Swans and lead fishing weights: a systematic review of deposition, impacts and regulations in Europe

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Abstract

Lead is a potent metabolic poison that is highly toxic to all life; the World Health Organization states that there is no safe level of exposure to lead. Despite its toxicity, lead is still commonly used in some recreational activities, such as the "sinkers" and other lead weights used in fishing. Swans foraging for grit to aid digestion may ingest discarded lead weights, resulting in lead poisoning. Here, we aimed to synthesise the information currently available on lead fishing weights and swans in Europe, to inform current and future policy. We carried out a systematic review of the literature from Europe on the deposition of lead fishing weights and the impacts of lead (Pb) poisoning associated with fishing weights on swans, as well as efforts to mitigate these impacts through regulations. On screening 605 papers identified during our literature searches, we found 45 relevant papers from which information could be extracted. These indicated that the deposition of lead fishing weights resulted in accumulated densities of up to 339 sinkers m^{-2} with an estimated deposition rate of *c.* 3,000 tonnes per year within the 27 EU countries alone. Elevated lead concentrations in blood and other tissues and organs of both Mute Swans *Cygnus olor* and Whooper Swans *Cygnus cygnus* have been reported. The mean blood lead concentrations in 19 of the 20 studies conducted where lead fishing weights were used exceeded the threshold of 20 μg dL^{-1} that is typically considered indicative of elevated blood lead concentrations. Moreover, the maximum blood lead concentration reported in each of the 20 studies exceeded the 20 μg dL^{-1} threshold, with a maximum reported value of 5,134 μg dL^{-1}. The mean kidney and liver lead concentrations estimated from our review were 642.5 μg g^{-1} DM and 79.3 μg g^{-1} DM, respectively, and elevated above background levels of 10 μg g^{-1} DM. Impacts of lead fishing weights on the breeding success of Mute Swans included a reduced chance of breeding successfully amongst individuals with elevated blood lead concentrations, as well as increased pre- and post-fledging cygnet mortality. Lead fishing weights have been a cause of mortality in both Mute Swans and Bewick's Swans *Cygnus columbianus bewickii*, accounting for up to 70% of swan deaths in areas with recreational fishing. Evidence from the UK demonstrates that lead poisoning due to fishing weights limited Mute Swan population size, which approximately doubled after key sizes of lead fishing weights were banned in 1987. Within Europe, statutory regulations on lead fishing weights currently exist only in the United Kingdom and Denmark, with some voluntary actions to reduce and phase-out such lead use in Belgium, the Netherlands and Sweden. Proposals which would ban lead fishing weights in all 27 European Union member states are currently under consultation. Despite some progress in reducing the impacts of lead fishing weights on swans in a few countries, the available data on swan mortality and elevated Pb concentrations in tissue and organs demonstrate that lead poisoning due to fishing weights remains a major threat to swans within Europe.

Key words: angling, lead poisoning, Pb effects on wildlife, plumbism, recreation impacts, sinkers.

Lead has been widely used by people throughout human history (Stroud 2015), yet lead is highly toxic to all life, with widespread evidence of impacts on human and wildlife health and the environment (Needleman 2004). The World Health Organization has stated unequivocally that there is no safe level of exposure to lead (WHO 2022). Decades of research have shown that the exposure of animals to lead can reduce fecundity and survival, with impacts on immunocompetence, as well as on behaviours such as movement, foraging and predator evasion (Haig *et al.* 2014; Williams *et al.* 2018). Through these effects, lead poisoning can ultimately limit animal population size in contaminated areas (Grade *et al.* 2018; Wood *et al.* 2019). Decades of research have also shown that human uses of lead affect a wide range of animal taxa, including many birds, mammals and reptiles (Haig *et al.* 2014; Williams *et al.* 2018; Grade *et al.* 2019; Chiverton *et al.* 2022; Humphries *et al.* 2022). In addition to population-level effects, the use of lead has implications for the sublethal welfare of individual animals, for example through physiological effects including reduced body condition (Newth *et al.* 2016).

Concern regarding the impacts of lead has resulted in increased regulation of the human use of lead, for instance by replacing lead in materials (*e.g.* paint, fuel and pipes) with non-toxic alternatives (Smith & Flegal 1995; Stroud 2015). Yet lead continues to be used in some recreational activities such as fishing and shooting, both of which involve discarding lead into the environment (Scheuhammer & Norris 1996; Scheuhammer & Thomas 2011).

Recreational fishing ("angling") is popular throughout Europe, with up to 23 million anglers in the 27 EU countries alone, and > 830,000 in the UK (European Chemicals Agency 2021; Environment Agency 2021). A variety of lead weights are

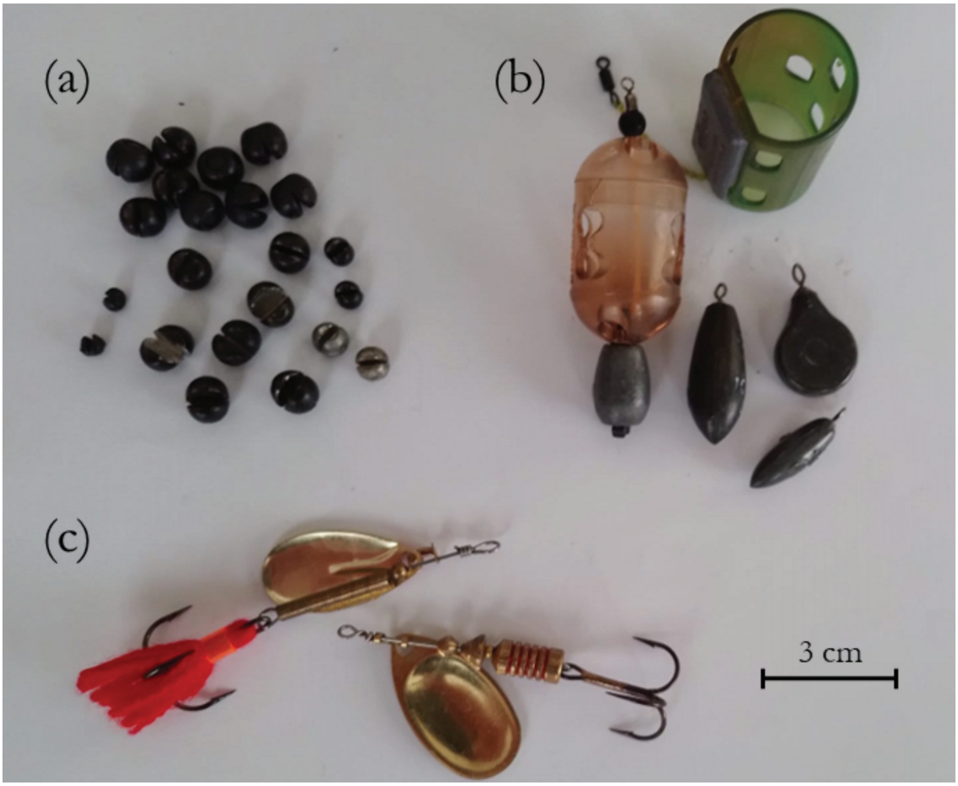

Figure 1. Some of the many different types of weights used in recreational fishing: (a) small sinkers, split-shot; (b) larger sinkers, ledger weights, feeders; (c) lures (in this case, two spinners). Note: all items shown are made of non-toxic materials.

used in recreational fishing, each available in a wide range of sizes and shapes, with different names in Europe and North America (Fig. 1). Small, round particles of lead, commonly pinched or threaded onto fishing line to provide weight are known as "sinkers" in North America and as "split-shot" in Europe. Larger and heavier weights ("sinkers" in North America but termed "ledger weights" in Europe) are typically tied onto fishing line, to tether a bait in place or to anchor a fishing rig to the bottom of the waterbody. Finally, to catch predatory fish, anglers may use artificial baits known as "lures", which often feature weighted parts made of lead, to imitate smaller fish or other aquatic prey animals. Numerous different lure types include "spinners" (featuring one or more passively revolving metallic blades around a metal shaft with a terminal hook), "spoons" (oblong metal lures shaped like a spoon), and "plugs", "wobblers", "crankbaits" or "jigs", which consist of a hook with a weighted head adorned with fabric or plastic to imitate prey species. These types of lead are used predominantly in "coarse" or "sport" fishing, rather than "game" or "fly" fishing which uses relatively little lead (Spray & Milne 1988). Lead can also be used in commercial fishing, for example in lead-core nets (Tateda *et al.* 2014); however, in this article we focus our attention on recreational fishing.

Archaeological evidence shows that lead weights have been used in fishing for thousands of years (Aleem 1972; Galili *et al.* 2002). Fishing gear containing lead weights were found during an archaeological investigation of a Roman shipwreck site in the eastern Mediterranean (Galili *et al.* 2013). Evidence also illustrates widespread use of lead fishing weights by fishing communities in the 12–13th centuries across Europe, from southeast England (Riddler & Trazaska-Nartowski 2009) to southern Belarus (Lyashkevich 2009).

Non-toxic alternatives are available for each different type of fishing weights, made of bismuth, tin, tungsten, steel or ceramic (Seed *et al.* 1995; Grade *et al.* 2019), yet lead remains in use across much of the world including Europe (European Chemicals Agency 2021). Lead weights in fishing rigs can be lost accidentally or discarded deliberately into aquatic and adjacent riparian habitats, for example when lead sinkers fall off the line or the rig becomes irretrievably snagged on underwater debris (Grade *et al.* 2019). Piscivorous birds such as divers (*Gavia* sp.) are also susceptible to ingesting lead from lures (such as jigs) which imitate prey fish, or from lost rigs fitted to prey species, *i.e.* "live baits" (Pokras *et al.* 2009; Grade *et al.* 2019). The total loss of anglers' lead into the environment can be substantial, with an estimated *c.* 3,000 tonnes of fishing lead discarded annually in European Union countries (European Chemicals Agency 2021), and *c.* 4,384 tonnes per year lost in the United States (Scheuhammer *et al.* 2003).

Swans' habitat use and behaviour make them highly susceptible to poisoning from lost and discarded lead fishing weights (Blus *et al.* 1994). In particular, their use of aquatic and adjacent riparian habitats show a very strong overlap with places where recreational fishing occurs (Sears 1989). When feeding in these habitats, the birds ingest small particles of grit, which they retain in their gizzard to aid the digestion of plant material (Gionfriddo & Best 1999); however, swans and other wildfowl (family: Anatidae) apparently do not distinguish between lead and stone particles, and so ingest discarded lead when foraging for grit (Franson *et al.* 2001).

Conservation organisations have long argued for legislation to regulate, phase-out and prohibit the use of lead in recreational activities such as fishing (UNEP–AEWA 2011). Currently, the need for such legislation within the European Union (EU) is being considered (European Chemicals Agency 2021). Scientific evidence that could inform such policy is published across a wide range of disciplines, and so reviews and synthesis articles are particularly valuable for helping to shape evidence-based legislation and policy. Whilst policy-makers can draw on earlier reviews of the scientific literature on swans and lead fishing weights (*e.g.* O'Halloran *et al.* 1991; Sears & Hunt 1991; Blus 1994), there is also a need for a compilation of more recent research. Here, we provide a state-of-the-art synthesis of the available information on lead fishing weights and swans in Europe, to inform current and future research and policy. We carried out a systematic review of the deposition and accumulation of lead fishing weights in swan habitats, and the impacts of the lead weights on the three native swan species found in Europe: Bewick's Swans *Cygnus columbianus bewickii*, Mute Swans *Cygnus olor* and Whooper Swans *Cygnus cygnus* (Rees *et al.* 2019). Furthermore, we also report the current and proposed future regulations on lead fishing weights in Europe.

Methods

We conducted searches in both Scopus (https://www.scopus.com/) and PubMed (https://pubmed.ncbi.nlm.nih.gov/), using the search terms detailed in Appendix 1. We also searched earlier reviews (O'Halloran *et al.* 1991; Sears & Hunt 1991; Blus 1994; Kirby *et al.* 1994; Michael 2006; Grade *et al.* 2019), and grey literature documents (*e.g.* UNEP–AEWA 2011; European Chemicals Agency 2021) identified through our own knowledge of the literature, to identify any articles which appeared relevant. In total, our searches identified 605 unique articles (Appendix 1).

These 605 papers were screened to assess their relevance against our criteria for inclusion in our systematic review. Our inclusion criteria for inclusion in our final dataset were that a document should report an original study of the impacts of lead fishing weights in Europe, on one of the following: (i) deposition and accumulation of lead in swan habitats, (ii) lead concentrations in swan tissues or organs, (iii) impacts on reproduction, (iv) impacts on mortality and (v) impacts on populations. We evaluated papers first by their title, then abstract, and finally full text, rejecting papers judged not to meet the inclusion criteria at each stage (Appendix 1), resulting in 45 papers published between 1973 and 2022 (Fig. 2) from 7 different countries, with 28 of which (62%) were from the UK, while Ireland and Poland accounted for 4 studies (9%) each (Fig. 3).

We extracted the following information from each: (i) deposition rate estimates, or accumulated densities of lead fishing weights in any aquatic or riparian habitats; (ii) lead concentrations in swan tissues or organs, measured at sites where recreational fishing occurs; (iii) information on lead poisoning effects due to fishing weights on any aspects of swan reproduction; (iv) the numbers of swans killed by lead poisoning due to fishing weights, as a percentage of all dead swans assessed; (v) effects of lead poisoning due to fishing weights on total numbers or population trends (including benefits that follow lead regulation). To aid comparisons between studies we standardised the reported values: lead deposition values were expressed as densities of lead weights m^{-2}; blood lead concentrations were standardised to values expressed as $\mu g\ dL^{-1}$, assuming 13.7 g Hb dL^{-1} after O'Halloran *et al.* (1991); and organ lead concentrations were standardised to values expressed as $\mu g\ g^{-1}$ dry mass (hereafter DM), with conversion from reported wet mass (WM) carried out using the *OrgMassSpecR* package in R (Dodder & Mullen 2017; R Core Team 2022). These calculations were based on the mean water content values reported by Hughes (1974) for freshwater-adapted Black

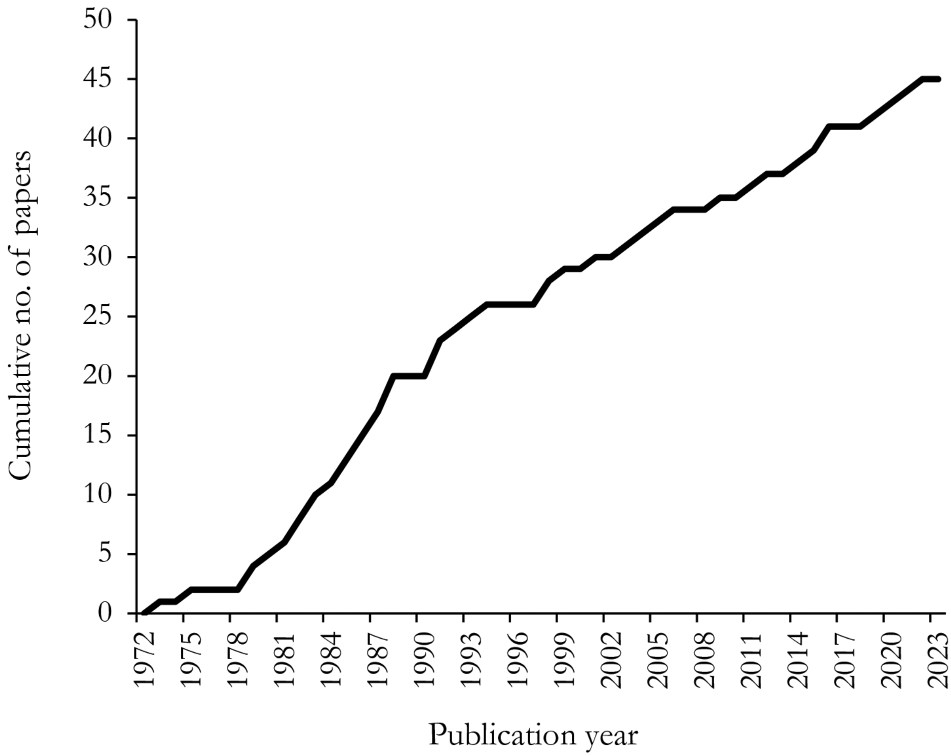

Figure 2. The cumulative number of papers on swans and lead fishing weights in Europe, included in our review.

Swans *Cygnus atratus*: 80.08% for kidney, 71.40% for liver, 78.55% for heart, 77.05% for muscle, 70.95% for adrenal gland and a mean of these values (75.61%) for all other tissue/organ types.

Further to the review of deposition and impacts, we obtained information on current and proposed regulations from relevant policy documents (Michael 2006; UNEP–AEWA 2011; European Chemicals Agency 2021). Relevant information included whether the regulations were statutory or voluntary, the start date, whether the restrictions applied to the import, sale or use of lead weights, as well as any caveats such as whether the regulations were limited to certain sizes of lead weights or certain habitats.

Maps showing: (i) the locations where the relevant studies identified in our review were conducted, and (ii) the different types of regulations on lead fishing weights within Europe, were produced using QGIS version 3.32.2.

Results

Deposition of fishing lead in the environment

We identified eight deposition rates estimates of lead fishing weights and a further 11

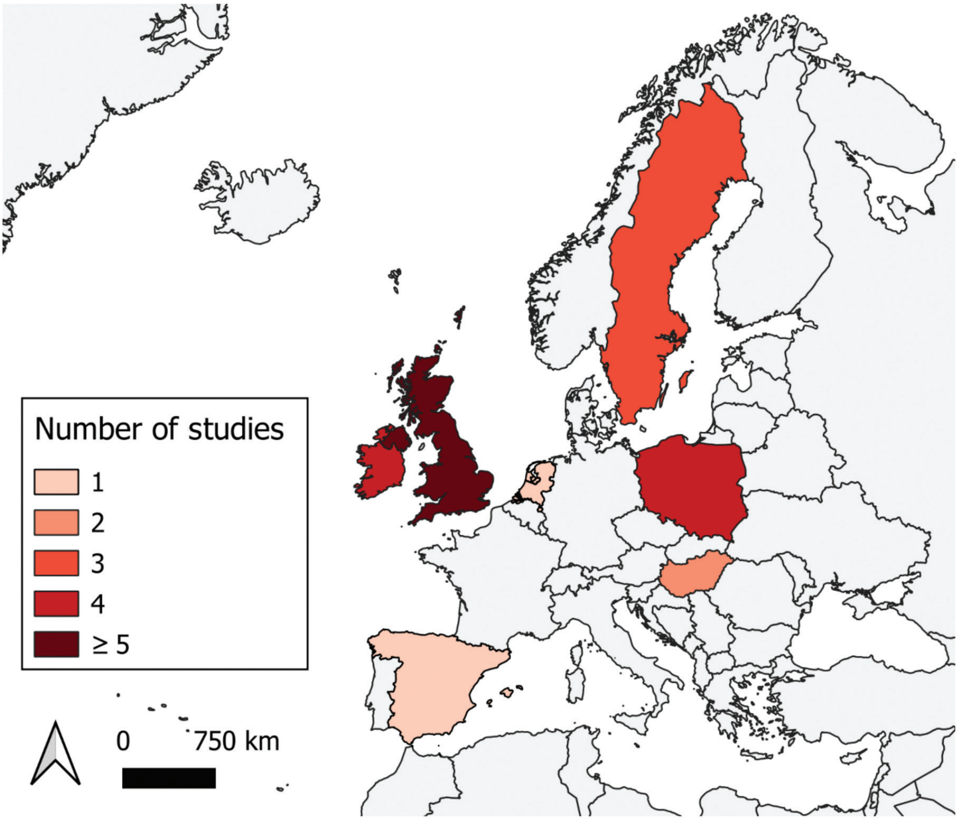

Figure 3. A map showing the number of studies used in our review that were carried out in each country. A total of 28 published studies were conducted in the UK.

estimates of accumulated lead densities, from 11 distinct studies (Table 1). These studies indicated that lead can be deposited at rates of up to 40 sinkers m^{-2} yr^{-1} and 12 g sinkers m^{-2} yr^{-1} (Sears 1988). Of the 6 studies which reported mean estimates of lead densities, the mean density was 31 sinkers m^{-2} (Table 1), although values up of 339 sinkers m^{-2} were identified (Bell *et al.* 1985). The most recent large-scale estimate of lead fishing weights deposition rates was *c.* 3,000 tonnes/year discarded into the 27 EU countries (European Chemicals Agency 2021). Jacks *et al.* (2001) estimated that, of the deposited fishing lead that remained unburied, *c.* 1% dissolved annually, and so deposited lead will typically be highly persistent in the environment.

Impacts of fishing lead on swans

Lead concentrations in swan tissues/organs

The search revealed 54 lead level concentrations from 12 different tissues from Mute Swans (n = 48) and Whooper Swans (n = 6), measured at sites where recreational fishing occurred. These were

Table 1. A summary of the deposition rates and accumulated densities of lead fishing weights within aquatic and riparian habitats in Europe. Units refer to individual sinkers unless otherwise stated.

Country/ region	Habitat	Deposition rate	Range	Units	Density	Range	Units	Reference
UK	Rivers/canals	6.6	–	m^{-1} yr^{-1}	–	–	–	Nature Conservancy Council (1981)
UK	Rivers	250.0	–	tonnes yr^{-1}	–	–	–	Birkhead (1982)
UK	Lake	1.5		m^{-2} yr^{-1}	120.5	26.0–339.0	m^{-2}	Bell *et al.* (1985)
UK	Lake	–	–	–	15.6	0.0–90.0	m^{-2}	Forbes (1986)
UK	Coastal	–	–	–	–	5.0–300.0	m^{-2}	Cryer *et al.* (1987)
UK	Rivers	–	–	–	4.3	0.9–11.4	m^{-2}	Sears (1988)
UK	Rivers	–	–	–	25.8	5.1–98.1	m^{-2}	Sears (1988)
UK	Rivers	–	–	–	8.6	1.8–31.2	g m^{-2}	Sears (1988)
UK	Rivers	19.5	2.5–40.1	m^{-2} yr^{-1}	17.9	2.3–36.7	m^{-2}	Sears (1988)
UK	Rivers	6.2	1.0–11.8	g m^{-2} yr^{-1}	5.7	0.9–10.7	g m^{-2}	Sears (1988)
UK	Lake	–	–	–	2.1	–	m^{-2}	Sears (1988)
Ireland	Lake	–	–	–	–	0.0–250.0	m^{-2}	O'Halloran *et al.* (1988a)
Sweden	Rivers	–	100–200	tonnes yr^{-1}	–	–	–	Jacks *et al.* (2001)
Spain	Coastal	–	–	–	2.8	0.3–6.7	g m^{-2}	Lloret *et al.* (2014)
EU	All aquatic and riparian	–	2,000–6,000	tonnes yr^{-1}	–	–	–	UNEP–AEWA (2011)
EU	All aquatic and riparian	3,000	2,000–7,000	tonnes yr^{-1}	–		–	European Chemicals Agency (2021)

The symbol – indicates no data reported.

reported for 12 different tissues or organs (adrenal glands, blood, bone, brain, breast muscle, feather, gizzard muscle, heart muscle, kidney, liver, pancreas and spleen) of Mute Swans (n = 48) and Whooper Swans (n = 6; Table 2). Lead concentrations were most frequently reported for blood, kidney and liver, which accounted for 37.0%, 18.5% and 22.2% of our identified values, respectively, with all other tissue or organ types combined accounting for only 22.2% of the values (Table 2).

The mean blood lead concentration of the 20 mean values identified in our review was 126.8 μg dL^{-1} (Table 2). The threshold of 20 μg dL^{-1} which indicates elevated blood lead concentrations (Franson & Pain 2011) was exceeded in 19 out of 20 reported mean values (Table 2). Moreover, the maximum blood lead concentrations for all 20 reported values exceeded the 20 μg dL^{-1} threshold (Table 2). The highest blood lead concentration reported was 5134 μg dL^{-1} (Martin 2016).

The mean kidney and liver lead concentrations of the mean values identified in our review were 642.5 μg g^{-1} DM (n = 10) and 79.3 μg g^{-1} DM (n = 12), respectively (Table 2). An earlier review by Franson & Pain (2011) suggested that kidney and liver concentrations of > 10 μg g^{-1} DM were elevated above background levels; this threshold was exceeded in 9 of the 10 reported mean values for kidneys, and in 9 of the 12 mean values reported for livers (Table 2).

Elevated lead concentrations in tissues and organs have long been known to indicate lead poisoning. For example, Birkhead (1982) found that Mute Swans which died from lead poisoning had significantly higher liver and kidney lead concentrations; the median kidney lead concentration in individuals that had ingested lead fishing weights was 908 μg/g DM compared to 8 μg/g DM in birds without weights.

Blood lead concentrations in areas with recreational fishing are typically highest during the fishing season. As an example, Birkhead (1983) found statistically significant variation between months in the median blood lead concentrations recorded among Mute Swans on the River Thames (southern UK), and these were highest during the fishing season. As well as the temporal link between recreational fishing and elevated lead concentrations, there is also evidence of spatial associations. Sears (1988) compared blood lead concentrations among Mute Swans living on gravel pits where fishing occurred, and those where no fishing occurred; swans living on the fished lakes had a statistically significant higher median blood lead concentration of 79 μg/dL (range = 4.0–962.0) compared with 21 μg/dL (range = 1–128) for swans on the unfished lakes.

Further evidence of the link between fishing and lead poisoning of swans is provided by comparative investigations of the susceptibility of different waterbird species. For example, Perrins *et al.* (2003) showed that among waterbirds within the River Avon catchment in western England, Mute Swans that foraged predominantly in the river where recreational fishing occurred, showed significantly higher blood lead concentrations than Canada Geese *Branta canadensis* which foraged predominantly on adjacent grasslands. Similarly, O'Halloran

Table 2. Lead concentrations in Mute Swan (MS) and Whooper Swan (WS) tissues and organs, measured at sites where recreational fishing occurs.

Species	Country	Years	Tissue/Organ	Mean	Range	Units	*n*	Reference
MS	UK	1973	Adrenal gland	23.0	16.0–29.0	μg g^{-1}	2	Simpson *et al.* (1979)
MS	Ireland	1983–2000	Blood	57.5	4.8–3,060.0	μg dL^{-1}	590	O'Halloran *et al.* (2002)
MS	Ireland	1984–1986	Blood	38.5	4.1–109.6	μg dL^{-1}	823	O'Halloran *et al.* (1991)
MS	Ireland	1984–1986	Blood	301.9	87.0–745.3	μg dL^{-1}	4	O'Halloran *et al.* (1988b)
MS	Ireland	2003–2006	Blood	30.8	0.8–572.7	μg dL^{-1}	127	O'Connell *et al.* (2009)
MS	Poland	2012–2013	Blood	24.1	6.4–67.5	μg dL^{-1}	49	Binkowski *et al.* (2016)
MS	Poland	2012–2015	Blood	23.9	2.8–67.5	μg dL^{-1}	45	Meissner *et al.* (2020)
MS	Poland	2015–2017	Blood	7.7	0.0–2,263.3	μg dL^{-1}	63	Kucharska *et al.* (2022)
MS	UK	1973	Blood	375.0	0.0–3,290.0	μg dL^{-1}	25	Simpson *et al.* (1979)
MS	UK	1980–1981	Blood	49.1	0.0–3,730.0	μg dL^{-1}	395	Birkhead (1983)
MS	UK	1980–1981	Blood	76.0	4.0–820.0	μg dL^{-1}	74	Birkhead & Perrins (1985)
MS	UK	1982	Blood	354.0	–	μg dL^{-1}	1	Birkhead *et al.* (1982)
MS	UK	1982–1985	Blood	77.0	1.0–2,715.0	μg dL^{-1}	502	Sears (1988)
MS	UK	1982–1985	Blood	79.0*	4.0–962.0	μg dL^{-1}	29	Sears (1988)
MS	UK	1983–1988	Blood	73.5	1.0–2,495.0	μg dL^{-1}	408	Sears & Hunt (1991)
MS	UK	2000–2001	Blood	108.0	2.1–3,010.4	μg dL^{-1}	238	Perrins *et al.* (2003)
MS	UK	2000–2002	Blood	104.0	0.2–2,360.2	μg dL^{-1}	921	Kelly & Kelly (2004)
MS	UK	2000–2002	Blood	55.9	0.2–2,360.2	μg dL^{-1}	43	Kelly & Kelly (2005)

Table 2 (*continued*).

Species	Country	Years	Tissue/Organ	Mean	Range	Units	*n*	Reference
MS	UK	2006	Blood	352.6	44.5–1,184.6	µg dL^{-1}	1	Cousquer (2006)**
MS	UK	2007–2012	Blood	69.7	0.2–5,133.6	µg dL^{-1}	1,474	Martin (2016)
MS	UK	1973	Bone	762.0	212.0–1,255.0	µg g^{-1}	4	Simpson *et al.* (1979)
MS	UK	1983–1988	Bone	146.5	7.0–2,776.0	µg g^{-1}	141	Sears & Hunt (1991)
MS	UK	1973	Brain	65.0	26.0–150.0	µg g^{-1}	5	Simpson *et al.* (1979)
MS	Ireland	1984–1986	Breast	83.7	15.2–200.4	µg g^{-1}	3	O'Halloran *et al.* (1988b)
MS	Hungary	1993	Feather	3.1	–	µg g^{-1}	1	Szabó (1998)
MS	Hungary	2015	Feather	1.1	< 0.5–4.1	µg g^{-1}	17	Grúz *et al.* (2015)
MS	Ireland	1984–1986	Gizzard	58.0	24.4–122.0	µg g^{-1}	3	O'Halloran *et al.* (1988b)
MS	Ireland	1984–1986	Heart	273.2	5.6–745.9	µg g^{-1}	3	O'Halloran *et al.* (1988b)
MS	Ireland	1984–1986	Kidney	974.4	92.4–1,676.7	µg g^{-1}	4	O'Halloran *et al.* (1988b)
MS	Sweden	1973–1977	Kidney	22.1	0.5–205.8	µg g^{-1}	58	Frank & Borg (1979)
MS	Sweden	1988–1990	Kidney	5.0	0.0–138.1	µg g^{-1}	47	Mathiasson (1993)
MS	UK	1973	Kidney	1,734.0	350.0–6,550.0	µg g^{-1}	18	Simpson *et al.* (1979)
MS	UK	1979–1981	Kidney	908.0*	105.0–5,225.0	µg g^{-1}	51	Birkhead (1982)
MS	UK	1982	Kidney	1,085.0	–	µg g^{-1}	1	Birkhead *et al.* (1982)
MS	UK	1983–1988	Kidney	341.0	27.0–6,560.0	µg g^{-1}	142	Sears & Hunt (1991)
MS	UK	1996	Kidney	233.1	2.0–869.0	µg g^{-1}	14	Pennycott (1998)
MS	Ireland	1984–1986	Liver	283.9	146.9–457.0	µg g^{-1}	4	O'Halloran *et al.* (1988b)

Table 2 (*concluded*).

Species	Country	Years	Tissue/Organ	Mean	Range	Units	*n*	Reference
MS	Poland	2009–2010	Liver	2.7	2.6–3.0	μg g^{-1}	3	Komosa *et al.* (2012)
MS	Sweden	1973–1977	Liver	5.6	0.3–55.9	μg g^{-1}	57	Frank & Borg (1979)
MS	Sweden	1988–1990	Liver	8.0	0.3–250.0	μg g^{-1}	47	Mathiasson (1993)
MS	UK	1973	Liver	117.0	51.0–206.0	μg g^{-1}	18	Simpson *et al.* (1979)
MS	UK	1979–1981	Liver	105.0*	10.0–562.0	μg g^{-1}	50	Birkhead (1982)
MS	UK	1979–1981	Liver	19.0	11.6–32.7	μg g^{-1}	6	Mudge (1983)
MS	UK	1982	Liver	120.0	–	μg g^{-1}	1	Birkhead *et al.* (1982)
MS	UK	1983–1988	Liver	46.0	0.4–404.0	μg g^{-1}	143	Sears & Hunt (1991)
MS	UK	1996	Liver	48.9	1.0–255.0	μg g^{-1}	14	Pennycott (1998)
MS	Ireland	1984–1986	Pancreas	175.1	41.4–278.8	μg g^{-1}	4	O'Halloran *et al.* (1988b)
MS	UK	1973	Spleen	67.0	27.0–117.0	μg g^{-1}	12	Simpson *et al.* (1979)
WS	Ireland	1984–1986	Blood	229.9	87.0–372.7	μg dL^{-1}	2	O'Halloran *et al.* (1988b)
WS	Ireland	1984–1986	Kidney	1,021.6	235.9–1,807.2	μg g^{-1}	2	O'Halloran *et al.* (1988b)
WS	Sweden	1973–1977	Kidney	100.4	0.5–381.5	μg g^{-1}	4	Frank & Borg (1979)
WS	Ireland	1984–1986	Liver	164.3	–	μg g^{-1}	1	O'Halloran *et al.* (1988b)
WS	Sweden	1973–1977	Liver	30.8	0.3–111.9	μg g^{-1}	4	Frank & Borg (1979)
WS	Ireland	1984–1986	Pancreas	1,742.5	1,640.0–1,845.0	μg g^{-1}	2	O'Halloran *et al.* (1988b)

All mass values are expressed as dry matter weight. * Median value reported in absence of a mean; ** Pb concentrations reduced by veterinary intervention; DM = dry mass; *n* indicates number of birds sampled; the symbol – indicates no data reported.

et al. (1988a) showed that urban grassland was not a cause of lead poisoning among waterbirds in Ireland, whereas lead-poisoned swans frequently contained ingested lead fishing weights.

Some evidence from time-series studies suggests that blood lead concentrations have been reduced through conservation actions. Sears & Hunt (1991) documented a significant decrease in median blood lead concentrations among Mute Swans from the River Thames, from a median of 127 μg dL^{-1} in 1984 to 22 μg dL^{-1} in 1987, following the introduction of regulations on lead fishing weights. The pronounced peaks in both the blood lead concentrations and the number of cases of swans with lead poisoning that coincided with the peak summer fishing season disappeared after the ban (Sears & Hunt 1991). UK Mute Swan blood lead concentrations fell from 119 μg dL^{-1} in 2007 to 59 μg dL^{-1} in 2012, in line with the expected trend following the 1987 regulations on lead fishing weights (Martin 2016). A long-term study of Mute Swans wintering at Cork Lough in Ireland found a decline in median blood lead concentrations between 1983/84 and 2006, which may have been linked to the provision of grit which reduced the ingestion of lead weights (O'Connell *et al.* 2009).

The data identified in the review showed considerable geographic bias; the UK and Ireland accounted for 52% and 26% of the reported values, with the remainder comprised of studies from Sweden, Poland and Hungary (Table 2). Whilst the literature searches found studies which reported lead concentrations in swan tissues and organs for other regions of Europe such as Italy (Isani *et al.* 2013) and Greece (Aloupi *et al.* 2017), these studies did not mention any recreational fishing within their study areas, and so we could not include them. Additionally, in other studies it was not clear whether lead poisoned birds were indicative of lead fishing weights, as the source(s) of lead were not specified, so these were also omitted from our assessment; for example, several studies mentioned "lead pellets" that were most likely ammunition rather than fishing weights (*e.g.* Clausen & Wolstrup 1979; Eskildsen & Grandjean 1984).

Impacts on reproduction

Two studies reported information on lead poisoning impacts on swan reproduction from fishing weights, both from the long-term study of lead poisoning among Mute Swans on the River Thames (Birkhead 1983; Birkhead & Perrins 1985). Lead can impact on the chance of breeding successfully; *e.g.* female Mute Swans with lead levels of > 200 μg dL^{-1} showed low probabilities of producing cygnets and surviving to the next breeding season, relative to individuals with lower blood lead concentrations, although these data were not analysed statistically (Birkhead 1983). Swan breeding success typically increases with prior breeding experience (Rees *et al.* 1996; Wood *et al.* 2016), and so the loss of individuals with breeding experience would be expected to reduce the breeding success of the entire population, although more research is needed to confirm this link.

Available evidence indicates that lead fishing weights have an impact on Mute Swan productivity via increased pre- and post-fledging mortality of juveniles (Birkhead &

Perrins 1985). Comparing stretches of the River Thames catchment with high and low levels of lead poisoning, Birkhead & Perrins (1985) found similar mean clutch sizes (7.4 *vs.* 6.8 eggs per pair) and hatching rates (4.9 *vs.* 4.8 cygnets hatched per pair) between the two areas. Swans in the high lead area, however, had significantly greater cygnet mortality for birds aged between 10–20 weeks, leading to lower fledging success (3.8 *vs.* 2.1 cygnets per pair), and there was significantly greater mortality of immature individuals aged between 1–3 years (1.4 *vs.* 0.8 fledged cygnets alive after 3 years). Cygnet mortality was there was significantly higher on the lower Thames where blood lead levels were highest (Birkhead 1983). Hence, areas with higher levels of lead poisoning are associated with lower recruitment into the breeding population.

Impacts on mortality

We found 20 estimates of the numbers of swan deaths caused by lead fishing weights (Table 3), including two reporting deaths of Bewick's Swans. *Post-mortem* examinations of swans wintering in eastern England found lead fishing weights in 7 of 25 Bewick's Swans (12%, Owen & Cadbury 1975), whilst Evans *et al.* (1973) mentioned that five out of an unspecified total number of dead Bewick's Swans were found, during *post-mortem* examinations, to have lead fishing weights within their gizzards. The mean percentage of Mute Swans killed due to lead fishing weights was 41.4%, from 18 estimates from the literature (Table 3), ranging from 0.4% in the Netherlands (Esselink & Beekman 1991) up to 70.0% in the UK (French 1984).

Aside from studies reporting lead fishing weights within the gizzards of dead swans (*e.g.* Owen & Cadbury 1975), further evidence of the link between mortality and lead poisoning due to fishing weights is shown by the close seasonal timing of lead poisoning events and the recreational fishing season. Systematic searches along the River Thames found that 57% of swans had died of lead poisoning, with greatest numbers of dead swans recovered during the peak recreational fishing season between July–October (Birkhead 1982). Whilst numbers of swans dying of lead poisoning correlated with the peak fishing season, the numbers of swans dying from other causes showed no such pattern, further suggesting that seasonal mortality patterns were linked to lead fishing weights (Birkhead 1982). Similarly, Sears (1988) showed that Mute Swan mortality due to lead poisoning reached a minimum outside of the fishing season. Swan mortality has also been shown to be linked to the amount of accumulated lead fishing weights; Sears (1988) found that across the different sections of the River Thames the percentage of Mute Swans that died of lead poisoning was positively correlated with the number of lead sinkers found within the top 2 cm of river sediment.

It is clear from our review that lead poisoning attributable to the birds ingesting fishing weights has been a key source of mortality for swans in Europe. The mechanisms through which lead kills swans and other animals are well understood; lead is a nonspecific metabolic poison that affects the nervous, vascular, renal and reproductive systems of animals (Franson & Pain 2011). Prior to death, poisoned

Table 3. Swan lead-induced mortality data reported from sites where recreational fishing occurs.

Species	Country	Years	Total no. swans examined	No. deaths from lead	% deaths due to lead	Reference
Bewick's Swan	UK	1969–1975	25	3	12.0	Owen & Cadbury (1975)
Bewick's Swan	UK	1970–1973	–	5	–	Evans *et al.* (1973)
Mute Swan	Ireland	1984–1987	101	69	68.3	O'Halloran *et al.* (1991)
Mute Swan	UK	1951–1989	183	39	21.3	Brown *et al.* (1992)
Mute Swan	UK	1970–1973	8	3	37.5	Owen & Cadbury (1975)
Mute Swan	UK	1971–1986	74	25	33.8	Wood *et al.* (2019)
Mute Swan	UK	1973–1980	327	164	50.2	Nature Conservancy Council (1981)
Mute Swan	UK	1975–1978	66	36	54.5	Hardman & Cooper (1980)
Mute Swan	UK	1979–1981	94	54	57.0	Birkhead (1982)
Mute Swan	UK	1979–1981	89	57	64.0	Birkhead & Perrins (1985)
Mute Swan	UK	1981–1982	320	224	70.0	French (1984)
Mute Swan	UK	1980–1981	210	105	50.0	Sears & Hunt (1991)
Mute Swan	UK	1983–1984	288	115	39.9	Sears & Hunt (1991)
Mute Swan	UK	1983–1985	166	66	39.8	Sears (1988)
Mute Swan	UK	1985–1986	241	93	38.6	Sears & Hunt (1991)
Mute Swan	UK	1987–1988	236	70	29.7	Sears & Hunt (1991)
Mute Swan	UK	1987–2014	584	36	6.2	Wood *et al.* (2019)
Mute Swan	UK	1995–1996	41	11	26.8	Pennycott (1999)
Mute Swan	UK	1996	14	8	57.1	Pennycott (1998)
Mute Swan	The Netherlands	1979–1989	227	1	0.4	Esselink & Beekman (1991)

The symbol – indicates no data reported.

individuals show symptoms such as a loss of fat reserves and reduced body condition, lethargy, anaemia and muscle weakness (Haig *et al.* 2014). Mute Swans that died from lead poisoning had statistically significantly lower body masses (mean ± s.e. = 5.7 ± 0.2 kg) than those that died from other causes (8.9 ± 0.4 kg; Birkhead 1982). Lead poisoned swans had lower body, gizzard, liver and spleen mass, relative to unpoisoned birds, although the statistical significance of these differences was not determined (Simpson *et al.* 1979).

Population-level impacts

Six studies provided information on population-level effects of lead fishing weights on swans (Appendix 1), with all reporting that the widespread use of lead fishing weights in Great Britain had resulted in local and/or national declines in Mute Swan numbers (Hardman & Cooper 1980; Birkhead & Perrins 1985; Ogilvie 1986; Thomas *et al.* 1987; Kirby *et al.* 1994; Wood *et al.* 2019). Evidence of local population declines was found in lowland regions where recreational fishing was common and hence lead use was higher (Ogilvie 1986). Numbers of Mute Swans on the River Avon during the 1970s fell sharply, at a time when lead poisoning was the major cause of death among swans in this area (Hardman & Cooper 1980). Similarly, Birkhead & Perrins (1985) using life table analysis, demonstrated that Mute Swan numbers on the lower River Thames had ceased to be self-supporting, in part due to the impacts of poisoning from lead fishing weights.

Mute Swans numbered an estimated 18,750 in Great Britain in the spring census of 1983 (Thomas *et al.* 1987), a decline of *c.* 8% relative to that census carried out in winter 1955/56 (Ogilvie 1986), whereas numbers would have been expected to have increased during this period in common with many other species of waterfowl (Madsen 1991; Rees *et al.* 2019). The greatest declines were reported from the regions of lowland Great Britain where coarse fishing predominantly occurred, whilst in areas where fly fishing predominated and hence few lead weights were used, Mute Swan numbers increased (Thomas *et al.* 1987; Kirby *et al.* 1994). Long-term changes in Mute Swan abundance in Great Britain between 1974/75 and 2012/13 were best explained (R^2 = 82%) by the change in the legal status of lead fishing weights, rather than arable food resources, winter temperatures or river water quality; the total population size approximately doubled following the regulations introduced in 1987 (Wood *et al.* 2019).

Regulations

The widespread and substantial impacts of lead fishing weights on swans and other taxa have led to concerted efforts by environmental organisations for regulations to reduce lead lost into the environment (UNEP–AEWA 2011). Given the widespread impacts on swans, it is perhaps unsurprising that swan conservationists have often been at the forefront of such calls for regulations; for example, the Third International Swan Symposium, held in Oxford (UK) in 1989, recommended "*that lead be replaced by non-toxic alternatives for both shooting and angling as soon as possible; and that regulatory authorities monitor progress and take necessary steps to ensure total* *elimination of these uses of lead, worldwide*" (Moser 1991). Similar actions were urged in the recommendations of the Fourth International Swan Symposium, held in Virginia (USA) in 2001 (Rees *et al.* 2002). Here, we synthesise information on current and proposed future regulations on lead fishing weights, including both statutory and voluntary bans.

Current regulations

There are currently legal restrictions on lead fishing weights in just two countries within Europe: Denmark and the United Kingdom (Table 4; Fig. 4). In 2002, Denmark introduced a ban on the import and sale of all sizes of lead fishing weights (Michael 2006; UNEP–AEWA 2011). Within the United Kingdom, there have been restrictions on the use, sale and import of lead fishing weights since 1987; the Control of Pollution (Anglers' Lead Weights) Regulations 1986, banned the import and supply of lead fishing weights, whilst the use of lead weights in England and Wales was prohibited by Regional Water Authorities and the Fisheries Byelaw amendment to the 1975 Salmon and Freshwater Fisheries Act (Sears & Hunt 1991; Perrins *et al.* 2003). However, these regulations only applied to weights within the size range 0.06–28.35 g, which were considered to be the most likely to be ingested and hence the greatest risk to swans.

Our review found no studies on the efficacy of the regulations in Denmark, but several studies have quantified the effects of the UK regulations. Sears & Hunt (1991) found that blood lead concentrations among Mute Swans in the River Thames catchment fell from a median of 127 $\mu g\ dL^{-1}$ in 1984 to 22 $\mu g\ dL^{-1}$ in 1987, whilst cases of lead poisoning among rescued swans declined from a peak of 59% in 1984 to 15% in 1988. The proportion of River Thames Mute Swans found to have died from lead poisoning also declined, from 0.50 in 1980/81 to 0.30 in 1987/88 (Sears & Hunt 1991). Furthermore, Wood *et al.* (2019) demonstrated that, following the implementation of the regulations on lead fishing weights in January 1987, the national proportion of Mute Swans killed by lead poisoning fell

Table 4. Current regulations on lead fishing weights within Europe.

Country	Year introduced	Import banned?	Sale banned?	Use banned?	Caveats
Belgium	2015	No	No	Voluntary	Voluntary; marine focus
Denmark	2002	Yes	Yes	No	–
Sweden	Mid-1990s	No	No	Voluntary	Voluntary; certain rivers only
The Netherlands	2018	No	No	Voluntary	Voluntary
United Kingdom	1987	Yes	Yes	Yes	0.06–28.35 g only

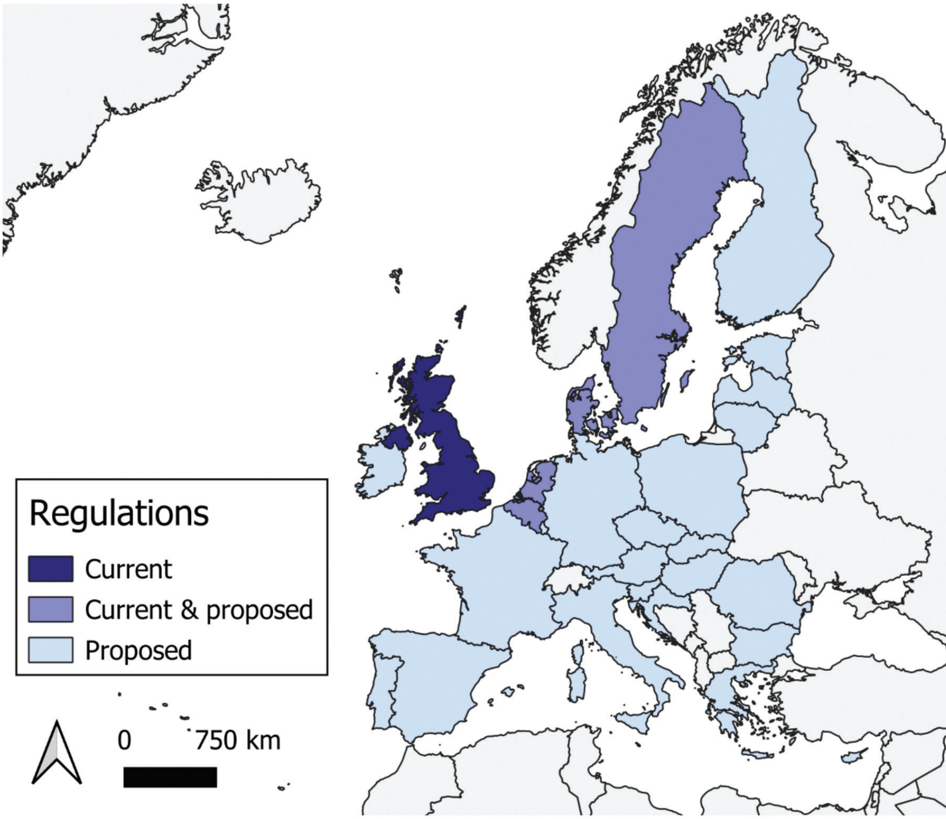

Figure 4. A map showing the status of regulations on lead fishing weights across Europe.

from 0.34 to 0.06, whereas the total Mute Swan population size approximately doubled, reaching *c*. 32,000 individuals by 2002 and stabilising thereafter.

Good compliance among anglers was critical to ensure that the lead regulations in the UK succeeded in allowing the Mute Swan population to recover (Perrins *et al*. 2002, 2003). Evidence that most anglers switched to non-toxic alternatives to lead weights after the regulations has been reported by studies which examined the equipment used by anglers. Fishing rigs removed from swans that had become entangled between 1996 and 2000 found that only 34 of 837 rigs (4.1%) contained illegal lead weights (Perrins *et al*. 2002), whilst a survey of 60 anglers in 2000 found only one individual (1.7%) using illegal lead weights (Perrins *et al*. 2003).

While the Mute Swan population in Great Britain showed a substantial recovery, the issue of lead poisoning has not been eliminated entirely, as after the regulations were introduced the proportion of Mute Swans that died of lead poisoning was found to be low (0.06) but not zero (Wood *et al*. 2019). Despite post-regulation reductions in blood lead concentrations (*e.g.* Sears & Hunt 1991), Perrins *et al.* (2003) and Kelly & Kelly (2004) reported that > 60% and 74% of Mute Swans, respectively, had elevated blood lead concentrations. Lead poisoning of Mute Swans in the UK after 1987 may be due to the sizes of lead fishing weights that can still be imported, sold and used legally (*e.g.* weights > 0.06 g or < 28.35 g); for example, Pennycott (1998) found that six out of eight lead poisoned Mute Swans subject to *post-mortem* examination had ingested large lead fishing weights. Alternative sources could include lead deposited before regulation which has persisted in sediments, non-compliance with the regulations, as well as lead from sources other than fishing weights, such as ammunition (Perrins *et al.* 2003; Newth *et al.* 2013). The extension of the existing regulations to cover all sizes of lead fishing weights would help to further reduce the problem of lead poisoning among swans.

In addition to the legal restrictions in force in Denmark and the United Kingdom, voluntary initiatives to phase-out lead fishing weights are in place in three other countries: Belgium, the Netherlands and Sweden (Table 4; Fig. 4). Belgium has had a voluntary scheme to promote non-toxic alternatives to lead among anglers since 2015, implemented as part of the EU Marine Strategy Framework Directive (2008/56/EC); however, this scheme focuses on sea fishing and so only swans using coastal habitats are likely to benefit (European Chemicals Agency 2021). Voluntary efforts to limit the use of lead fishing weights by anglers have existed in the Netherlands since 2018, with the aim of a voluntary phase-out by 2027 (European Chemicals Agency 2021). Voluntary restrictions on lead use by anglers on certain rivers in Sweden have existed since the mid-1990s, alongside voluntary efforts to phase-out lead use (UNEP–AEWA 2011). However, an assessment conducted in 2007 concluded that the quantity of lead used by Swedish anglers between 1995–2005 had not decreased, suggesting that this voluntary process has made little progress (UNEP–AEWA 2011).

Regulations in progress

In 2019 the European Commission asked the European Chemicals Agency (ECHA) to assess the risks posed by the use of lead in ammunition and in fishing, and to propose any necessary restrictions, under the auspices of the European Union's regulations on Registration, Evaluation, Authorisation and Restriction of Chemicals (EU REACH). Following a call for evidence, two major restrictions on lead fishing weights have been suggested under the draft EU REACH proposals, as part of a broader set of restrictions on lead. The first is a ban on the sale and use of lead sinkers and lures following transition periods of 3 years for ≤ 50 g weights and 5 years for > 50 g weights. The second proposal is an immediate ban on the use of lead sinkers when the sinker is deliberately dropped into water, for example those use in lead drop off techniques. The EU REACH proposals would regulate lead fishing weights in all 27 European Union member states: Austria, Belgium, Bulgaria, Croatia, Republic of Cyprus, Czechia, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, the Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain and Sweden (Fig. 4). Consultations on these proposals occurred in 2022 and 2023, and we currently await publication of the final proposals, covering fishing and ammunition, that will be put before the European Parliament for a vote among law makers. Certain non-EU countries such as Norway, Iceland and Lichtenstein have previously adopted EU regulations on lead, such as the recent EU ban on the use of lead shotgun ammunition over wetlands; however, it is not clear at this stage whether any of these countries would adopt the EU REACH regulations. Moreover, whilst the EU cannot directly regulate the use of fishing weights in non-member states, there could be indirect effects. For example, if the proposed EU REACH regulations were to cause international fishing tackle manufacturers to move away from the use of lead in their products, for instance by reducing the market for lead fishing gear and hence making the production of such items less profitable, then the regulations could indirectly affect the use of lead outside of the EU.

Discussion

Deposition of lead and its impacts on swans

A growing body of evidence shows that lead fishing weights are lost and discarded into the environment (Forbes 1986; Jacks *et al.* 2001), where they are ingested by swans (Sears & Hunt 1991), causing elevated concentrations in blood and organs such as livers and kidneys (Simpson *et al.* 1979; O'Connell *et al.* 2009), leading to a range of sublethal impacts on individuals (reviewed by Grade *et al.* 2019), reducing breeding success (Birkhead & Perrins 1985), causing deaths (Birkhead 1982; Wood *et al.* 2019) and ultimately limiting the population size of swans (Wood *et al.* 2019).

Our review revealed evidence of impacts of lead fishing weights on all three swan species native to Europe: Bewick's Swans, Mute Swans and Whooper Swans. Most of the evidence found pertained to Mute Swans, which accounted for 19 of the 20 estimates of blood lead concentrations, 18 of 20 estimates of mortality, and all of the studies on reproduction and population-level impacts. As such, our findings concur with previous analyses (*e.g.* Blus 1994) that lead fishing weights within Europe impact predominantly on Mute Swans. In contrast, lead poisoning of Whooper Swans and Bewick's Swans is mostly, but not entirely, caused by lead ammunition rather than fishing weights (Newth *et al.* 2013). Across much of their range within Europe, Mute Swans use aquatic habitats to a greater extent than either Bewick's Swans or Whooper Swans, and it is in these aquatic habitats that recreational fishing occurs (Kirby *et al.* 1994). Peak recreational fishing activity takes place during summer (Birkhead 1983; Sears & Hunt 1991), when Bewick's Swans and Whooper Swans are present only at high-latitude sites where relatively little recreational fishing occurs, unlike Mute Swans which are resident all year across much of the continent (Lehikoinen 2020; Wood & Włodarczyk 2020). The greater consequence for Mute Swans of the availability of lead fishing weights in the environment therefore may reflect the more extensive spatial and temporal overlap of Mute Swan foraging habitat with recreational fishing.

Despite the evidence for the effects of ingesting lead fishing weights for swans and other waterbirds, population trends of both Mute and Whooper Swans within Europe are currently stable or increasing (Laubek *et al.* 2019; Rees *et al.* 2019; Wood *et al.* 2019; Brides *et al.* 2021), with both species showing expanding ranges within Europe (Lehikoinen 2020; Wood & Włodarczyk 2020). In particular, the Mute Swan population within Great Britain recovered following the 1987 regulations on lead fishing weights and is now stable (Wood *et al.* 2019). To date, however, there has been no research that has addressed how swan population trends and distribution in Europe would have changed in the absence of lead poisoning due to fishing weights. Previous research has demonstrated that for some species mortality due to lead poisoning may be insufficient to cause a population to switch from growth to decline, but that population will grow more slowly than it would in the absence of lead poisoning (Meyer *et al.* 2016). Aside from population-level effects, the continued use of lead fishing weights in Europe has implications for the welfare of individual swans. The findings of our review demonstrate that lead concentrations in swan tissues and organs are consistently elevated above safe background levels, and so sublethal and lethal effects of lead poisoning on swans are likely.

In contrast to Mute Swans and Whooper Swans, the number of Bewick's Swans wintering in Europe has declined since 1995 (Beekman *et al.* 2019); however, expert opinion (Nagy *et al.* 2012) together with the limited evidence of impacts identified in this review, suggest that lead fishing weights have probably not played a major role in this decline. Yet, given that lead fishing weights are a known cause of mortality for Bewick's Swans (Evans *et al.* 1973; Owen *et al.* 1975), the continued use of lead fishing weights across much of Europe will only hinder conservation efforts to restore this species.

While our review identified a considerable evidence base which future conservation action and policy could draw upon, some key knowledge gaps remain. Our review shows clear geographical and temporal biases in the currently available literature. A total of 28 of the 45 relevant papers identified in our review were conducted in the UK, whereas parts of central, southern and eastern Europe were not well-represented in the literature (Fig. 3). Similarly, 25 of the 45 relevant papers were published during the 1980s and 1990s (Fig. 2). Despite these biases, however, there is arguably sufficient information to cause concern, that the high rates of lead deposition and the resulting impacts on swans that were documented in the UK and Ireland, are also occurring in other parts of Europe. Moreover, the scale and magnitude of the impacts found for the UK should be sufficient to warrant comprehensive legal restrictions on the use of lead fishing weights, as swan conservationists have argued previously (Moser 1991). Similar calls to action have already been made by the African-Eurasian Waterbird Agreement (UNEP–AEWA 2011).

The ultimate fates of lead fishing weights lost or discarded into the environment also remains a key knowledge gap. During our review we did not find any studies that assessed the rates at which lead fishing weights were buried within sediments, ingested by swans or transported downstream by water flows (in particular seasonal high spate flows). The low (*c.* 1%) annual rate at which lead weights dissolve in aquatic environments (Jacks *et al.* 2001) suggests that, in the absence of removal mechanisms such as burial or removal by flow, lead fishing weights are likely to persist within the environment for long periods of time. In the absence of studies on lead fishing weights, research on the fates of lead ammunition may be informative, given the physical similarities between shotgun ammunition and sinkers. For example, an earlier study by Kanstrup *et al.* (2020) found long-term persistence of lead ammunition which, whilst generally lighter than lead fishing weights (so may take longer to sink into the sediment), were still available to waterfowl 33 years after a ban on the use of lead ammunition over wetlands in Denmark. The indirect evidence for fishing weights is somewhat mixed; the correlation between the fishing season and incidences of lead poisoning (Birkhead 1983), together with the rapid benefits to swans documented where regulations have been introduced (Sears & Hunt 1991; Wood *et al.* 2019), suggests that the availability of lead fishing weights to swans may attenuate more rapidly in some habitats. However, the continued elevated blood lead concentrations (Perrins *et al.* 2003; Martin 2016) and mortality due to lead poisoning after regulation (Wood *et al.* 2019), could be in part due to the persistence of lead fishing weights. Assessments similar to Kanstrup *et al.* (2020) could improve our knowledge on the fates of fishing weights and the duration that these will remain an environmental threat.

While there is a growing body of evidence which demonstrates the ecological impacts of lead fishing weights, as illustrated by the findings of our review, to date there has been a lack of research into the attitudes of anglers towards the use of lead and its impacts on swans and the environment, in contrast with the situation for lead ammunition (*e.g.* Newth *et al.* 2019). Such research is valuable because of the sociological and political dimensions of lead regulations (Cromie *et al.* 2015). For example, what factors influence anglers to switch to non-toxic weights, and to either comply or resist regulations that restrict the use of lead fishing weights? To help design and implement future policy, research is needed into the motivations and barriers associated with adopting non-lead fishing tackle, as has been conducted outside of Europe for instance in the USA (Leszek 2015).

The literature which could be included in our review was limited in some cases by a lack of information in the original studies on the source(s) of lead that caused poisoning amongst swans, or whether recreational fishing occurred at the study site. This is a particular problem for sites at which there are multiple possible sources of lead deposition, such as fishing and shooting. This issue could be addressed where studies state whether key potential sources of lead contamination such as recreational fishing or shooting are known to occur in the study area. Studies could also combine field surveys of study sites for the presence of lead fishing weights, *post-mortem* analyses of dead swans to quantify incidences of lead weight ingestion, and isotopic analyses to identify the likely source(s) of lead poisoning (*e.g.* Scheuhammer & Templeton 1998; Binkowski *et al.* 2016). In addition, for migratory populations that use larger numbers of sites, telemetry studies could be used to infer where lead poisoning occurs (Ely & Franson 2014).

Previous research has argued that monolingual searches can affect the findings of literature reviews if this causes critical publications written in other languages to be missed (Nuñez & Amano 2021). Here, we carried out our searches in Scopus and PubMed in English only, and of the 45 relevant papers from which we extracted information in our review, only one was written in a language other than English (Szabó 1998). However, search tools such as Scopus do index many non-English articles with translated titles and abstracts, and so our searches in English would have identified relevant non-English articles. We also supplemented our review with additional articles based on our knowledge of the literature and the references contained within other key documents, such as the report compiled following the call-for-evidence for the EU REACH restrictions on lead ammunition and fishing weights (European Chemicals Agency 2021). This document cites publications in a number of different European languages, but crucially we found no additional non-English publications to add to our dataset. We note that previous reviews on lead fishing weights (*e.g.* UNEP-AEWA 2011; Grade *et al.* 2019) also found very few papers written in languages other than English.

Conclusion

The future of lead fishing weights in Europe remains uncertain. Already banned in two countries, with some voluntary restrictions in three others (Table 4), the new regulations proposed following the recent (2022–2023) EU REACH consultation process would see lead fishing weights banned from much of Europe. At this stage, however, it remains unclear whether these proposed restrictions will eventually be adopted into law. Even if the EU REACH restrictions do succeed in banning the use of lead fishing weights within the European Union, previously deposited lead will continue to threaten swans and other wildlife for many years to come (Jacks *et al.* 2001). Evidence from the UK, however, shows that where compliance by anglers is high, swans can recover quickly following the introduction of restrictions on lead fishing weights (Sears & Hunt 1991; Wood *et al.* 2019). Despite some progress in reducing the impacts of lead fishing weights on swans in a few countries such as Denmark and the UK, the available data on swan mortality and elevated Pb concentrations in tissue and organs demonstrate that lead poisoning due to fishing weights remains a threat to swans within Europe.

Data availability

The data extracted from the original papers are presented within the manuscript. Additional data and the analytical R code associated with our study can be accessed via the following DOI: https://doi.org/10.6084/m9.figshare.22321030

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Appendix 1. The numbers of studies identified in each step of our systematic review. The number of unique references is fewer than the sum of the references for each topic because some papers were relevant to more than one topic.

Topic	Search terms	No. papers via Scopus	No. papers via PubMed	No. additional papers	Total unique papers	No. papers after title screening	No. papers after abstract screening	No. papers after full text screening
Deposition	(lead OR Pb) AND (fish* OR angl*) AND (weight* OR sinker*) AND (deposition)	100	34	10	130	16	14	11
Breeding	(Cygnus OR swan) AND (lead OR Pb) AND (breed* OR fecund* OR product* OR reproduct*)	84	6	0	83	11	4	2
Tissues/ organs	(Cygnus OR swan) AND (lead OR Pb) AND (blood OR kidney OR liver OR tissue OR organ)	228	123	4	292	56	38	24
Mortality	(Cygnus OR swan) AND (lead OR Pb) AND (survival OR mortality OR death)	119	59	6	155	35	26	15
Population	(Cygnus OR swan) AND (lead OR Pb) AND (population)	75	39	2	98	25	11	6
All unique references	–	–	–	–	**605**	**99**	**64**	**45**