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Economic assessment of wild bird mortality induced by the use of lead gunshot in European wetlands

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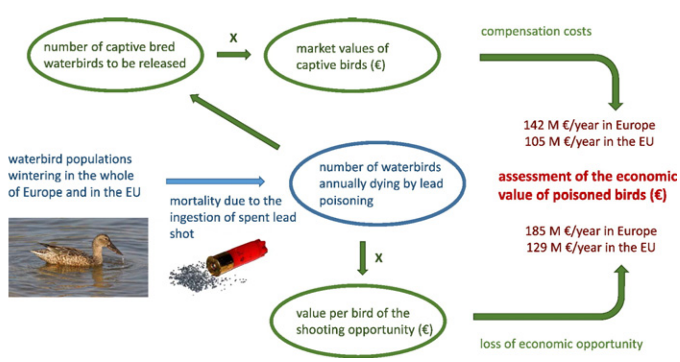
HIGHLIGHTS

- Waterbirds mortality due to lead shot ingestion is a relevant political issue.
- New methods to assess economic cost of lead poisoning on waterbirds are proposed.
- Cost estimates convert biological data into relevant information for policy.
- Restocking with captive birds in Europe would cost 105–142 million euros per year.
- Lost shooting opportunities imply an annual GVA reduction of 129–185 million euros.

GRAPHICAL ABSTRACT



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ABSTRACT

In European wetlands, at least 40 bird species are exposed to the risk of lead poisoning caused by ingestion of spent lead gunshot. Adopting a methodology developed in North America, we estimated that about 700,000 individuals of 16 waterbird species die annually in the European Union (EU) (6.1% of the wintering population) and one million in whole Europe (7.0%) due to acute effects of lead poisoning. Furthermore, threefold more birds suffer sub-lethal effects. We assessed the economic loss due to this lead-induced mortality of these 16 species by calculating the costs of replacing lethally poisoned wild birds by releasing captive-bred ones. We assessed the cost of buying captive-bred waterbirds for release from market surveys and calculated how many captive-bred birds would have to be released to compensate for the loss, taking into account the high mortality rate of captive birds (72.7%) in the months following release into the wild. Following this approach, the annual cost of waterbird mortality induced by lead shot ingestion is estimated at 105 million euros per year in the EU countries and 142 million euros in the whole of Europe. An alternative method, based upon lost opportunities for hunting caused by deaths due to lead poisoning, gave similar results of 129 million euros per year in the EU countries and 185 million euros per year in the whole of Europe. For several reasons these figures should be regarded as conservative. Inclusion of deaths of species for which there were insufficient data and delayed deaths caused indirectly by lead poisoning and effects on reproduction would probably increase the estimated losses substantially. Nevertheless, our results suggest that the benefits of a restriction on the use of lead gunshot over wetlands could exceed the cost of adapting to non-lead ammunition.

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1. Introduction

It has been known since the end of the 18th century that ingestion of lead pellets causes deaths of birds both in terrestrial and aquatic habitats (Calvert, 1876; Grinnell, 1894). In wetlands, ducks, geese, swans, waders, rails, flamingos and other waterbirds ingest spent gunshot from the soil surface and from mud. Differences among waterbird species in the proportion of sampled birds with ingested lead pellets in the alimentary tract tends to be consistent, despite large differences for a given species among regions and countries (Green and Pain, 2016; Mateo, 2009). Species which take in large-diameter grit to grind up their food in the muscular gizzard and which feed on large seeds tend to ingest lead pellets frequently, whereas species which ingest small-diameter grit and feed on leaves rarely ingest gunshot (Bellrose, 1959; Mateo, 2009; Mateo et al., 2000; Thomas et al., 1977). These observations support the idea that waterbirds ingest gunshot pellets because they mistake them for grit or food items. Both grit and lead pellets are usually retained until they are totally milled (Del Bono and Braca, 1973). Infrequently, some pellets pass through the alimentary tract and are eliminated in the faeces, but their mass is greatly reduced by then, owing to the combined effects of mechanical abrasion and gastric acid (Plouzeau et al., 2011). Substantial amounts of lead derived from ingested gunshot are absorbed by the digestive system of the bird and enters their bloodstream (Rodríguez et al., 2010). Given the high toxicity of this metal (De Francisco et al., 2003), the ingestion of just one pellet can be enough to cause the death of a small or medium-sized duck by primary poisoning (Guillemain et al., 2007; Mautino and Bell, 1986; Olney, 1960).

In wetlands open to hunting, the density of lead pellets lying in superficial sediments may reach very high densities, up to hundreds per m^2 (Bianchi et al., 2011; Mateo, 2009). In Europe, the highest pellet densities have been recorded in north-western countries and in the Mediterranean region, where most of western Palearctic Anatidae (ducks, geese and swans) congregate to overwinter (Scott and Rose, 1996). Therefore, waterbird populations are exposed to a substantial risk of lead pellet ingestion. The proportion of wildfowl found in Europe with ingested gunshot is normally high, both in hunter-shot birds and in birds dead from other causes (Green and Pain, 2016; Mateo, 2009; Pain et al., 2015). According to a conservative estimate, based upon the prevalence of pellet ingestion in 17 waterfowl species wintering in Europe, around one million Anatidae die every year as a consequence of lead poisoning, which corresponds to 8.7% of the wintering population (Mateo, 2009).

Raptors living in wetlands are also exposed to the risk of secondary poisoning with lead when they depredate or scavenge lead-contaminated animals. The intoxication may occur when a raptor eats a waterbird with lead pellets in the digestive tract, with elevated lead levels in its tissues or with embedded shot-in pellets, including un-retrieved quarry that has been wounded or killed by hunters (Helander et al., 2009; Mateo, 2009; Mateo et al., 1999; Pain, 1991; Pain et al., 1993, 1997; Wayland and Bollinger, 1999). Such events are likely to occur frequently, given the high prevalence of waterbirds with ingested and/or embedded shot pellets revealed by several studies (Falk et al., 2006; Guillemain et al., 2007; Tavecchia et al., 2001).

Because of the high prevalence of lead poisoning in waterbirds, the issue is addressed in several Multilateral Environmental Agreements (Stroud, 2015). These include the UNEP-CMS Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA), which was approved in 1995 (Beintema, 2001). Furthermore, in recent decades many countries have been adopting partial or total bans on the use of lead ammunition to avoid or reduce the accumulation of spent lead gunshot in wetlands (AEWA Secretariat, 2008). In this framework, the European Union (EU), as a signatory party of the AEWA Agreement, in 2015, started a process to assess whether a generalized ban could be introduced under the Regulation for the Registration, Evaluation, Authorization and Restriction of Chemical Substances (REACH), adopted to improve the protection of human health and environment from the risks posed by chemicals. For this purpose, the European Commission, in accordance with Article 69 (1) of the REACH Regulation, has requested the European Chemicals Agency (ECHA) to assess the possible risks posed by lead gunshot to human health and the environment, particularly to aquatic bird species, and the need for EU-wide action. ECHA has prepared an Annex XV dossier (ECHA, 2017). Restriction proposals need to contain a description of the risks as well as information on the health and environmental benefits, the associated costs and other socio-economic impacts.

In this paper we aim to provide an estimate of the economic value of the waterbirds which are lost annually because of poisoning by spent lead gunshot. We do this by i) proposing two new approaches to quantify monetary damages caused by injuries to waterbirds and ii) applying these approaches to evaluate the economic value of waterbirds poisoned by lead pellets in the 28 EU Member States and in the whole of Europe. In the last few decades, several methods have been developed to quantify monetary damages for injuries caused to wildlife, habitat, and the services they provide (Ando and Khanna, 2004; Hampton and Zafonte, 2003). A practical way to assess Natural Resource Damage (NRD) is to evaluate the cost of remediation and/or restoration interventions (Burger, 2008; Cole, 2010). When NRD has a relevant impact on birds, three different procedures can be followed to recover the affected populations: 1) implementation of habitat restoration projects with potential ecological benefits for birds (Norton and Thomas, 1994; Zafonte and Hampton, 2005); 2) reduction of mortality deriving from other causes that can be prevented more easily (Cole and Dahl, 2013); 3) restocking/reintroduction programmes to replace birds that die because of human-related causes. The first two procedures have been applied especially at local levels where compensatory actions can be effective, while the last method is widely adopted by hunters in many European countries to counteract the effects of overhunting and enhance their hunting opportunities (Champagnon, 2011; Söderquist, 2015), or as part of conservation projects (Pacheco and McGregor, 2004; Tavecchia et al., 2009). We used the third of these methods and estimated the costs involved in replacing the loss of waterbirds poisoned by lead shot used in aquatic habitats, through the release of captive-bred birds. In the case of waterbirds, restocking costs can be estimated because most species are reared in captivity and sold either as ornamental birds or hunting decoys. Furthermore, in Europe three million hand-reared mallards (*Anas platyrhynchos*) are estimated to be released annually to enhance hunting opportunities (Champagnon, 2011; Champagnon et al., 2016; Söderquist, 2015). These circumstances offer the opportunity to assess the value of each bird and also to evaluate the effectiveness of restocking programmes. A further consideration is that studies carried out on mortality rates revealed that released captive-bred waterbirds have a life expectancy considerably lower than wild individuals (Schladweiler and Tester, 1972; Söderquist et al., 2013; Tavecchia et al., 2009). The main reasons for their low survival are: 1) inadequate development of the digestive system in juveniles fed with artificial food and their consequent inability to adapt to natural food; 2) inexperience of captive birds not used to search for food in natural habitats; 3) inadequate behavioural responses to predators (Champagnon, 2011; Champagnon et al., 2012). To counterbalance this additional post-release mortality, restocking programmes should foresee the release of a number of birds largely exceeding the losses that they are intended to compensate. This implies extra costs to be evaluated in NRD assessments because a high proportion of released waterbirds are expected to die before their use value is realised during the hunting season.

An alternative method for NRD evaluation is to estimate the opportunity cost of waterbird hunting foregone by hunters because of the deaths of lead-poisoned birds. We evaluate the per capita Gross Value Added GVA) of hunted ducks and geese in the UK and use this, together with estimates of annual waterbird deaths caused by lead poisoning, to evaluate economic loss at the European and EU levels.

2. Materials and methods

We used two methods to estimate NRD caused by the use of lead gunshot in the European wetlands. First, we calculated how much it costs to replace lead-poisoned wild birds with captive-bred ones and second we estimated the economic value of the shooting opportunities foregone by hunters because of the deaths of the lead-poisoned birds. The first method estimates the cost of providing compensation as an estimate of cost of damage and the second estimates the opportunity cost of the damage itself.

Both methods require the acquisition of data on the number of individuals for each species that die annually because of the ingestion of lead pellets. The first method also requires information on the mortality rate of captive birds in the months following the release into the wild, so as to calculate how many captive-bred birds have to be released to compensate the loss of wild birds, and the market values of captive birds to be released. The second method requires an estimate of the value per bird of the shooting opportunity foregone because of the death of a bird by lead poisoning.

2.1. Species and number of birds annually poisoned by lead gunshot

An estimate of the number of birds poisoned annually by lead gunshot can be inferred for each species from the estimated European population size and the additional per capita annual mortality rate due to lead shot ingestion. The additional mortality rate was estimated from information on the prevalence of ingested lead shot using the approach of Bellrose (1959). We reviewed the literature to list the avian species living in wetlands and obtained the most recent available information on their population size in Europe. We also gathered data on the proportion of birds found with ingested gunshot in the gizzard. Since lead-induced mortality occurs especially in winter, when hunting is intensively practised (DeStefano et al., 1995; Mateo et al., 1999; Pain et al., 1997), we based our assessment on the wintering populations estimated through the International Waterbird Census (IWC) (Delany, 2005). We extracted the population sizes from the additional data files attached to the European Red List of Birds (BirdLife International, 2015). Given that population sizes are expressed with a minimum-maximum range, we took the mean of minimum and maximum values. For each species, we obtained estimates that referred both to the whole continent and the 28 EU Member States only. This latter estimate was obtained by adding the birds wintering in Croatia to the EU-level assessments originated in 2012 (before Croatia joined the European Union as its 28th Member State), from the national reports compiled to fulfill the obligation under Art. 12 of the Birds Directive (BirdLife International, 2015).

Mortality rates due to the ingestion of lead shot were evaluated for those species with sufficient available information. Mortality rates were calculated following the method proposed by Bellrose (1959) for the mallard in North America and adopted by Mateo (2009) and Pain et al. (2015) to assess the number of Anatidae annually dying as a consequence of lead poisoning in Europe and in the UK, respectively. This method involves calculating distinct mortalities according to the level of lead contamination, expressed in seven classes based on the number of ingested lead shot: 1 = 1 shot, 2 = 2 shot, 3 = 3 shot, 4 = 4 shot, 5 = 5 shot; 6 = 6 shot, 7 = ≥ 6 shot.

$$\text{Mortality } (\%) = \sum_{i=1}^{7} d_i = \frac{p_i}{h_i} \cdot t \frac{m_i}{100} \quad (1)$$

where d = % of birds dead due to lead poisoning; p = shot prevalence; h = hunting bias correction factor; t = turnover correction factor; m = mortality rate; i = class of lead contamination (as described above).

Prevalence (p) - The prevalence of lead shot ingestion is determined counting the number of pellets found in the gizzards of hunted birds. We utilised the values calculated by Mateo (2009), updated with more recent data (Mateo et al., 2014). We assumed that the mean distribution of the number of pellets in European waterfowl found with ingested gunshot was: 1 shot = 47.1%, 2 = 15.7%, 3 = 5.4%, 4 = 6.3%, 5 = 3.5%, 6 = 2%, >6 = 19.9% (Mateo, 2009).

Hunting bias (h) - Correction factors are introduced to compensate for the higher vulnerability to hunting of waterfowl with ingested lead gunshot. We divided shot prevalence by 1.5, 1.9, 2.0, 2.1, 2.2, 2.3 and 2.4 for birds with 1, 2, 3, 4, 5, 6, >6 ingested shot, respectively. These reductions are based on the results of an experimental test carried out in North America, through the release of ringed wild mallards dosed with a variable number of lead gunshot and thereafter recovered by hunters (Bellrose, 1959).

Population turnover (t) - The retention time of lead pellets in the gizzard of ducks is short (2–4 weeks) and this fact causes an underestimate when we evaluate the incidence of ingested gunshot in hunted birds. To avoid biases, we used a correction factor of 7.25, calculated on the basis of a 20-day turnover period (Bellrose, 1959) and a mean hunting season of 145 days (145/20 = 7.25).

Mortality rate (m) - Mortality rates depend on the number of ingested lead gunshot. In accordance with Bellrose (1959), we assumed that birds with 1, 2, 3, 4, 5, 6, >6 ingested shot have rates of mortality of 9, 23, 30, 36, 43, 50 and 75%, respectively.

In formula (1), the prevalence of lead gunshot ingestion p corrected per hunting bias h and population turnover t represents the percentage of birds that ingested lead gunshot and likely suffered welfare effects (s).

$$\text{Birds suffering welfare effects } (\%) = \sum_{i=1}^{7} s_i = \frac{p_i}{h_i} \cdot t \quad (2)$$

The percentage of birds suffering sub-lethal effects as a consequence of lead pellet ingestion can be estimated by subtracting the mortality value calculated though the formula (1) from the value obtained using formula (2).

The number of birds dying annually because of poisoning by lead shot was calculated by applying the additional mortality rate, obtained as above, to the populations of each species in the family Anatidae wintering in Europe and in the 28 EU Member States.

2.2. Mortality of captive-bred birds released into the wild

Post-release mortality of hand-reared birds was assessed through a bibliographic review and an ad hoc study in the field. From several researches on mallards carried out with different methodologies we deduced mortality rates for the period spanning from the release into the wild to the onset of the hunting season, so as to exclude the additional mortality due to hunting. Furthermore, we tracked 19 captive-bred mallards equipped with GPS-GSM data loggers (Ecotone Saker series) and released between 2.2.2016 and 8.3.2016 in four wetlands in N-E Italy, obtaining mortality data up to the end of September, before the opening of the hunting season.

The mortality rate of captive birds was used to calculate how many hand-reared individuals have to be released to compensate the annual loss of wild waterbirds dying as a result of the ingestion of lead gunshot (formula (3)).

$$\text{Number of captive-bred birds to release} (N) = Np/(1-pd) \quad (3)$$

where Np = number of birds died due to lead-poisoning; pd = proportion of captive-bred birds expected to die before the onset of the hunting season.

2.3. Economic value of captive-bred birds

We evaluated the commercial value of each species through a market survey, carried out from October 2016 to January 2017. Prices were requested or acquired from websites of animal dealers located in five EU countries. Values were in euros (EUR). Prices in UK pounds

(GBP) and Romanian lei (RON) were converted into euros with an exchange rate of 1 GBP = 1.1730 EUR and 1 RON = 0.2219 EUR, respectively. When different, we averaged the price of male and female birds. When more than one price was available from a single country, we calculated and used the mean value. The monetary cost of the waterbirds poisoned by lead gunshot was assessed by attributing a value to those captive-bred individuals that should be released to compensate the loss.

2.4. Economic value shooting opportunities foregone because of lead-poisoned birds

We used data on the total annual GVA of all sport shooting of game animals in the UK in GBP given by Public and Corporate Economic Consultants Limited (PACEC, 2014). PACEC (2014) does not give a breakdown of GVA by type of shooting (gamebirds, waterfowl, deer, etc.), but an earlier analysis of data for 2004 in PACEC (2006: page 21) gives the proportions of all UK gun-days (1 gun-day is one day spent by a person shooting a particular type of quarry animal). We assumed that the proportion of gun-days spent shooting ducks and geese in 2004 could be multiplied by the total GVA of shooting from PACEC (2014) to give the approximate GVA for shooting ducks and geese in the UK in 2014. We wished to express the GVA as a per capita value per bird shot. We therefore divided the GVA for UK duck and goose shooting by the total number of ducks and geese shot in 2004, taken from PACEC (2006). The numbers of mallard shot in the UK changed little between 2004 and 2013 (Aebischer, 2013) and this is the most important quarry species of ducks and geese, so assuming that the value from 2004 can be used to represent numbers of Anatidae shot in 2014 seems reasonable. Finally, we converted the GVA per bird from GBP to euros using the exchange rate 1 GBP = 1.1730 EUR and multiplied by the estimated number of additional deaths caused by lead poisoning to give the GVA of shooting opportunities foregone in Europe because of lead-poisoned birds.

3. Results

3.1. Species and number of birds annually poisoned by lead gunshot

We found evidence of lead pellet ingestion for 40 different avian species largely found in aquatic habitats and regularly occurring in Europe (Table 1). The family Anatidae accounts for the highest number of species (27), followed by Scolopacidae (5) and Rallidae (4). Among raptors, the species most exposed to secondary poisoning are the white-tailed eagle (*Haliaeetus albicilla*) and the European marsh harrier (*Circus aeruginosus*) (Pain et al., 2009).

Data on gunshot prevalence, wintering populations and economic value of captive-bred individuals were available for a subset of 16 species of Anatidae (Table 2), for which mortality rates induced by lead poisoning was calculated. These species account for 14,777,900 and 11,898,564 birds wintering annually in Europe and in the EU, respectively. Mortality rates ranged from 0% in birds mostly feeding on the sea bottom (greater scaup *Aythya marila*) or in meadows, grassland and agricultural fields (greater white-fronted goose *Anser albifrons*, barnacle goose *Branta leucopsis*) to >20% in ducks more strictly related to wetlands (northern pintail *Anas acuta*, common pochard *Aythya ferina*). Overall, we estimate that about 700,000 individuals die annually due to lead poisoning in the EU, corresponding to 6.1% of the wintering population; this figure increases to one million (7.0%) when we consider whole Europe. More than half of lead-poisoned individuals are estimated to be mallards and tufted ducks (*Aythya fuligula*). Furthermore, about 18–21% of the wintering waterbirds, corresponding to 2.2 million individuals in the EU and 3.1 in the whole of Europe, suffer sub-lethal effects as a consequence of lead gunshot ingestion (Table 2).

3.2. Mortality of captive-bred birds released into the wild

We found values of post-release mortality of captive-reared mallards in the literature ranging from 40 to 75%. In Minnesota, Schladweiler and Tester (1972) followed 56 radio-marked 6 week old individuals within 21 days after the release and reported a non-hunting mortality of 71% (40 individuals), but with significant differences (from 33% to 100%), depending on the release site and predation pressure. A similar study on 137 females radio-marked in Maryland revealed a mortality of 75% over a 160-day period following the release (Stanton et al., 1992). In the Camargue, Champagnon et al. (2012) found a mortality of about 75% in 300 ringed young birds during the flightless period (from release on June 19 to September). Another analysis based on ringing-recovery of 584 captive-bred mallards in the Camargue estimated different mortality values in birds released in wetlands with different hunting regimes (about 40% in hunting estates vs 70% in protected areas), probably owing to the reduced impact of predation in hunted areas, where the control of foxes is a common practice (Champagnon, 2011). The post-release mortality of captive-bred mallards equipped with GPS-GSM devices and released in Italy is shown in Fig. 1. At the beginning of the hunting season only 16% of the released birds was still alive (3 out of 19). Mortality was mainly due to predation (n = 10); one bird was killed by a collision with a car and the other five died from unknown causes.

When examining mortality data deriving from the bibliographic review and our original research, we observed that most values (4 out of 6) are included in a very restricted range (70–75%), whereas two are outliers (40% and 84%). We opted to calculate the mean post-release mortality of captive-bred mallards omitting the outliers. This gave a value of 72.7%. The figure of 40% estimated by Champagnon (2011) in actively-managed hunting estates was excluded because it was probably biased to be atypically low by the control of foxes carried out in the release sites until the onset of the hunting season. The control of natural predators is not a common practice in most of Europe and requires additional costs that cannot be computed in our economic assessment. We decided also to exclude the value of 84% found at the end of our field study, given the relatively low number of tracked birds and the potential negative effects of GPS-GSM devices on ducks (Kesler et al., 2014).

Given the absence of detailed post-release mortality data referred to other species, in our computation we assumed that captive waterbirds other than mallards are affected by a similar mortality level when released into the wild. By combining these results, we estimate that about 3.8–2.6 million waterbirds would need to be released to compensate for the mortality caused by poisoning with ingested lead gunshot in Europe and in the 28 EU, respectively (Table 2).

3.3. Economic value of captive-bred birds

We collected prices per captive-bred bird for 17 species from nine dealers located in five countries within the European Union (three in UK, two in France and Romania, one in Italy and one in Spain). Mean prices of each species are reported in Table 3. Mallard was the cheapest species, followed by the greylag goose (*Anser anser*) and several ducks with large wintering populations (gadwall *Mareca strepera*, Eurasian wigeon *M. penelope*, tufted duck). Swans were the most expensive species. Overall, the annual cost for replacing lead poisoned waterfowl with captive-bred individuals is estimated at 105 million euros in the EU countries and 142 million euros in the whole of Europe (Table 4). Mallards and tufted ducks alone account for about 37–39% of the amount.

3.4. Economic value shooting opportunities foregone because of lead-poisoned birds

PACEC (2014) estimated the total GVA of shooting in the UK at 2,000,000,000 GBP in 2014. PACEC (2006) estimated that 7.7% of all UK gun-days in 2004 were spent on shooting ducks and geese. Hence, we estimate the GVA of shooting of ducks and geese in the UK in 2014 as

Table 1
European avian species largely related to aquatic habitats and reported as ingesting lead gunshot from the environment.

English name	Scientific name	Conservation status[a]	Countries[b]	References[c]
Ruddy duck	*Oxyura jamaicensis*[d]	NE	ES, US	Mateo et al., 2001; Perry and Artmann, 1979.
White-headed duck	*Oxyura leucocephala*	EN - VU	ES	
Mute swan	*Cygnus olor*	LC - LC	CA, GB, IE, IT	Bowen and Petrie, 2007.
Whooper swan	*Cygnus cygnus*	LC - LC	GB, IE, JP	Ochiai et al., 1992.
Tundra swan	*Cygnus columbianus*	EN[W] - EN[W]	CA, GB	Bowen and Petrie, 2007.
Barnacle goose	*Branta leucopsis*	LC - LC	GB	Pain et al., 2015.
Canada goose	*Branta canadensis*[d]	LC - NE	GB, US	Newth et al., 2012, Szymczak and Adrian, 1978.
Greylag goose	*Anser anser*	LC - LC	ES, GB	De Francisco et al., 2003.
Pink-footed goose	*Anser brachyrhynchus*	LC - LC[W]	GB	
Greater white-fronted goose	*Anser albifrons*	LC - LC[W]	JP	Ochiai et al., 1993.
Common eider	*Somateria mollissima*	VU - EN	US	Franson et al., 1995.
Common scoter	*Melanitta nigra*	LC - LC	CA	Lemay et al., 1989 in Brown et al., 2006.
Common goldeneye	*Bucephala clangula*	LC - LC	FI, FR, GB, NL, SE	
Common shelduck	*Tadorna tadorna*	LC - LC	GB	Olney, 1965.
Marbled teal	*Marmaronetta angustirostris*	VU - CR	ES	
Red-crested pochard	*Netta rufina*	LC -LC	ES	
Common pochard	*Aythya ferina*	VU -VU	CH, ES, FI, FR, GB, GR, IT, SE	
Ferruginous duck	*Aythya nyroca*	LC - LC	ES	Mateo et al., 2001.
Tufted duck	*Aythya fuligula*	LC - LC	CH, DK, ES, FI, FR, GB, GR, SE	
Greater scaup	*Aythya marila*	VU[W] - VU	US	Bellrose, 1959.
Garganey	*Spatula querquedula*	LC - VU	FR	
Northern shoveler	*Spatula clypeata*	LC - LC	ES, FR, GB, US	Bellrose, 1959.
Gadwall	*Mareca strepera*	LC - LC	ES, FR, GB, NL	
Eurasian wigeon	*Mareca penelope*	LC - VU	DK, ES, FR, IT, SE	
Mallard	*Anas platyrhynchos*	LC - LC	CH, DK, ES, FI, FR, GB, GR, HU, NL, NO, PL, PT, SE, US	Bellrose, 1959, Binkowski and Sawicka-Kapusta, 2015.
Northern pintail	*Anas acuta*	LC - VU	CH, DK, ES, FI, FR, GB, GR, SE, US	Bellrose, 1959.
Common teal	*Anas crecca*	LC – LC	CH, ES, FR, GB, GR, IT[e]	
Greater flamingo	*Phoenicopterus roseus*	LC - LC	ES, FR, IT	
Western water rail	*Rallus aquaticus*	LC - LC	FR	
Purple swamphen	*Porphyrio porphyrio*	LC - LC	ES	
Common moorhen	*Gallinula chloropus*	LC - LC	FR, GB, US	Jones, 1939.
Common coot	*Fulica atra*	NT – LC	CH, ES, FR, PL	Binkowski and Sawicka-Kapusta, 2015
Pied avocet	*Recurvirostra avosetta*	LC – LC	ES	Guitart et al., 1994b.
Black-tailed godwit	*Limosa limosa*	VU – EN	ES, FR, IT	
Ruff	*Calidris pugnax*	LC – EN	FR, IT[e]	
Dunlin	*Calidris alpina*	LC – LC	CA	Kaiser et al., 1980.
Common snipe	*Gallinago gallinago*	LC – LC	FR, GB	
Jack snipe	*Lymnocryptes minimus*	LC – LC	FR	
Western marsh-harrier	*Circus aeruginosus*	LC - LC	ES, FR	
White-tailed sea-eagle	*Haliaeetus albicilla*	LC - LC	DE, GL, SE	Helander et al., 2009.

[a] IUCN Red List Categories assessed at a pan-European (left) and EU (right) level. LC = least concern; NT = Near Threatened; VU = vulnerable; EN = endangered; CR = critically endangered; NE = not evaluated; [W] = assessment based on wintering populations (BirdLife International, 2015).
[b] CA = Canada; CH = Switzerland; DE = Germany; DK = Denmark; ES = Spain; FI = Finland; FR = France; GB = United Kingdom; GL = Greenland; GR = Greece; HU = Hungary; JP = Japan; IE = Ireland; IT = Italy; NL = the Netherlands; NO = Norway; PT = Portugal; SE = Sweden; US = United States of America.
[c] Due to the large amount of literature for some species, only selected references are listed; when references are non indicated, see Mateo (2009).
[d] Introduced in Europe.
[e] Unpublished data.

Table 2
Annual additional mortality and morbidity caused by lead poisoning from ingested gunshot of waterfowl in Europe and in the European Union (28 Member States).

Species	Lead shot ingestion prevalence % (n[a])	Estimated mortality %	Estimated individuals suffering sub-lethal effects %	Wintering population in Europe n	Wintering population in the EU n	Estimated mortality in Europe n	Estimated mortality in the EU n	Estimated individuals suffering sub-lethal effects in Europe n	Estimated individuals suffering sub-lethal effects in the EU n
Tundra swan	0.2 (516)	0.2	0.8	22,400	22,000	45	44	179	176
Barnacle goose	0.0 (61)	0.0	0.0	718,500	718,500	0	0	0	0
Greylag goose	4.4 (203)	4.5	13.5	1,002,500	956,700	45,113	43,052	135,338	129,155
Pink-footed goose	2.7 (73)	2.8	8.2	422,500	422,500	11,830	11,830	34,645	34,645
G. white-fronted goose	0.0 (30)	0.0	0.0	1,960,000	1,866,750	0	0	0	0
Common goldeneye	16.0 (156)	16.2	48.8	440,000	376,250	71,280	60,953	214,720	183,610
Red-crested pochard	12.4 (97)	12.5	37.5	374,000	46,705	46,750	5838	140,250	17,514
Common pochard	23.1 (2333)	23.4	70.6	241,500	112,200	56,511	26,255	170,499	79,213
Tufted duck	10.5 (4208)	10.6	32.4	1,545,000	1,222,500	163,770	129,585	500,580	396,090
Greater scaup	0.0 (11)	0.0	0.0	218,500	213,514	0	0	0	0
Northern shoveler	10.4 (1515)	10.5	31.5	324,000	260,160	34,020	27,317	102,060	81,950
Gadwall	3.8 (816)	3.8	11.2	209,000	169,175	7942	6429	23,408	18,948
Eurasian wigeon	2.1 (1518)	2.1	6.9	2,295,000	2,087,000	48,195	43,827	158,355	144,003
Mallard	11.9 (20,927)	12.1	36.9	3,730,000	2,355,000	451,330	284,955	1,376,370	868,995
Northern pintail	31.5 (977)	31.9	96.1	160,000	130,610	51,040	41,665	153,760	125,516
Common teal	4.7 (43,069)	4.7	14.3	1,115,000	939,000	52,405	44,133	159,445	134,277
Total				14,777,900	11,898,564	1,040,230	725,881	3,169,609	2,214,092

[a] n represents the number of examined specimens.

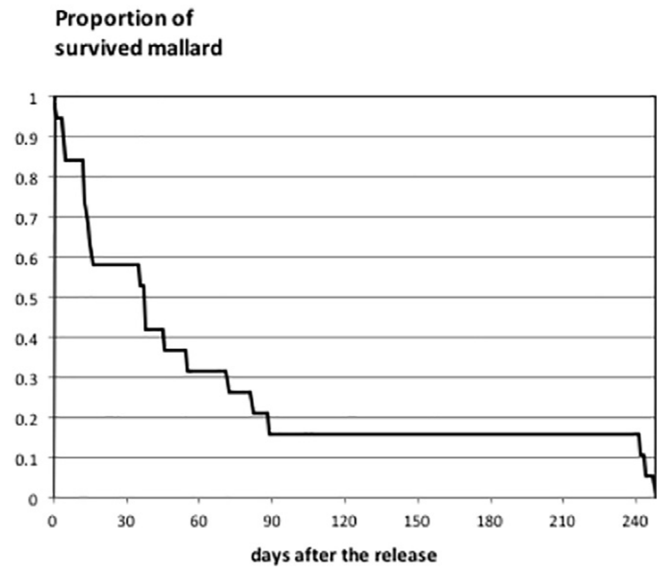

Fig. 1. Cumulative survival of 19 captive-bred mallards equipped with GPS-GSM Ecotone devices and released between 2.2.2016 and 8.3.2016 in four wetlands in N-E Italy.

Table 4
Number and economic value of captive-bred waterbirds that should be released annually in Europe and in the European Union (28 Member States) to replace wild birds died due to the ingestion of lead gunshot.

Species	Captive-bred birds to release annually (n)		Estimated costs (euros)	
	In Europe	In the EU	In Europe	In the EU
Tundra swan	164	161	74,010	72,689
Pink-footed goose	43,333	43,333	3,163,333	3,163,333
G. white-fronted goose	0	0	0	0
Greylag goose	165,247	157,698	6,940,385	6,623,308
Barnacle goose	0	0	0	0
Eurasian wigeon	176,538	160,538	8,120,769	7,384,769
Gadwall	29,092	23,548	1,309,121	1,059,668
Common teal	191,960	161,659	9,022,106	7,597,989
Mallard	1,653,223	1,043,791	29,758,022	18,788,242
Northern pintail	186,960	152,618	10,843,663	8,851,818
Northern shoveler	124,615	100,062	6,729,231	5,403,323
Red-crested pochard	171,245	21,385	6,678,571	834,018
Common pochard	207,000	96,171	9,729,000	4,520,057
Tufted duck	599,890	474,670	26,995,055	21,360,165
Greater scaup	0	0	0	0
Common goldeneye	261,099	223,269	22,976,703	19,647,692
Totals	3,810,367	2,658,905	142,339,970	105,307,070

154,000,000 GBP. PACEC (2006) estimated that 1,017,000 ducks and geese were shot in the UK in 2004. Assuming that this value can also be used for 2014, we therefore estimate a per capita GVA value for the UK of 151.42 GBP or 177.62 euros. We multiplied this by the total number of deaths per year caused by lead poisoning from Table 2 to give the economic value shooting opportunities foregone because of lead-poisoned birds. We excluded the deaths of tundra swans from this calculation because they are not usually shot by hunters, or at least not legally, and losses therefore do not have an opportunity cost for shooting. This gives a GVA of lost shooting opportunities for the whole of Europe of 185 million euros per year or 129 million euros per year for the EU countries.

4. Discussion

The methods we adopted allowed us to make two estimates of annual economic losses due to the ingestion of spent gunshot by waterfowl in Europe, one based upon replacement costs and another on the opportunity cost of waterbirds not available for shooting because of lead poisoning. Both estimates use the same values for the annual number of waterbird deaths in Europe and the EU derived using the methods of Bellrose (1959). Some uncertainty in the estimated annual number of deaths is due to the use of experimental and observational results for mallards and the assumption that they can be applied to other species. However, mallards alone account for about 40% of the overall lead-induced mortality estimated in Europe. Small-sized ducks (common teal *Anas crecca*, northern shoveler *Spatula clypeata* and tufted duck), that are more likely to die after the ingestion of a single lead gunshot (Guillemain et al., 2007), account for a further 24% of mortality. Lead-induced mortality of wild waterfowl is probably lower in the geese and swans ingesting small numbers of gunshot, because they have much larger body size than mallards. However, geese and swans together contributed <10% of the overall estimated mortality. Hence, our estimates of annual mortality are unlikely to be strongly biased by our assumptions based upon studies of mallards.

Another source of uncertainty in the estimate of the total number of deaths caused by lead-poisoning is due to some of the data on proportions of birds with ingested gunshot being derived from studies conducted several decades ago. Even if partial or total bans on the use of lead shot in wetlands were already introduced in the 1980′s, in most European countries restrictions have been adopted more recently to comply with AEWA provisions (AEWA Secretariat, 2008). Hence, in those countries with effective bans there might now be a lower density of lead shot in superficial sediments than in the past, as was observed in North America few years after the introduction of restrictions on the use of lead gunshot (Anderson et al., 2000; Samuel and Bowers, 2000; Stevenson et al., 2005). However, the small amount of documentation available for European countries of how effective partial and total bans on lead ammunition use have been indicates that compliance has been poor in some countries. Studies carried out in the UK and in Spain (Cromie et al., 2010; Mateo et al., 2014; Newth et al., 2012) suggest that the effectiveness of restrictions of the use of lead shot is partial and variable, dependent on several factors, including the extent of restrictions in each country, level of awareness among wildfowl hunters, their willingness to comply, and enforcement intensity. Considering the high variability across Europe of these factors, it becomes virtually impossible to assess how the introductions of local/national bans could have reduced waterfowl mortality. Currently few countries (e.g. Denmark, the Netherlands) have imposed a generalized ban on the use of lead ammunition over wetlands. Most states opted for partial restrictions concerning some areas or groups of species (e.g. Italy, United Kingdom) or avoided to

Table 3
Prices (in euros) of captive-bred waterfowl and coot sold in Spain (ES), France (FR), United Kingdom (GB), Italy (IT) and Romania (RO). For each country in the right column is reported the number of dealers from whom the prices originated. Means are in the last column. n.a. = data not available.

Species	ES		FR		GB		IT		RO		Mean
Tundra swan	n.a.	0	n.a.	0	452	2	450	1	n.a.	0	451
Pink-footed goose	n.a.	0	n.a.	0	47	1	100	1	n.a.	0	73
G. white-fronted goose	n.a.	0	138	2	n.a.	0	90	1	n.a.	0	114
Greylag goose	n.a.	0	58	2	23	1	45	1	n.a.	0	42
Barnacle goose	65	1	n.a.	0	38	2	45	1	n.a.	0	49
Eurasian wigeon	65	1	59	2	32	2	30	1	n.a.	0	46
Gadwall	65	1	53	2	32	1	30	1	n.a.	0	45
Common teal	65	1	61	2	32	2	30	1	n.a.	0	47
Mallard	30	1	17	2	n.a.	0	8	1	18	2	18
Northern pintail	65	1	47	2	30	3	35	1	111	1	58
Northern shoveler	70	1	66	2	45	3	35	1	n.a.	0	54
Red-crested pochard	55	1	41	2	31	3	30	1	n.a.	0	39
Common pochard	65	1	58	2	29	2	35	1	n.a.	0	47
Tufted duck	65	1	55	2	32	2	30	1	n.a.	0	45
Greater scaup	n.a.	0	n.a.	0	n.a.	0	50	1	n.a.	0	50
Common goldeneye	115	1	110	2	76	3	50	1	n.a.	0	88
Common coot	n.a.	0	n.a.	0	n.a.	0	32	1	n.a.	0	32
n of species priced/dealers	11	1	12	2	13	3	17	1	2	2	

introduced restrictions at all (e.g. Greece, Ireland) (AEWA Secretariat, 2008; ECHA, 2017; Mateo, 2009).

Our bibliographic review revealed that most waterbirds are exposed to the risk of lead gunshot ingestion (Table 1). However, there is little information on some taxonomic groups likely to be affected and so they were not included in our estimate of the annual number of waterbird deaths. There are few data on waders and rails, and therefore our knowledge on the effects of lead gunshot on these groups of waterbirds is limited compared with the large amount of evidence collected for ducks, geese and swans. Lead gunshot ingestion also probably occurs in some species not listed in Table 1, such as the northern lapwing (*Vanellus vanellus*) for which eco-toxicological studies revealed high levels of lead in tissues (Guitart et al., 1994a). Information required to assess the economic value of the losses caused by the ingestion of lead gunshot in Europe was available for just 16 species of Anatidae, comprising 40% of the species listed in Table 1. For the remaining species, we lacked data on the prevalence of lead gunshot ingestion and/or the size of the wintering population across Europe and costs of captive-bred individuals. Yet, some species not included in our evaluation are heavily affected by lead poisoning and could contribute in a significant way to increase the economic value of NRD due to the use of lead gunshot over wetlands. This is the case for the common coot (*Fulica atra*), a rail wintering in Europe with a large population, estimated in 3,740,000 and 2,440,000 individuals in the whole continent and in 28 EU Member States, respectively (BirdLife International, 2015). A rough evaluation of the number of common coots lethally poisoned by lead gunshot can be obtained by applying the same methods proposed by Bellrose (1959) for the mallard and already used in this paper to assess mortality of Anatidae. With a mean prevalence of lead gunshot ingestion of 10.6% derived from bibliographic data (Mateo et al., 2000; Mondain-Monval et al., 2002; Mudge, 1983; Pain, 1990; Thomas, 1975), we can estimate a 10.7% annual mortality due to lead poisoning in the common coot. Therefore, we can assume that 400,180 coots are lead-poisoned annually in whole Europe and 261,080 in the EU. Since this species is not frequently kept in captivity, it is not included in the price lists of most waterbird dealers. We were able to obtain just one economic evaluation from an Italian seller who indicated 32 euros per bird (Table 3). If we assume that captive-bred coots have the same post-release mortality of ducks, geese and swans (see Tavecchia et al., 2009), the monetary value of the lethally poisoned common coots can be quantified in 42.7 million euros in the whole Europe and 27.8 million in the EU. Since the common coot is just one of the 24 species listed in Table 1 and not included in our economic assessment of the cost of birds died as a consequence of the ingestion of lead gunshot, we argue that our estimate of the monetary value of the losses in waterbird populations due to the use of lead is likely to be markedly too low. It is also worth to be noted that in Table 1 are included some globally threatened species (Mateo et al., 2001), requiring targeted conservation efforts by the European Union and the Member States. The continued use of lead gunshot is likely to frustrate the conservation programmes adopted for such species, consequently causing a motiveless waste of public resources.

Our estimates of replacement costs based upon per capita costs of captive-bred birds and estimates of the proportion of them that die before the hunting season are also subject to uncertainties. The market surveys of the prices of captive-bred waterbirds that we used in our assessment were of mean prices of birds sold at retail in different parts of Europe. Reduction of costs could perhaps have been obtained by buying wholesale. However, the higher the number of birds bought from a single seller, the higher would be the translocation costs needed to move the birds to dispersed release sites, to avoid unnatural densities in restricted areas. We did not include these transport costs, which would, at least in part, diminish any savings in purchase price from buying wholesale. We used post-release survival results for captive-bred mallards and applied them to all waterbird species. The assumption that captive waterbirds other than mallards are affected by a similar mortality level when released into the wild is supported by the evidence that most captive waterbirds face similar problems after release (Green et al., 2005; Tavecchia et al., 2009).

Our alternative estimate of the annual economic value of waterbirds killed by lead poisoning based upon lost opportunities for hunting gave remarkably similar values to those based upon replacement costs. For the EU 28 we estimated an annual loss of 105 million euros per year from replacement costs and 129 million euros per year from lost hunting opportunities. The equivalent costs for the whole of Europe were 142 million and 185 million euros per year. It could be argued that the deaths of some birds because of lead poisoning are compensated for by density-dependent enhancement of the survival of the birds that were not poisoned. If this was a large effect, it might make our estimate of GVA lost too large. However, the most thorough analyses of common hunted waterfowl indicate that density dependence is too weak to have large effects of this kind (Nichols et al., 2015).

Our findings largely agree with the assessment carried out in North America by Norton and Thomas (1994) to estimate the economic value of wild ducks shot by hunters and un-retrieved ("crippling losses"). As the sale of waterfowl is prohibited by law in North America and harvested ducks consequently have no direct market value, these authors adopted an approach based on the evaluation of the costs required to manage an equivalent area of wetlands to produce the same number of wild ducks as those lost due to crippling. They found a mean value of 67.80 US dollars per duck produced, corresponding to about 129 revalued dollars (assuming that 1.00 dollar in 1990 had the same buying power as 1.91 in 2017) and 120 euros. This is reasonably similar to the mean values per bird obtained in our assessments, ranging between 137 and 145 euros per bird for replacement costs and 178 euros per bird based upon lost hunting opportunities.

Our economic assessment took into account only the immediate mortality due to the ingestion of lead gunshot, but avoided considering the longer-term impacts of lead-induced morbidity on waterbird demography. Moderate or even low exposure of birds to lead may result in a decrease of survival and productivity (Burger, 1995). Lead-impaired individuals in controlled experiments showed altered immunologic function and altered behaviours, including locomotion, righting and depth perception, and they may therefore survive and breed poorly in the wild even if lead poisoning does not have an immediate fatal outcome (Burger and Gochfeld, 1985). Similar results originated from surveys carried out on wild birds (Berny et al., 2015; Kelly and Kelly, 2005; Vallverdú-Coll et al., 2015, 2016).

In combination, acute lead poisoning and longer term sub-lethal effects of lead are likely to have a significant effect at population level (Anderson et al., 2000; Guillemain et al., 2007; Tavecchia et al., 2001). Concern arises particularly when we consider species with a very high prevalence of individuals suffering lead poisoning, such as northern pintail and common pochard (Table 2). The hypothesis that ingested lead gunshot might be affecting population trends of wintering wildfowl has received support from correlation studies of Europe-wide population trends in relation to gunshot ingestion prevalence and recent research by Green and Pain (2016), who found a significant negative correlation in mean population growth rate in the UK across eight duck species during the period 1990/1991 to 2013/2014 with two independent measures of the prevalence of ingested lead gunshot in the UK and Europe. We emphasise that our assessment of the economic cost of lead poisoning does not include effects on population size. If waterbird populations are lower than they might otherwise be because of lead poisoning, as seems likely, the economic costs would be larger than our estimates.

Notwithstanding our results are likely to be conservative, they suggest that the benefits of a restriction on the use of lead gunshot over wetlands could largely exceed the cost of adapting to non-lead ammunition at least in the EU, where societal costs of a lead ban have been recently estimated in 35–61 million euros/year (ECHA, 2017).

5. Conclusions

Our analyses indicate that about one million individuals die annually in Europe and an additional three million suffer sub-lethal effects as a consequence of lead gunshot ingestion. The annual economic loss related to waterbird mortality by ingested spent lead gunshot from sport shooting lies in the range 100–200 million euros. These estimates are subject to uncertainty, but the removal of the most likely potential sources of bias would result in higher values for annual economic cost. Inclusion of deaths of species for which there were insufficient data and delayed deaths caused indirectly by lead poisoning and effects on reproduction would probably increase the estimated losses substantially. Nevertheless, our results suggest that the benefits of a restriction on the use of lead gunshot over wetlands could exceed the cost of adapting to non-lead ammunition.

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