervention (any collapsed set of events with a lethal intervention event among them is assigned to the lethal intervention set). Finally, datum 1 stratum 3 is measured by the number of days between event 5 and the end of the calendar year and assigned to non-lethal intervention. If event 2 had zero other events in its township area (not shown), then datum 2 stratum 1 would be measured to the end of the calendar year. A similar process was followed for the largest spatial scale of neighborhood of townships (320 miles^2).

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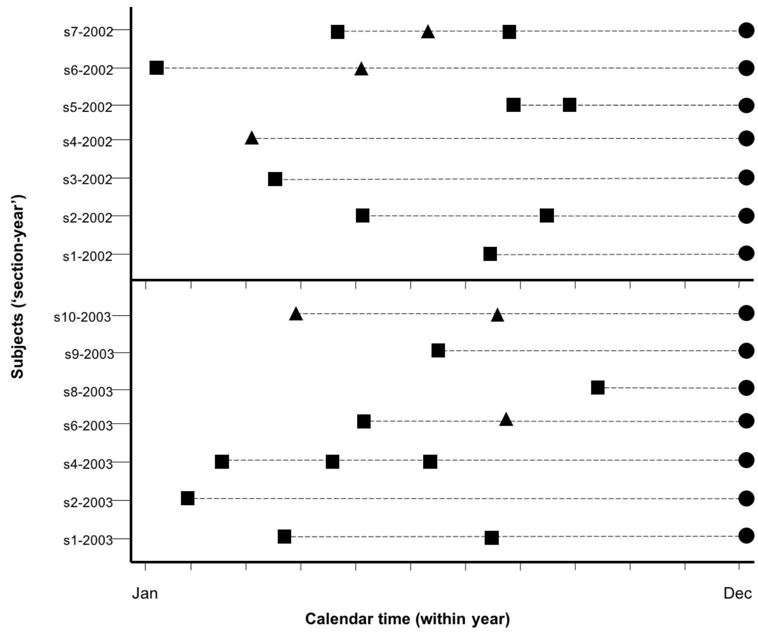

Fig 3. Transforming depredation records to a survival analysis format. We present lethal interventions (triangles) and non-lethal interventions (squares) connected by a dashed line that measures the delay to recurrence or censorship (circles). We illustrate using data from two subsequent years. Subjects are identified as combinations of vicinity (section, township or neighborhood) and year (i.e.: section s1-2002) on the y-axis. The first figure for each subject represents when the first depredation event in that year occurred, which is the date follow-up started for that 'section-year'. Each subject then follows a chronology of subsequent depredation events through the year, treated with either intervention. Stratum 1 considers the initial intervention implemented and the delay to recurrence to the next depredation event, or censoring if no other events occurred (i.e.: first figure to second figure in dashed line for each subject). Stratum 2 considers the next sequence of depredation events (i.e.: delay from second figure to third figure). Due to our construction of subjects, a particular section (sections 1, 2, 4 and 6, for example) can appear in multiple years, represented with a different 'section-year' combination (for example, s1-2002 and s1-2003).

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scales on. We measured delay to recurrence in that buffer, repeating the process for subsequent depredation events using each event's buffer (i.e., a moving window). The process of assigning depredation events at scales larger than the section (Fig 2) was designed to avoid pseudo-replication (once the effect of a pair of events was measured at a lower scale, that estimate of delay to recurrence was never used again at larger scales).

The use of three spatial scales allowed us to detect depredation recurrence beyond the original sites (spill-over effects) following interventions. Our process for collapsing depredation events (Fig 2) produced a conservative assessment of spill-over effects because we eliminated pseudo-replication of estimates of risk of recurrence across scales. The disadvantage of our approach was declining sample sizes that reduced the power of the tests at larger scales and thereby potentially increased Type II error.

Statistical tests

We measured delay to recurrence in days between each pair of successive depredation events as in Figs 2 and 3, and produced survival functions for each treatment following Hosmer, Lemenshow & May [31]. A survival function describes the probability of observing a time interval between two depredation events, T, greater than some stated value t, $S(t) = P(T>t)$, where t is days. Thus, survival functions provide, for every time t, the probability of 'surviving' (in this case, not experiencing a depredation event) up to that time, and describe these probability distributions (survival distributions). Survival analysis comprises a set of statistical methods used to quantify and test survival function differences between treatment groups of subjects [32].

At the smallest spatial scale, we defined our subjects as the sections in which depredations occurred. Thus, sections are analogous in biomedical research to the patient receiving treatments. In this case, the section receives lethal or non-lethal treatment of wolves. Note that this differs from prior research that defined wolf pack territories as the subjects [4].

Subjects enter the analysis after the initial depredation event, and remain in the analysis until December 31st of that year; hence, our subjects arise from a particular vicinity (i.e., section, township or neighborhood) in a particular calendar-year (1998 to 2014) (Fig 3). Depredation events, along with their respective treatments and measures of recurrence were organized into strata based on their order of occurrence for each subject (Fig 2). Each year a new set of strata was created, starting with stratum 1 again. The end of each calendar year represented a 'reset' point after which we assumed independence of subjects because both wolves and livestock are mostly removed from each other's reach until the next grazing period. Based on this classification of subjects and strata, we clustered our analysis on a unique identifier reflecting a particular vicinity-year combination, e.g., ID_TRS_Yr [33]. This approach accounts for potential spatial and temporal auto-correlation among strata within subjects, e.g., all depredation events for the same subject experienced during a particular year are assumed correlated. It also avoids pseudo-replication of observed depredation events from the same subject as if they were independent of other depredation events in that same year, e.g., ID_TRS_2000's stratum 1 and stratum 2 observations are correctly identified as belonging to the same subject, rather than belonging to two different subjects (pseudo-replication). In the Discussion, we examine potential pseudo-replication concerns in our dataset and in prior approaches.

We employed general and stratified log-rank tests (Chi-squared statistic) to compare the survival distributions for delay to recurrence in both treatments. We then used a conditional Cox recurrent event, gap time model [31] to compare the associations between treatments and risk of recurrence. The Cox model allowed us to estimate hazard ratios (HR) for relative risk of recurrence between treatments by characterizing how the hazard function (H) changed as a function of survival time and subject covariates; $S(t) = e^{-H(t,x,\beta)}$, where t is study time (the period of observation or follow-up period after inclusion in study until end of the calendar year), x is a covariate we describe below, and β is the parameter estimate of x.

The *stratified* conditional Cox model accounts for risk of recurrence for the ith depredation event being influenced by the occurrence of a previous (i-1)th depredation event and the treatment following it, so that each subject is included in the risk set (the number of subjects experiencing a depredation event) for the ith depredation event only if it experienced the (i-1)th depredation event. For example, in our section-scale analysis, 31 subjects experienced a first recurrent depredation event, whereas 120 did not experience any recurrence (Stratum 1, Tables B & C in S1 File).

The stratified Cox model considers only those subjects experiencing that first recurrent depredation event in the second stratum (Stratum 2, n = 31; Table A in S1 File), repeating the process for subsequent strata until end of the calendar year. The stratified Cox model allowed us to estimate general treatment effects while accounting for event order and the treatment applied to the previous event.

We ran univariate and multivariate conditional Cox models at each spatial scale. Univariate models included only our response variable (delay to recurrence) comparing our two treatments, whereas multivariate models incorporated calendar year. Including calendar year was essential because the gray wolf was down-listed to threatened in Michigan on April 1, 2003, and subsequently went through 12 or more reclassifications and permit issuances that precluded or allowed wolf-killing by the state ([34], and Table 1]) as the protection afforded wolves was reduced or increased.

Given that treatment effects could change over time as wolves, livestock, people, and ecosystems might change with environmental conditions, we also ran multivariate models incorporating a time-varying covariate (tvc) for treatments [31]. Our tvc consists of an interaction of treatment with study time. The use of a tvc is strongly recommended for evaluating and handling non-proportional hazards (PH), given PH is an underlying assumption of survival modelling [31]. A non-proportional hazard occurs when the treatment effect changes over time (instead of remaining constant) relative to the pseudo-control, so that the hazard ratio for the treatment changes over time. Hence, if the parameter estimate for the tvc were found to be significant, the conditional Cox model with tvc would be more robust and reliable than without the tvc because it corrected for non-proportional hazards in our treatments. When the tvc is not significant, its inclusion in the model is not warranted.

Authorities on stratified Cox models also express concerns about strong inference depending on the risk set per stratum [31, 35]. The latter authors did not settle on a particular number observations per treatment per stratum; however, the Cox models depend on a measure of variability within-strata to detect deviations from chance differences between treatments, therefore we excluded strata with <10 depredation events or which lacked events for both treatments. This conservative step left us with 3 strata at the section scale, 1 stratum at the township scale, and 2 strata at the neighborhood scales (S1 File). Thus, our final sample at the section scale consisted of 151 subjects (independent section-years) with 199 depredation events, including 56 recurrent depredation events; the final sample at the township scale consisted of 125 subjects with 125 depredation events, including 24 recurrent depredation events; and the final sample at the neighborhood scale consists of 106 subjects with 125 depredation events, including 25 recurrent depredation events (S1 File).

We assessed the robustness of models to within-subject correlation by running a variant of a random-effects approach called frailty models ([35]; S2 File). If high-risk and low-risk farms exist due to factors extrinsic to treatments, years, or the tvc, then subject identity should inform gap time models [17, 36]. Frailty models assess the goodness of fit of the treatment variable by including random effects of subject identity [35], which is considered useful when recurrence time might be influenced by unmeasured factors [31, 37].

We also built models with subsets of the data to evaluate potential confounding effects and robustness of the primary models described above. We built a model with data 'post-2003', after lethal management was episodically permitted, and by reclassifying lethal management with zero wolves killed as 'lethal' because the infrastructure and attendant human influences would be the same whenever traps were laid regardless if wolves were live-trapped and killed. We refer to the latter condition as 'traps placed'. We present alternative models in supporting information (S2–S4 Files).

Finally, we used Spearman rank correlations (r_s) to correlate delay to recurrence with number of wolves killed for lethal treatments only and for 'traps placed'. We conducted all analyses in Stata 14 (StataCorp, College Station, TX, 2015; protocol DOI: 10.17504/protocols.io.j2rcqd6).

Results

Between 1998 and May 2014 there were 199 depredations in Michigan with as many management interventions. Of the 199, 31 resulted in lethal intervention (16%) and 168 resulted in non-lethal intervention (84%) (Fig 1).

Table 2. General and stratified log-rank (χ^2) tests examining difference between treatments' (lethal and non-lethal) survival distributions (measuring risk of recurrence) after wolf depredations, for all spatial scales.

	Spatial scale of analysis		
	Section	**Township**	**Neighborhood**
SUBJECTS AND 'FAILURES'			
TOTAL DEPREDATION EVENTS	199	125	125
Failures (recurrent events)	56	24	25
SURVIVAL FUNCTIONS			
Log rank test (χ2)	0.27	1.44	0.08
p-val	0.603	0.23	0.772
Stratified Log-rank test (χ2)	0.48	-	0.28
p-val	0.488	-	0.593

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Section scale

Log rank tests could not distinguish the survival functions between treatments (df = 1, general survival functions test: $\chi^2 = 0.27$, P = 0.604; stratified [by order of depredation events for subjects] test: $\chi^2 = 0.48$, P = 0.488; Table 2). All univariate (treatment only) and multivariate (treatment and calendar-year) Cox models suggest that lethal intervention was associated with a non-significant reduction in risk of recurrence when compared to non-lethal intervention (Table 3). The section-scale models including a time-varying covariate (tvc) were not significant, so the PH assumption was not violated (tvc P>0.05). The multivariate model including treatment and year suggests lethal intervention only weakly reduced risk of recurrence (slowing recurrence) by 27%, but that was not a statistically significant difference (HR = 0.73, P = 0.326; Table 3). This model also revealed an increasing risk of recurrence (hastening recurrence) by 9% each calendar-year (HR = 1.09, P = 0.022). Lethal intervention was not significantly different from non-lethal intervention in our frailty model (HR = 0.48, P = 0.158; Table A in S2 File), with the model suggesting significant frailty (omitted or unobserved

Table 3. Main results of Cox models measuring risk of recurrence between treatments (lethal and non-lethal) implemented after wolf depredations, for all spatial scales.

	Spatial scale of analysis					
	Section		**Township**		**Neighborhood**	
PROPORTIONAL HAZARD MODELS	***Interv***	***Interv & year***	***Interv***	***Interv & year***	***Interv***	***Interv & year***
Standard cox (stratified)						
Intervention HR (SD)	0.77 (0.22)	0.73 (0.23)	0.48 (0.308)	0.46 (0.29)	0.80 (0.340)	0.72 (0.34)
p-val	0.36	0.326	0.255	0.224	0.644	0.486
year HR (SD)	-	1.09 (0.04)*	-	1.05 (0.05)	-	1.14 (0.07)*
p-val	-	0.022	-	0.28	-	0.024
Standard cox with tvc (stratified)						
Intervention HR (SD)	0.48 (0.21)*	0.46 (0.21)	1.87 (1.47)	1.78 (1.38)	0.84 (0.62)	0.80 (0.63)
p-val	0.099	0.091	0.425	0.458	0.818	0.778
tvc(Intervention) HR (SD)	1.01 (0.01)*	1.01 (0.01)	0.97 (0.01)**	0.97 (0.01)**	0.99 (0.01)	1.00 (0.01)
p-val	0.057	0.068	0.001	0.001	0.928	0.852
year HR (SD)	-	1.09 (0.04)*	-	1.05 (0.05)	-	1.14 (0.07)*
p-val	-	0.023	-	0.281	-	0.023

Significance

* if p-val < .05

** if < .01.

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Table 4. Spearman correlation between delay to recurrence and number of wolves killed after depredation events followed by lethal intervention (wolves killed ≥ 0), for all spatial scales.

	Section	Township	Neighborhood
Spearman's rho	0.107	0.212	0.295
p-val	0.5591	0.2994	0.1354

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covariates) remaining in the model (P = 0.006). For those depredation events followed by lethal intervention, we found no correlation between delay to recurrence and the number of wolves killed (Spearman's rho = 0.107, P = 0.559, Table 4; 'traps placed': Spearman's rho = 0.076, P = 0.657; Table C in S3 File).

Township scale

Our dataset consisted of 125 depredations, 26 followed by lethal intervention (21%) and 99 followed by non-lethal intervention (79%). Log rank tests could not distinguish the survival functions between treatments (df = 1, general test: $\chi^2 = 1.44$, $P = 0.23$; Table 2). Likewise, all Cox models revealed no significant differences between treatments (Table 3). The township-scale models including a tvc were significant, suggesting the PH assumption was violated (tvc P<0.05). Hence, we focus our analysis on the model including the tvc. Lethal intervention increased risk (hastening recurrence) by 22%, but this was not statistically significant (treatment HR = 1.78, P = 0.458). However, our tvc, which accounts for non-proportional hazards, hints at a minimal (3%) reduction in risk over follow-up time (tvc HR = 0.97, P = 0.001). Calendar-year was not significant (HR = 1.05, P = 0.281). Differences between treatments were not significant in our frailty model (HR = 0.45, P = 0.242; Table A in S2 File). For those events followed by lethal intervention, we found no correlation between delay to recurrence and the number of wolves killed (Spearman's rho = 0.212, P = 0.299, Table 4; 'traps placed': Spearman's rho = 0.233, P = 0.224; Table C in S3 File).

Neighborhood scale

Our dataset consisted of 125 depredations, 26 followed by lethal intervention (21%) and 99 followed by non-lethal intervention (79%). Again, log rank tests could not distinguish survival functions between treatments (general test: $\chi^2 = 0.08$, $P = 0.772$; stratified test: $\chi^2 = 0.28$, $P = 0.594$). Similarly, all Cox models revealed no differences between treatments (Table 2). The neighborhood-scale models including a tvc were not significant, so the PH assumption was not violated (tvc P>0.05). Lethal intervention only weakly reduced the risk of recurrence (slowing recurrence) by 28% but this difference was not significant (treatment HR = 0.72, P = 0.486; Table 3). We found a statistically significant increase in risk of recurrence (hastening recurrence) of 14% every calendar-year (HR = 1.14, P = 0.024). The frailty model showed no significant differences between treatments (HR = 0.80, P = 0.67; Table A in S2 File).

For those events followed by lethal intervention, we found no evidence of a correlation between time to recurrence and the number of wolves killed (Spearman's rho = 0.295, P = 0.135, Table 4; 'traps placed': Spearman's rho = 0.161, P = 0.395; Table C in S3 File).

For all spatial scales, all effects of treatment remained consistent for the 'traps placed' condition, when limiting the data to post-2003 depredation events, 'skip-a-year' dataset and when removing a special case (S2–S4 Files).

Discussion

We retrospectively evaluated whether lethal interventions by the State of Michigan in response to wolf predation on domestic animals (depredations) between 1998–2014 resulted in lower risk of recurrence of depredations than if no wolves were killed. We found the delay to recurrence of depredations was unrelated to the number of wolves killed at all spatial scales. We found lethal management did not significantly shorten or lengthen the interval to the next depredation relative to non-lethal interventions. A small, statistically insignificant reduction in the risk of depredation at the section level was offset by a similar and also statistically insignificant increase in the risk of depredation at the township scale, which is about half the size of a wolf pack territory, and then a similar decrease in risk at the scale of neighborhoods of townships, which are four times larger than the average wolf pack territory [30]. None of these differences were statistically significant using a battery of tests.

Our methods or alternative models accounted for potential violations of the proportional hazards assumption, unlike a prior study of wolves in the Northern Rocky Mountains (see below); accounted for within-subject correlation; were unaffected when we restricted analysis to the period after 2003 when lethal interventions first became legal; and accounted for a change in definition of lethal methods to include the installation of lethal methods that did not kill any wolves (S3 File). There is evidence for the effect of lethal intervention changing slightly over the course of a single calendar year at the township scale, through a minimal reduction in risk over follow-up time. We also detected variation between individual farms in their time to recurrence of depredations. Given the apparent, net ineffectiveness of lethal intervention and the uncertainty about potential biases in a retrospective analysis of sparsely documented government interventions, we recommend ethical, gold-standard, random-assignment experiments be used before further lethal management is authorized to prevent depredations.

Overall, our analysis suggests that any potential beneficial effects of lethal interventions locally would be offset by detrimental effects for neighboring farms in the same township. If the small, local improvements were considered biologically, ethically, or economically important to one farm, then one would also have to admit the associated costs to neighboring farms and the biological, ethical and economic importance to that farm. Therefore, given the evidence available, we cannot conclude that lethal management had the desired effect of preventing future livestock losses.

Over the 17 years of our study, the risk of depredation increased by 9 and 14% per year at the section and neighborhood (smallest and largest) scales, respectively, in our main dataset. However, this effect of year is insignificant in our post-2003 dataset (S4 File). In addition to changes in wolf densities locally that may have occurred, there may also have been changes in proportion of pasture, prey density, land cover, farm size, road density, among other variables that predict depredations at local scales [17, 38]. Also, prior work indicated smaller packs were more often implicated in livestock depredations than larger packs [23]. Therefore, the notion that higher densities of wolves locally will result in more depredations is not well supported, as opposed to the idea that a recolonizing population encounters more livestock as a result of recolonizing more and more of their historic range over time.

We present our results guardedly rather than as a definitive conclusion about effectiveness because of insurmountable uncertainties about the government data. Retrospective analyses to evaluate the effectiveness of interventions to prevent predation on livestock are fraught with uncertainty because of various biases or challenges presented by field conditions [2]. For example, treatments were not assigned randomly and changing conditions over time locally were not documented. The unintentional error may have been random but we are unable to rule out systematic error (bias), whether intentional or unintentional. The government dataset we analyzed had undocumented variability in data collection and intervention, including possible systematic selection bias affecting which areas received which interventions.

Selection (or enrollment) bias would arise if subjects entered the study under varying conditions that affected outcomes. All sections containing farms (subjects) entered our study because of a verified depredation, but subjects entered at different times and some farm owners might have responded to depredations in undocumented ways including poaching wolves. Likewise, attrition bias would arise if subjects left the study for reasons that were not random with respect to their outcomes. This would occur systematically if a subset of the interventions led farmers not to complain in the future despite facing depredations, or to take matters into their own hands, as above. Compensation was offered throughout the study as well as state-financed non-lethal deterrence when lethal intervention was unavailable, so attrition by withholding complaints seems unlikely to have been frequent or widespread. However, we would guess that non-intervention might be construed as unhelpful by complainants, leading some of them to intervene independently. We consider unreported wolf-killings to be a more pronounced confounding variable after 2003, when state lethal management was allowed (Table 1), substantiated by a recent inference that allowing state killing of wolves seems to have potentially increased poaching of Michigan and Wisconsin wolves [34]. By definition, poaching can only confound tests of non-lethal deterrence because poaching following lethal intervention would only increase the number of wolves killed (undetectably in our context), but not change the nature of that lethal intervention. We do not see how poaching could confound the apparent reversal of effects of lethal control across our three geographic scales of analysis.

Furthermore, treatment bias would arise if methods of intervention were not standardized. Treatment bias certainly arose among non-lethal deterrents because different complainants received different types of non-lethal methods and we do not know if they maintained or installed the methods appropriately or identically. Non-lethal deterrents were presumably negotiated with complainants and therefore most prone to treatment bias that would confound our results. However, only 8% of our eventual sample received non-lethal deterrents. Moreover, we have no data on other deterrents or precautions unilaterally implemented by complainants. Lethal interventions were more uniform in method [20] but we did not receive precise, detailed information on implementation (number of trap-nights, exact locations, etc.). Moreover, if lethal interventions were spatially segregated from other types of interventions, then selection bias might have applied systematically because farms perceived to be higher-risk might have received lethal interventions preferentially and also be expected to have recurrent depredations. This might have resulted in significant, between-subject variability. Such a bias would not explain the spill-over effect we detected. Intermittent authority for lethal intervention led to the same spatial units receiving all types of intervention (**S1 Data File**). Given that authority for the state to kill wolves after verified depredations was granted or withheld by federal decisions unrelated to area attributes or recent depredation complaints and in several years of the study even high-risk areas received no interventions [16], it seems unlikely that lethal control authority for Michigan coincided with risky years. Therefore, any treatment bias (intervening lethally at sites that were inherently more likely to have recurrence of depredation) would have to occur at the spatiotemporal scale of individual farms within years. We addressed within-subject variability using a frailty model (S2 File), which revealed the presence of confounding effects at the section level, but the treatment effect remained statistically insignificant.

Finally, wolf abundance was unlikely to confound our tests because the number of wolves within our spatial units was unlikely to change substantially from one incident to the next within a small area within one year.

In sum, we find ample reason to expect confounding variables would weaken inference from a retrospective, quasi-experimental test of interventions to prevent livestock loss. Our attempts to detect and screen for biases were necessarily imperfect because we could not assign treatments randomly nor could we retrospectively assess if interventions were assigned haphazardly or subjectively. Our analyses controlled for variation in risk due to time and inter-farm differences using tvc and frailty models (S2 File), but could not ultimately control for transient changes in risk associated with wolves, people, or other wildlife. Moreover, we were not able to account for illegal wolf-killing that might have added to treatment bias affecting non-lethal interventions.

Nevertheless, there is value in the scientific examination of on-the-ground programs of predator management as they are actually carried out by the organizations that discharge them. Avoidance of selection, treatment or measurement biases would require enforcement of strict protocols that are rare worldwide [2, 39–41]. In addition to understanding how the strongest inference arises from gold-standard experiments without bias, wildlife managers have a responsibility to continually evaluate their particular actions and policies to ascertain if they are effective at accomplishing the goals set by the broadest society, and to remedy or terminate them if they are found to be ineffective, as evidence-based policy-making demands.

An example of a gold-standard design that might achieve strong inference would be random-assignment of treatment to different, large areas (e.g., 324 km^2) with uniform treatments, in which measurement is unbiased by blinding or independent, third party monitors, and data analysis is conducted by independent, third-party analysts without financial conflicts of interest involving the government or livestock industry. However, such an experiment would have to address the ethical implications for both animals and people of removing wild animals, possibly exposing more livestock to spill-over effects, and the broad public interest in preserving both wildlife and livelihoods. A step in that direction, albeit imperfect, may be to temporarily relocate predators to captivity until the analysis period ended in each area.

If our results are supported by a gold-standard experiment, we propose a hypothesis for two long-standing phenomena about human perceptions of conflicts with predators and the perceived effectiveness of interventions. We observe that killing predators is widely perceived to be effective (e.g., in our region: [42, 43], yet afterwards real and perceived risks appear to increase [44]. The spill-over effect may be responsible. Our hypothesis builds on the idea first articulated by Haber [45] that killing wolves can trigger pack disruption which might lead to more livestock predation than done by intact packs. If our inference about spill-over effects is confirmed, then we hypothesize that the perceived effectiveness of lethal methods stems from a few livestock owners who report preventive benefits, while neighboring livestock owners report increasing losses because of the spill-over effect from the former farms. The adverse effects of killing wolves as a response to depredations might thereby be obscured by anecdotal accounts and misperceptions.

Our results appear to contradict those of the [4] in the Northern Rocky Mountains (NRM) for the period 1989–2012. Although [4] conducted similar survival analyses, they found lethal methods significantly reduced the risk of recurrence, and that killing an entire wolf pack was more effective than the killing of a subset of members of a pack. They reported only a marginal difference between partial pack removal and no removal if wolves were killed within the first 7 days following a depredation event and no difference if 14 days elapsed. Most lethal interventions in Michigan were probably partial pack removals (median wolves killed = 1, **S1 Data File**) so our results are consistent. However, other differences in results between their study and ours could be due to different sites and methods.

The analysis in [4] included more varied methods of lethal intervention and the landscapes differ (theirs being mountainous and wider while Michigan's is flatter and surrounded by water on three sides, with attendant differences in vegetation, lake effects, human population density, wolf migration, livestock husbandry practices, etc.). In addition, the survival analyses employed by [4] differed from ours in ways that we could not resolve despite several email exchanges with the lead author and the analyst co-author.

First, [4] did not account for treatment effects beyond a single spatial scale (see Box 1). Their analysis was restricted to the affected wolf pack territory, despite their own reports that killing wolves had at times scattered surviving pack members beyond their original territory [10, 13, 46]. This previous research would argue for an analysis that examined neighboring areas potentially affected by spill-over from scattered survivors.

Second, apparent shortcomings of the statistical modeling in [4] may have affected its results. Their measure of delay to recurrence for full pack removals spans the time from death of the last pack member to the time when a new pack attacked livestock in the same territory. This measure of delay to next depredation artificially inflates effectiveness because it incorporates a potentially long timespan before a new pack establishes, which probably includes many time-consuming events unrelated to the intervention (e.g., immigration, breeding). By contrast, our method censored observations at the end of each year, so subjects were compared on a more-equal footing after intervention. For partial removal and no removal interventions in [4], the territory was still occupied by wolves so delays probably did not include as many time-consuming demographic events (if any). Although we understand that their intent was to analyze if depredations could be delayed for longer by killing entire wolf packs, we would argue that the appropriate control for the evaluation of this intervention would be sites with suitable wolf habitat but without an established pack because of events unrelated to killing wolves, such as recolonization of vacant habitat.

Using a biomedical analogy, [4] identified the hospital bed (the pack territory) as the subject rather than the patient (the wolf pack), regardless if the wolf pack is the same or if it dies and is replaced by a new pack. Researchers continued measuring the delay to the next infection (depredation) in that bed over time, without correcting for the delay to arrival of a new patient to that bed if a previous patient dies. The delay to the next infection once a patient dies is contingent on the arrival of a new patient to that empty bed, which has little to do with the intervention implemented to the bed other than making it available for a new patient (with full pack removal). By contrast, in our study the patient (area) is the only patient, each infection receives a treatment, and delay to next infection is always measured for the same patient with a reset each year.

Third, differences with [4] could also potentially arise from different handling of the proportional hazards (PH) assumption. We evaluated the compliance of our models with the PH assumption through the inclusion of a time-varying covariate (tvc) [31]. A significant tvc affects both our treatment hazard ratios and their significance, (e.g., Table 3). We assume that [4]'s team employed other model diagnostics to evaluate their compliance with the PH assumption, but they did not report such diagnostic tests. Until the summary data are published, we cannot agree with the conclusions in [4].

Finally, some might argue that by defining our subjects as area-years and including the same area over different years we pseudo-replicated non-independent samples. In our dataset, only 16 out of 106 sections had depredation incidents in multiple years. To address that concern, we built an alternative model in which areas were omitted in succeeding years (S5 File). Results for this dataset are consistent with our main results at the section scale (S5 File).

Conclusions

Lethal interventions by the State of Michigan against wolves in the vicinities of verified livestock losses did not appear to reduce future losses. We view our findings as preliminary pending experiments with stronger inference. Our inferences could not overcome a lack of systematic information on government interventions and no effort to control for their treatments, despite a call for such shortly after the legalization of lethal removal of wolves in 2003 [47]. We detected a potential spill-over of depredations from the farm receiving lethal intervention onto neighboring farms. Given this evidence for interactions in depredations over significant areas, we must look with skepticism upon any previous or future results which analyze the functional effectiveness of lethal control but do not take these spatial relationships into account. Further, given the severe ethical issues involved in implementing harmful or lethal interventions, the lack of effectiveness of these interventions argues for their curtailing in favor of non-lethal alternatives that are effective. In the State of Michigan, there is strong scientific evidence [2] for the effectiveness of at least two non-lethal methods (fladry and livestock guarding dogs; 7–8). No peer-reviewed scientific study has ever shown lethal methods to be effective in Michigan. Indeed, our review of [4] above suggests no study in the USA has yet proven with strong inference that killing wolves is effective in preventing future livestock losses [2, 39–41]. Although it may seem obvious that killing a predator whose jaws are about to lock on a calf should protect the calf, government lethal methods are not implemented in that way. Virtually all are indirect methods such as traps placed far from the depredation site and long after a calf is killed. Therefore, rigorous scientific evaluations are a necessary prerequisite before implementing an intervention, especially given the ethical and legal obligations to balance protection of livestock and wild animals for the broad public interest. The US Endangered Species Act mandates the use of the "best scientific and commercial data available" when making conservation and management decision for listed species.

Following recommendations for ethical wildlife management [48, 49], lethal management should be discontinued, as currently the harm it causes wolves and livestock is not offset by benefits. If lethal methods are still necessary in some situations [48, 49], these should be constantly monitored and evaluated by independent third parties to measure their effectiveness or lack thereof [48].

Supporting information

S1 File. Distribution of observations and recurrent events between treatments and strata for all spatial scales.
(DOCX)

S2 File. Results from frailty mfodels for main dataset, for all spatial scales.
(DOCX)

S3 File. Results for 'traps placed' dataset.
(DOCX)

S4 File. Results for post-2003 dataset.
(DOCX)

S5 File. Results for 'skip-a-year' dataset and outlier exclusion.
(DOCX)

S1 Data File. Livestock depredation events involving gray wolves in the state of Michigan, USA (1998–2014).
(XLSX)

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S1 File: Distribution of observations and recurrent events between treatments and strata for all spatial scales

<u>Section scale</u>: At the section scale, we restricted analyses to 3 strata due to lack of depredation events (n<10) in subsequent strata (Table A).

Table A. Number of depredation events per intervention type, by strata (S#)

	Number of observations (n)											
Intervention / Stratum	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12
Lethal	23	3	5	1								
Non-lethal	128	28	12	7	4	3	3	3	3	3	3	1

Table B. Number of censored (0) and recurrent (1) depredation events for non-lethal treatment, by strata (S#)

	Number of observations (n)											
Rec event / Stratum	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12
0	100	12	8	3	1						2	1
1	28	16	4	4	3	3	3	3	3	3	1	

Table C. Number of censored (0) and recurrent (1) depredation events for lethal treatment, by strata (S#)

	Number of observations (n)			
Rec event / Stratum	S1	S2	S3	S4
0	20	2	1	1
1	3	1	4	

<u>Township scale</u>: At the township scale, we restricted analyses to one strata due to lack of depredation events for both treatments in stratum 2 (Tables D-F).

Table D. Number of depredation events per type of intervention type, by strata (S#)

	Number of observations (n)		
Intervention / Stratum	S1	S2	S3
Lethal	26	2	
Non-lethal	99	22	3

Table E. Number of censored (0) and recurrent (1) depredation events for non-lethal treatment, by strata (S#)

	Number of observations (n)		
Rec event / Stratum	S1	S2	S3
0	78	19	3
1	21	3	

Table F. Number of censored (0) and recurrent (1) depredation events for lethal treatment, by strata (S#)

	Number of observations (n)	
Rec event / Stratum	S1	S2
0	23	2
1	3	

<u>Neighborhood scale</u>: At the neighborhood scale, we restricted analyses to 2 strata due to lack of depredation events (n<10) in subsequent strata (Table G).

Table G. Number of depredation events per type of intervention type, by strata (S#)

	Number of observations (n)			
Intervention / Stratum	S1	S2	S3	S4
Lethal	20	6	1	
Non-lethal	86	13	5	1

Table H. Number of censored (0) and recurrent (1) depredation events for non-lethal treatment, by strata (S#)

	Number of observations (n)			
Rec event / Stratum	S1	S2	S3	S4
0	72	8	4	1
1	15	5	1	

Table I. Number of censored (0) and recurrent (1) depredation events for lethal treatment, by strata (S#)

	Number of observations (n)		
Rec event / Stratum	S1	S2	S3
0	15	5	1
1	4	1	

S2 File: Results from frailty models, for all spatial scales

Frailty models evaluate the goodness of fit of the treatment variable including within-subject random effects that might confound the effect of treatment [35]. In our context, frailty models assume that there are high-risk and low-risk subjects due to factors extrinsic to treatments or individual depredation events [17, 36]. Frailty models are useful when survival time is influenced by unmeasured factors [31, 37]. Important differences between subjects can confound the apparent effect of treatment [35]. Our frailty model results supported our main models, showing an insignificant reduction in risk of recurrence following lethal intervention, at all scales. Only the section scale frailty model revealed high heterogeneity due to within-subject effects (Table A). The section scale of our models is precisely the level at which one would expect the most frailty (within-subject factors would be felt at the spatial scale closest to an individual farm). For example, two farms with unusually high numbers of depredations (see S5 File) might have been detected by our frailty model. Accounting for this heterogeneity thus increased the magnitude of the coefficient for the effect of the intervention (HR = 52%) relative to our main model, although the coefficient for lethal intervention remains statistically insignificant.

Table A. Results from frailty models measuring risk of recurrence between treatments (lethal and non-lethal) implemented after depredation events, for all spatial scale

	Spatial scale of analysis		
	Section	**Township**	**Neighborhood**
Frailty models			
Intervention COEF (SD)	-0.74 (0.522)	-0.73 (0.620)	-0.22 (0.510)
p-val	0.158	0.242	0.67
frailty COEF (SD)	4.65 (1.68)**	0.05	0.03
p-val	0.006	-	-

*Significance: * if p-val <.05; ** if <.01.*

S3 File: Results for 'traps placed' dataset

In the alternate 'traps placed' model, non-lethal management with zero wolves killed was reclassified as 'lethal' because the infrastructure of killing wolves and attendant human influences on the habitat were treated as similar to lethal intervention in which wolves died.

Log rank tests could not distinguish the survival functions between treatments at any spatial scale (all tests P>0.05, Table A). All Cox models also echo our main results, suggesting a statistically insignificant effect of lethal intervention relative to no intervention (Table B). Our most robust models at each scale suggest that lethal intervention was associated with a statistically insignificant reduction in risk of recurrence compared to no intervention at the section scale (treatment HR=0.76, P=0.435); a non-significant increase in risk of recurrence at the township scale (although the hazard ratio increases), signaling a greater risk of recurrence (treatment HR=2.35, P=0.247; tvc HR=0.96, P=0.006); and a non-significant reduction in risk of recurrence at the neighborhood of townships scale (treatment HR=0.64, P=0.327). The risk of recurrence also seemed to increase with calendar-year at all spatial scales (Table B), but this effect was not statistically significant at the township scale (P=0.153). Also, consistent with our main results, we found no evidence of a correlation between delay to recurrence and the number of wolves killed at any spatial scale for those depredation events followed by lethal intervention (Spearman's rho P>.05; Table C).

Table A. General and stratified log-rank (χ^2) tests examining difference between treatments' (lethal and non-lethal) survival distributions (measuring risk of recurrence) after wolf depredations, for all spatial scales, for the 'traps placed' dataset.

	Spatial scale of analysis		
	Section	**Township**	**Neighborhood**
INCIDENTS AND 'FAILURES'			
TOTAL DEPREDATION EVENTS	199	125	125
Failures (recurrent events)	56	24	25
SURVIVAL FUNCTIONS			
Log rank test (χ2)	0.44	0.86	0.40
p-val	0.5072	0.3534	0.5296
Stratified Log-rank test (χ2)	0.46	-	0.68
p-val	0.4989	-	0.4094

*Significance: * if p-val <.05; ** if <.01.*

Table B. Main results of Cox models measuring risk of recurrence between treatments (lethal and non-lethal) implemented after wolf depredations, for all spatial scales, for the 'traps placed' dataset.

	Spatial scale of analysis					
	Section		**Township**		**Neighborhood**	
PROPORTIONAL HAZARD MODELS	*Interv*	*Interv & year*	*Interv*	*Interv & year*	*Interv*	*Interv & year*
Standard cox (stratified)						
Intervention HR (SD)	0.78 (0.26)	0.76 (0.27)	0.60 (0.35)	0.58 (0.33)	0.69 (0.32)	0.64 (0.29)
p-val	0.453	0.435	0.38	0.34	0.430	0.327
year HR (SD)	-	1.09 (0.04)*	-	1.05 (0.05)	-	1.14 (0.06)*
p-val	-	0.022	-	0.292	-	0.022
Standard cox with tvc (stratified)						
Intervention HR (SD)	0.55 (0.25)	0.54 (0.26)	3.46 (2.82)	3.31 (2.70)	0.78 (0.57)	0.75 (0.58)
p-val	0.19	0.191	0.128	0.141	0.737	0.711
tvc(Intervention)	1.01 (0.01)	1.01 (0.01)	0.95 (0.02)**	0.95 (0.02)**	1.00 (0.01)	1.00 (0.01)
p-val	0.129	0.15	0.006	0.006	0.821	0.769
year HR (SD)	-	1.09 (0.04)*	-	1.05 (0.05)	-	1.14 (0.06)*
p-val	-	0.023	-	0.291	-	0.022

*Significance: * if p-val <.05; ** if <.01.*

Table C. Spearman correlation between delay to recurrence and number of wolves killed after depredation events followed by lethal intervention (wolves killed $\geq$ 0), for all spatial scales, for the 'traps placed' dataset.

	Section	**Township**	**Neighborhood**
Spearman's rho	0.076	0.233	0.161
p-val	0.6571	0.224	0.3954

*Significance: * if p-val <.05; ** if <.01.*

S4 File: Results for post-2003 dataset

Below we present results for our alternate post 2003 dataset. In this dataset, we limited depredation events to those occurring in or after 2003, after lethal management was episodically permitted.

Log rank tests could not distinguish the survival functions between treatments at any spatial scale (all tests P>0.05, Table A). All Cox models are also consistent with our main results, suggesting a statistically insignificant effect of lethal intervention relative to no intervention (Table B). Our most robust models at each spatial scale suggest that lethal intervention was associated with a statistically insignificant reduction in risk of recurrence compared to no intervention at the section scale (treatment HR=0.67, P=0.187); a statistically insignificant increase in risk of recurrence at the township scale (treatment HR=1.21, P=0.795; tvc HR=0.97, P=0.001, suggesting minimal reduction in risk over follow-up time); and a statistically insignificant reduction in risk of recurrence at the neighborhood of townships scale (treatment HR=0.68, P=0.413). The risk of recurrence also seemed to increase with calendar-year at all spatial scales (Table B), but this effect was not statistically significant (P>0.05) at any scale in this dataset. Also, consistent with our main results, we found no evidence of a correlation between delay to recurrence and the number of wolves killed at any spatial scale for those depredation events followed by lethal intervention (Spearman's rho P>0.05; Table C).

Table A. General and stratified log-rank (χ^2) tests examining difference between treatments' (lethal and non-lethal) survival distributions (measuring risk of recurrence) after wolf depredations, for all spatial scales, for the post 2003 dataset.

	Spatial scale of analysis		
	Section	**Township**	**Neighborhood**
INCIDENTS AND 'FAILURES'			
TOTAL DEPREDATION EVENTS	174	103	103
Failures (recurrent events)	56	21	24
SURVIVAL FUNCTIONS			
Log rank test (χ2)	1.11	1.94	0.57
p-val	0.2927	0.1632	0.4495
Stratified Log-rank test (χ2)	1.15	-	0.88
p-val	0.2845	-	0.347

*Significance: * if p-val <.05; ** if <.01.*

Table B. Main results of Cox models measuring risk of recurrence between treatments (lethal and non-lethal) implemented after wolf depredations, for all spatial scales, for the post 2003 dataset.

	Spatial scale of analysis					
	Section		Township		Neighborhood	
PROPORTIONAL HAZARD MODELS	*Interv*	*Interv & year*	*Interv*	*Interv & year*	*Interv*	*Interv & year*
Standard cox (stratified)						
Intervention HR (SD)	0.67 (0.20)	0.67 (0.20)	0.43 (0.28)	0.45 (0.29)	0.66 (0.31)	0.68 (0.32)
p-val	0.169	0.187	0.188	0.215	0.376	0.413
year HR (SD)	-	1.03 (0.05)	-	1.04 (0.08)	-	1.09 (0.09)
p-val	-	0.529	-	0.611	-	0.284
Standard cox with tvc (stratified)						
Intervention HR (SD)	0.42 (0.19)	0.42 (0.19)	1.41 (1.09)	1.44 (1.41)	0.71 (0.51)	0.74 (0.55)
p-val	0.054	0.059	0.661	0.643	0.632	0.690
tvc(Intervention)	1.01 (0.01)	1.01 (0.01)	0.97 (0.01)**	0.97 (0.01)**	1.00 (0.01)	1.00 (0.01)
p-val	0.071	0.072	0.001	0.001	0.886	0.873
year HR (SD)	-	1.03 (0.05)	-	1.04 (0.08)	-	1.09 (0.09)
p-val	-	0.533	-	0.623	-	0.282

*Significance: * if p-val <.05; ** if <.01.*

Table C. Spearman correlation between delay to recurrence and number of wolves killed after depredation events followed by lethal intervention (wolves killed $\geq$ 0), for all spatial scales, for the post 2003 dataset.

	Section	Township	Neighborhood
Spearman's rho	0.107	0.212	0.295
p-val	0.5591	0.2994	0.1354

*Significance: * if p-val <.05; ** if <.01.*

S5 File: Results for 'skip-a-year' dataset and outlier exclusion

Below we present results for our alternate 'skip-a-year' model for the section scale, created to address potential pseudo-replication concerns. In this model, sections are left outside the study for one year if they experienced depredations during the previous year. A one-year period outside the study seems a sensible amount of time for the section to enter the study as a different subject (section-year combination) independent from its prior inclusion..

Consistent with our main and supplementary analyses, log rank tests could not distinguish the survival functions between treatments at the section scale (df=1, general test: χ^2=0.83, P=0.363; stratified test: χ^2=1.27, P=0.488; Table A). Results for our Cox models are also consistent with our main results, suggesting a statistically insignificant effect of lethal intervention relative to non-lethal intervention (Table B). The models including tvc indicate that violating the proportional hazards assumption is not a concern (tvc P>0.05). Our main model, including treatment and year, shows that lethal intervention was associated with a statistically insignificant 45% reduction in risk of recurrence compared to non-lethal intervention at the section scale (treatment HR=0.52, P=0.145). The risk of recurrence also seemed to increase with calendar-year (HR=1.07; P=0.104), but this effect was not statistically significant. Also consistent with our main results, we found no evidence of a correlation between delay to recurrence and the number of wolves killed for those depredation events followed by lethal intervention (Spearman's rho P>0.05; Table C).

Additionally, we ran our tests on the dataset excluding one outlier farm which had an atypically high number of depredations (see http://www.mlive.com/news/index.ssf/2013/11/john_koski_part_1_tour_the_far.html). Although this restricted the analysis to one stratum at each spatial scale, the results for both section and township echo our main results. Furthermore, the result for increased risk for the township scale became stronger (p=0.093) in this model.

Table A. General and stratified log-rank (χ^2) tests examining difference between treatments' survival distributions at the section level for the 'skip-a-year' dataset.

	Section
SUBJECTS AND 'FAILURES'	
TOTAL DEPREDATION EVENTS	166
Failures (recurrent events)	38
SURVIVAL FUNCTIONS	
Log rank test (χ2)	0.83
p-val	0.3633
Stratified Log-rank test (χ2)	1.27
p-val	0.2602

*Significance: * if p-val <.05; ** if <.01.*

Table B. Results of Cox models at the section level for the 'skip-a-year' dataset.

	Section	
PROPORTIONAL HAZARD MODELS	*Interv*	*Interv & year*
Standard cox (stratified)		
Intervention HR (SD)	0.55 (0.24)	0.52 (0.234)
p-val	0.166	0.145
year HR (SD)	-	1.07 (0.043)
p-val	-	0.104
Standard cox with tvc (stratified)		
Intervention HR (SD)	0.51 (0.24)	0.48 (0.229)
p-val	0.149	0.125
tvc(Intervention) HR (SD)	1.00 (0.009)	1.00 (0.009)
p-val	0.771	0.826
year HR (SD)	-	1.07 (0.043)
p-val	-	0.105

*Significance: * if p-val <.05; ** if <.01.*

Table C. Spearman correlation between delay to recurrence and number of wolves killed for depredation events followed by lethal intervention (wolves killed $\geq$ 0) at the section level for the 'skip-a-year' dataset.

	Section
Spearman's rho	0.193
p-val	0.3547

*Significance: * if p-val <.05; ** if <.01.*

Supplemental Data 1: Livestock depredation events involving gray wolves in the State of Michigan, USA

INC_UID	INC_UID Date	TRS	Intervention(trichotomy)	# wolves killed
1	21 05 1998	**N21W26	None	
2	26 05 1998	**N23W07	None	
3	28 08 1998	**N21W03	None	
4	22 04 1999	**N30W13	None	
5	1 05 1999	**N30W28	None	
6	24 05 1999	**N21W02	None	
7	28 09 1999	**N30W08	None	
8	17 04 2000	**N33W13	None	
9	13 06 2000	**N46W32	None	
10	7 07 2000	**N43W17	None	
11	21 08 2000	**N21W02	None	
12	22 09 2000	**N21W08	None	
13	16 09 2001	**N39W08	None	
14	17 09 2001	**N47W05	None	
15	27 09 2001	**N34W23	None	
16	17 01 2002	**N26W19	None	
17	23 02 2002	**N35W32	None	
18	23 03 2002	**N03W11	None	
19	3 06 2002	**N42W10	None	
20	20 06 2002	**N41W15	None	
21	15 08 2002	**N28W27	None	
22	26 08 2002	**N10W03	None	
23	19 09 2002	**N34W29	None	
24	1 10 2002	**N42W21	None	
25	14 10 2002	**N10W09	None	
26	25 02 2003	**N26W10	None	
27	31 05 2003	**N10W21	None	
28	9 07 2003	**N28W14	None	
29	17 07 2003	**N35W13	None	
30	30 07 2003	**N10W10	Lethal	2
31	4 08 2003	**N10W21	Non-lethal	
32	20 08 2003	**N10W18	None	
33	25 08 2003	**N10W21	None	
34	29 08 2003	**N10W31	None	
35	30 08 2003	**N10W18	None	
36	2 09 2003	**N10W18	None	
37	13 09 2003	**N33W16	None	
38	15 09 2003	**N10W18	None	
39	30 09 2003	**N03W08	None	
40	14 06 2004	**N21W10	Lethal	1
41	22 06 2004	**N23W20	Non-lethal	
42	7 07 2004	**N21W34	None	
43	27 07 2004	**N21W34	None	
44	29 07 2004	**N41W19	Lethal	1
45	6 08 2004	**N13W09	Non-lethal	
46	14 08 2004	**N40W14	Lethal	1
47	27 08 2004	**N35W34	Lethal	1

48	4 09 2004	**N03W01	None	
49	15 10 2004	**N03W01	None	
50	21 10 2004	**N01W13	None	
51	12 04 2005	**N34W21	Lethal	3
52	1 06 2005	**N18W07	None	
53	24 07 2005	**N01W35	Lethal	0
54	27 07 2005	**N01W34	Lethal	0
55	29 07 2005	**N13W07	Lethal	1
56	3 10 2005	**N23W07	None	
57	23 10 2005	**N20W14	None	
58	2 11 2005	**N20W14	None	
59	21 02 2006	**N20W10	Non-lethal	
60	27 04 2006	**N41W15	None	
61	10 05 2006	**N41W15	None	
62	21 05 2006	**N34W21	None	
63	25 05 2006	**N41W15	Lethal	1
65	29 05 2006	**N39W11	Lethal	1
66	3 06 2006	**N41W15	Lethal	4
68	28 07 2006	**N34W12	Lethal	1
69	10 08 2006	**N34W08	None	
70	29 03 2007	**N10W09	None	
71	8 05 2007	**N41W15	Lethal	7
72	26 05 2007	**N38W17	Lethal	2
73	17 06 2007	**N01W32	None	
74	3 07 2007	**N27W25	Lethal	0
75	6 07 2007	**N41W15	None	
76	10 07 2007	**N41W15	Lethal	2
77	17 07 2007	**N10W09	None	
78	12 08 2007	**N34W10	Lethal	2
79	17 08 2007	**N34W20	Lethal	1
80	17 08 2007	**N34W12	None	
81	20 08 2007	**N34W04	None	
82	28 08 2007	**N10W31	None	
83	11 10 2007	**N01E24	None	
84	22 10 2007	**N01W03	None	
85	13 02 2008	**N41W15	Lethal	0
86	25 02 2008	**N41W15	Lethal	4
87	28 03 2008	**N39W09	None	
88	11 05 2008	**N34W03	None	
89	21 06 2008	**N35W24	None	
90	26 06 2008	**N41W15	Lethal	1
91	1 07 2008	**N41W15	None	
92	16 07 2008	**N34W29	Lethal	1
93	21 07 2008	**N27W25	None	
94	30 07 2008	**N41W15	None	
95	4 08 2008	**N10W10	None	
96	19 08 2008	**N34W15	None	
97	1 09 2008	**N39W11	Lethal	2
98	14 09 2008	**N35W24	None	

99	8 03 2009	**N41W15	Non-lethal	
100	27 03 2009	**N41W15	Non-lethal	
101	2 04 2009	**N27W35	Non-lethal	
102	17 04 2009	**N46W14	Non-lethal	
103	1 05 2009	**N41W15	Lethal	1
104	30 07 2009	**N34W22	Lethal	0
105	11 08 2009	**N10W21	Non-lethal	
106	26 08 2009	**N41W15	None	
107	26 08 2009	**N10W31	Non-lethal	
108	13 09 2009	**N10W31	Non-lethal	
109	7 10 2009	**N10W31	Non-lethal	
110	31 10 2009	**N01W08	None	
111	14 11 2009	**N10W09	Non-lethal	
112	23 04 2010	**N41W15	None	
116	27 04 2010	**N43W12	None	
119	3 05 2010	**N41W15	None	
122	6 05 2010	**N10W21	Non-lethal	
124	9 05 2010	**N25W02	None	
125	11 05 2010	**N41W15	None	
126	12 05 2010	**N38W18	None	
127	12 05 2010	**N41W15	None	
128	17 05 2010	**N41W15	None	
129	20 05 2010	**N11W34	Non-lethal	
130	20 05 2010	**N41W15	None	
132	24 05 2010	**N41W15	None	
137	2 06 2010	**N11W34	Non-lethal	
139	6 06 2010	**N41W15	None	
140	7 07 2010	**N10W09	None	
142	23 07 2010	**N38W18	None	
143	1 08 2010	**N41W15	None	
145	8 08 2010	**N47W34	None	
146	10 08 2010	**N41W15	None	
148	18 08 2010	**N34W12	None	
149	19 08 2010	**N38W18	None	
150	20 08 2010	**N10W03	Non-lethal	
151	26 08 2010	**N25W01	None	
152	2 09 2010	**N34W29	None	
153	2 09 2010	**N34W20	None	
154	4 09 2010	**N34W29	None	
155	5 09 2010	**N10W31	None	
156	6 09 2010	**N11W34	Non-lethal	
157	10 09 2010	**N41W15	None	
158	24 11 2010	**N32W01	None	
159	28 11 2010	**N10W31	None	
160	21 03 2011	**N41W15	None	
161	25 03 2011	**N41W15	None	
162	29 03 2011	**N41W15	None	
163	4 04 2011	**N41W15	None	
164	8 04 2011	**N41W15	None	

165	13 04 2011	**N33W33	None	
166	15 04 2011	**N41W15	None	
167	20 04 2011	**N41W15	None	
170	25 04 2011	**N41W15	None	
171	25 04 2011	**N38W18	None	
173	27 04 2011	**N46W14	None	
178	5 05 2011	**N41W15	None	
179	6 05 2011	**N34W15	None	
180	8 05 2011	**N38W17	None	
182	17 05 2011	**N11W34	None	
183	25 05 2011	**N38W17	None	
184	20 06 2011	**N11W34	None	
185	5 07 2011	**N34W20	None	
186	15 07 2011	**N11W34	None	
187	22 07 2011	**N10W03	None	
188	5 08 2011	**N34W12	None	
189	6 08 2011	**N10W31	None	
190	8 08 2011	**N35W24	None	
191	19 08 2011	**N41W15	None	
192	21 08 2011	**N35W26	None	
193	22 08 2011	**N10W31	None	
194	25 08 2011	**N23W04	None	
195	25 08 2011	**N41W15	None	
196	2 09 2011	**N34W12	None	
197	9 10 2011	**N10W31	None	
198	8 11 2011	**N01W02	None	
199	9 04 2012	**N38W17	Lethal	4
200	19 04 2012	**N47W29	Lethal	1
201	23 04 2012	**N35W29	Lethal	1
202	25 04 2012	**N45W07	None	
203	1 05 2012	**N41W15	None	
205	7 05 2012	**N41W15	None	
206	8 05 2012	**N34W15	None	
206.5	28 04 2012	**N34W15	Lethal	1
207	11 05 2012	**N34W29	None	
208	14 05 2012	**N38W27	None	
209	14 05 2012	**N38W17	None	
210	16 05 2012	**N41W15	Lethal	1
211	17 05 2012	**N35W21	None	
212	19 05 2012	**N38W17	None	
213	22 05 2012	**N34W12	None	
214	24 05 2012	**N41W28	Lethal	1
215	25 05 2012	**N35W24	None	
216	25 05 2012	**N38W17	None	
217	25 05 2012	**N41W15	None	
218	2 06 2012	**N34W18	None	
219	2 06 2012	**N41W15	None	
220	25 06 2012	**N41W15	None	
221	14 07 2012	**N41W15	None	

222	14 07 2012	**N10W33	None	
224	18 07 2012	**N41W15	None	
225	22 07 2012	**N26W26	None	
226	23 07 2012	**N10W33	None	
227	1 08 2012	**N41W15	None	
228	7 08 2012	**N10W33	None	
230	27 08 2012	**N41W15	None	
231	4 09 2012	**N41W15	None	
232	27 09 2012	**N41W15	None	
233	8 10 2012	**N11W25	None	
234	14 11 2012	**N19W32	None	
235	13 04 2013	**N01W10	None	
236	29 04 2013	**N35W11	Lethal	2
237	2 05 2013	**N10W15	None	
238	14 05 2013	**N34W18	None	
239	23 05 2013	**N34W18	Lethal	2
240	7 07 2013	**N03W02	None	
241	25 07 2013	**N38W17	None	
242	7 08 2013	**N04W33	None	
243	9 08 2013	**N41W28	Lethal	1
244	16 08 2013	**N34W29	Lethal	1
245	28 08 2013	**N35W24	None	
246	9 09 2013	**N38W17	None	
247	17 09 2013	**N36W03	None	
248	10 10 2013	**N10W27	None	
249	13 10 2013	**N11W25	None	
250	16 10 2013	**N10W27	Lethal	1
251	16 10 2013	**N04W15	None	
252	31 10 2013	**N02W20	None	
253	22 03 2014	**N10W10	None	
254	27 03 2014	**N38W17	None	
255	1 04 2014	**N10W10	None	
256	27 04 2014	**N38W17	None	
258	16 05 2014	**N33W30	None	
259	23 05 2014	**N38W17	None	
260	27 05 2014	**N10W15	None	