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Lead fishing sinkers and jigs in Canada: Review of their use patterns and toxic impacts on wildlife

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Abstract

More than 5 million Canadians take part in recreational angling each year, spending over 50 million days fishing on open water. Recreational anglers contribute to environmental lead deposition through the loss of lead fishing sinkers and jigs. East year lost or discarded fishing sinkers and jigs amounting to an estimated 500 tonnes of lead, and representing up to 14% of all nonrecoverable lead releases in Canada, are deposited in the Canadian environment. Wildlife, primarily piscivorous birds and other waterbirds, ingest fishing sinkers and jigs during feeding, when they either mistake the sinkers and jigs for food items or grit or consume lost bait fish with the line and weight still attached. Lead fishing weights that weigh less than 50 g and are smaller than 2 cm in any dimension are generally the size found to be ingested by wildlife. Ingestion of a single lead sinker or lead-headed jig, representing up to several grams of lead, is sufficient to expose a loon or other bird to a lethal dose of lead. Lead sinker and jig ingestion has been documented in 10 different wildlife species in Canada. In the United States, ingestion of lead sinkers and jigs by 23 species of wildlife, including loons, swans, other waterfowl, cranes, pelicans, and cormorants, has been documented. Evidence gathered to date indicates that lead sinker and jig ingestion is the only significant source of elevated lead exposure and lead toxicity for Common Loons *Gavia immer* and the single most important cause of death reported for adult Common Loons in eastern Canada and the United States, frequently exceeding deaths associated with entanglement in fishing gear, trauma, disease, and other causes of mortality.

Except for a few local or regional instances, available data indicate that Common Loon populations are stable or increasing through most of their Canadian range. There is currently insufficient information to answer the question of whether mortality through lead sinker poisoning may be having population-level effects on loons anywhere in Canada or to estimate with confidence the minimum frequency of poisoning that, combined with the effects of other environmental stressors, would be required to significantly affect population dynamics. The most critical areas of new knowledge that are required to enable confident estimates of the population effects of lead sinker poisoning in loons are accurate life history data using individually marked birds to derive important population parameters for local or regional loon populations in Canada; DNA analyses to better define "populations"; a better understanding of the interactions of multiple environmental stressors that may influence population dynamics; and incorporation of these multiple stressors into a large-scale spatial analysis using geographic information systems. Such research would be expensive and time-consuming, requiring long-term monitoring of substantial numbers of banded individuals from several selected populations.

There are numerous viable alternative materials for producing fishing sinkers and jigs, including tin, steel, bismuth, tungsten, rubber, ceramic, and clay. Tin, steel, and bismuth sinkers and bismuth jigs are the most common commercially available alternatives in Canada. Many of the available alternative products are currently more expensive than lead; however, switching to these products is anticipated to increase the average angler's total yearly expenses by less than 1% (~$2.00). Nevertheless, the continued availability of (cheaper) lead products has made it difficult for the manufacture and sale of nontoxic alternatives to achieve commercial viability.

Some limited regulatory actions have been taken to reduce the use of lead sinkers and jigs both in Canada and elsewhere. In 1987, Britain banned the use of lead fishing sinkers weighing less than 28.35 g. The United States has banned the use of lead sinkers and jigs in three National Wildlife Refuges and in Yellowstone National Park and is currently considering further action. New Hampshire, Maine, and New York have ratified statewide regulations prohibiting the use of lead sinkers beginning in 2000, 2002, and 2004, respectively. Environment Canada and Parks Canada prohibited the possession of lead fishing sinkers or lead jigs weighing less than 50 g by anglers fishing in National Wildlife Areas and National Parks under the *Canada Wildlife Act* and the *National Parks Act*, respectively, in 1997. However, these latter two regulations are of limited geographic scope, covering <3% of Canada's land mass, and they affect only about 50 000 (<1%) of the estimated 5.5 million recreational anglers in Canada. Currently, the majority of recreational anglers continue to use lead sinkers and jigs.

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Introduction

Recreational anglers use various sizes and shapes of lead sinkers and jigs to sink hooks, lures, or bait while fishing. Ingestion of lost sinkers and jigs occurs during feeding, when waterbirds mistake them for food items such as seeds or shelled invertebrates such as small snails or clams. Fish-eating wildlife, particularly loons, most often ingest lead sinkers when they consume lost bait fish with the hook, line, and sinker still attached.

Ingestion of small lead fishing weights has resulted in the lead poisoning and death of waterbirds in Britain, the United States, and Canada. Evidence presented in a previous assessment (Scheuhammer and Norris 1995) indicated that the use of small lead fishing sinkers and jigs presented a risk of toxicity and mortality for Common Loons *Gavia immer*, particularly in areas of intense freshwater sport angling.

In response to the evidence presented in Scheuhammer and Norris (1995), Environment Canada, in partnership with Parks Canada (Department of Canadian Heritage), initiated regulatory actions to prohibit the possession and use of small lead sinkers and jigs by recreational anglers fishing in National Wildlife Areas and National Parks beginning in 1997. This parallel regulatory initiative was carried out under the *Canada Wildlife Act* and the *National Parks Act* and targeted lead sinkers and jigs weighing less than 50 g.

Since the initiation of these regulatory actions, the Canadian Wildlife Service (CWS) has continued to assess the effects of lead fishing sinkers and jigs on waterbirds through documentation of cases of mortality due to lead sinker ingestion. The present report updates scientific information presented in Scheuhammer and Norris (1995) and discusses initiatives taken by Canada and other countries to manage the issue of lead sinker ingestion and poisoning in wild birds.

1. Recreational angling in Canada

1.1 Estimating the number of recreational anglers in Canada

Recreational angling in Canada is managed mainly by the federal Department of Fisheries and Oceans (DFO) and its provincial and territorial counterparts. In 1975, these agencies initiated the Survey of Recreational Fishing in Canada. Included in this nationally coordinated survey were estimates of the total number of recreational anglers across Canada, the level of angling effort in various regions, and the social and economic importance of this activity across Canada. The survey is conducted every five years.

The Nature Survey, formerly known as The Importance of Wildlife to Canadians or the Wildlife Survey, was initiated in 1981 by Environment Canada in cooperation with federal, provincial, and territorial governments to obtain information about Canadians' recreational interests in wildlife and nature (Filion et al. 1993; DuWors et al. 1999). Starting with the 1991 survey, Environment Canada began incorporating questions pertaining to participation in recreational angling activities. Together, the DFO and Environment Canada surveys provide information on recreational angling across Canada from 1975 to 1996. Estimates of subsistence angling are not included in these surveys.

DFO surveys indicate that, between 1975 and 1995, the number of licensed anglers in Canada ranged from 4.64 to 5.18 million (DFO 1994, 1997, unpubl. data), whereas the Environment Canada surveys estimated 5.4 million anglers in 1991 and 4.2 million in 1996 (Filion et al. 1993; DuWors et al. 1999) (Table 1). These figures are slightly higher than estimates by Scheuhammer and Norris (1995) that were based on adult resident licensed anglers only. However, an estimated 1.06–1.65 million unlicensed individuals, including children and adults not required to purchase a licence, also participate in recreational angling in Canada. Unlicensed anglers represent about 17–26% of total angling participation (DFO 1994, 1997). Overall, about 5.5 million people fish in Canada every year, or approximately 1 in 5 Canadians (DFO 1997; DuWors et al. 1999).

1.2 Geographic distribution of recreational anglers in Canada

The majority of recreational anglers fish within their province or territory of residence. Almost two-thirds of all annual open water recreational angling takes place in Ontario and Quebec (Fig. 1) (DFO 1997). A small proportion of Canadians (10% of all anglers) travel outside their province or territory of residence to fish. In addition, an estimated 800 000 tourists, primarily from the United States, travel to Canada every year to fish, representing about 12% of all anglers in Canada (Filion et al. 1993; DFO 1994, 1997; DuWors et al. 1999).

There is considerable regional variation in total (freshwater plus saltwater) angling effort, ranging from 9 days per angler per year in the Yukon to 24 days in Newfoundland (Fig. 2). Annual freshwater angling effort generally exceeded saltwater effort, ranging from 9 days per participant in the Yukon to 19 days per participant in Nova Scotia (DFO 1994). Fishing in freshwater environments accounts for 85% of all recreational angling effort (61 million days) in Canada, and the majority of this (55.5 million days) is open water activity (rather than ice fishing) (DFO 1997).

1.3 Angling pressure in Canada and the United States

Angling pressure was calculated by determining the total number of days spent angling (fresh plus salt water) in various regions of Canada (based on DFO survey results) and the United States and dividing by the regional area to obtain an estimate of the mean number of angling days per square kilometre per year. Overall, there was a general trend of increasing angling pressure from west to east-central North America, with the highest levels reported in southern Ontario and the northeastern United States.

1.3.1 Canada

In 1995, DFO estimated that 4.6 million licensed Canadian anglers spent 55.5 million days open water fishing, or an average of about 12 days per angler (DFO 1997). An additional 5.5 million days were spent fishing through ice; however, these data are not included in our analysis of angling pressure, because lead poisoning associated with fishing tackle is primarily an open water phenomenon. Subsistence angling was not reported in the DFO surveys.

For the present report, total land area was used in calculating angling pressure in the absence of specific data for the area covered by lakes, rivers, etc. Thus, our estimates of angling pressure may not coincide with creel census or other local angling surveys.

Table 1
Recreational anglers in Canada (adapted from Filion et al. 1993; DFO 1994, 1997; DuWors et al. 1999)

	Recreational anglers in Canada (× 1 000 000)								
	Adult licensed anglers				Children and other unlicensed participants				
Year	Residents	Non-residents	Non-resident non-Canadians	Total licenced anglers	Residents	Non-residents	Non-resident non-Canadians	Total unlicensed anglers	Total number of anglers in Canada
DFO surveys									
1975	3.70	0.98	NA[a]	4.68	1.40	0.25	NA	1.65	6.33
1980	4.20	0.97	NA	5.17	0.85	0.21	NA	1.06	6.23
1985	4.20	0.19	0.79	5.18	1.40	0.05	0.14	1.59	6.77
1990	4.00	0.19	0.77	4.96	NA	NA	NA	NA	4.96[b]
1995	3.68	0.20	0.76	4.64	NA	NA	NA	NA	4.64[b]
Mean	**3.96**	**0.51**	**0.77**	**4.93**	**1.22**	**0.17**	–	**1.43**	**5.79**
1991 Wildlife Survey	5.44		0.86	6.30	NA	NA	NA	NA	6.30
1996 Nature Survey	4.20		0.54	4.74	NA	NA	NA	NA	4.74
Mean	**4.82**		**0.70**	**5.52**	–	–	–	–	**5.52**

[a] NA = data not available.
[b] Data source notes that these estimates are minimum values due to an inability to determine the numbers of unlicensed anglers.

Figure 1
Proportion of the total number of open water recreational angling days spent within various regions of Canada (total = 55.5 million angler days per year)

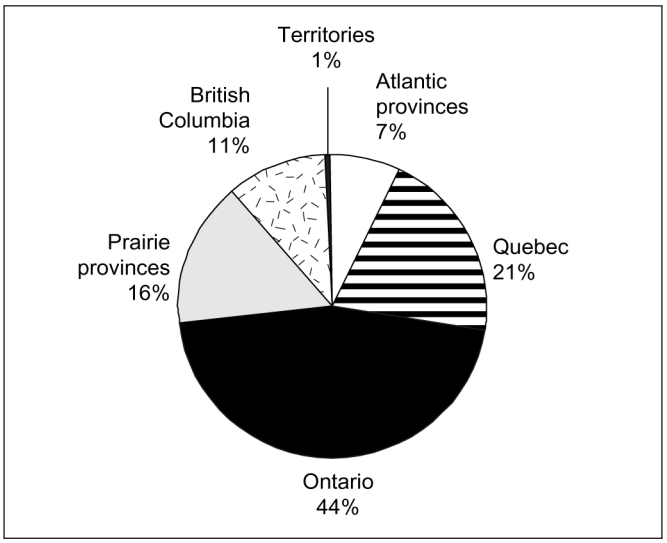
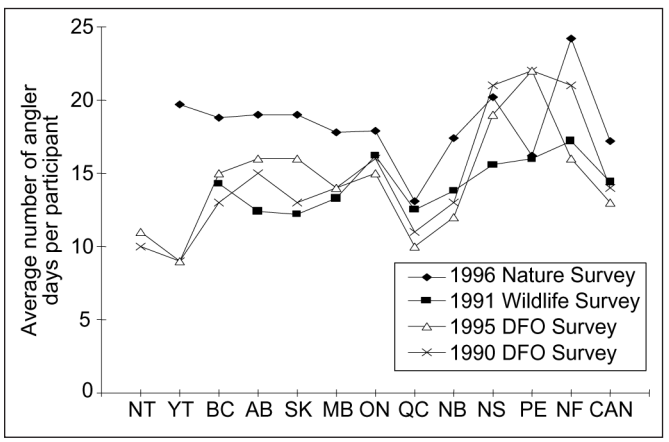


Figure 2
Estimated average annual angling effort per individual angler, by province and nationally (data from Filion et al. 1993; DFO 1994, 1997; DuWors et al. 1999)



Angling pressure reported at the provincial/territorial level ranged from <1 angler day per square kilometre in the Northwest Territories to over 47 angler days per square kilometre in Prince Edward Island (DFO 1997).

Additional information on angling effort was obtained from a 1991 survey in Ontario, which estimated angling effort for eight administrative regions (OMNR 1993) (Fig. 3). Within Ontario, where about 44% of all Canadian angling occurs, the southern regions were subject to the majority of the angling pressure. Angling pressure ranged from 4.2 days per square kilometre in the Northern Region to over 230 days per square kilometre annually in the Central Region. Patterns of angling effort within individual provinces are not currently available for other areas of Canada.

1.3.2 United States

In 1996, an estimated 35 million people 16 years of age or older participated in recreational fishing (exclusive of ice-fishing activity) in the United States, spending over 625 million days fishing, or an average of 18 days per angler (U.S. Department of the Interior 1997). U.S. state angling pressure estimates are considerably greater than those for Canadian provinces and territories, ranging from 6.9 angler days per square kilometre in Nevada and Montana to over 800 angler days per square kilometre in New Jersey (Fig. 4). Although the number of anglers in the United States was stable between 1991 and 1996, the number of angling days rose from 511 million to over 600 million, leading to a substantial increase in angling pressure in some states.

1.4 Summary

- In 1995, 4–6 million Canadian anglers spent 50–60 million days fishing in open water — on average, between 10 and 13 days per angler per year.
- Freshwater angling represents 85% of all recreational angling in Canada; 65% of this occurs in Ontario and Quebec.

Figure 3
Estimated recreational angling pressure in various regions of Ontario in 1991 (OMNR 1993)

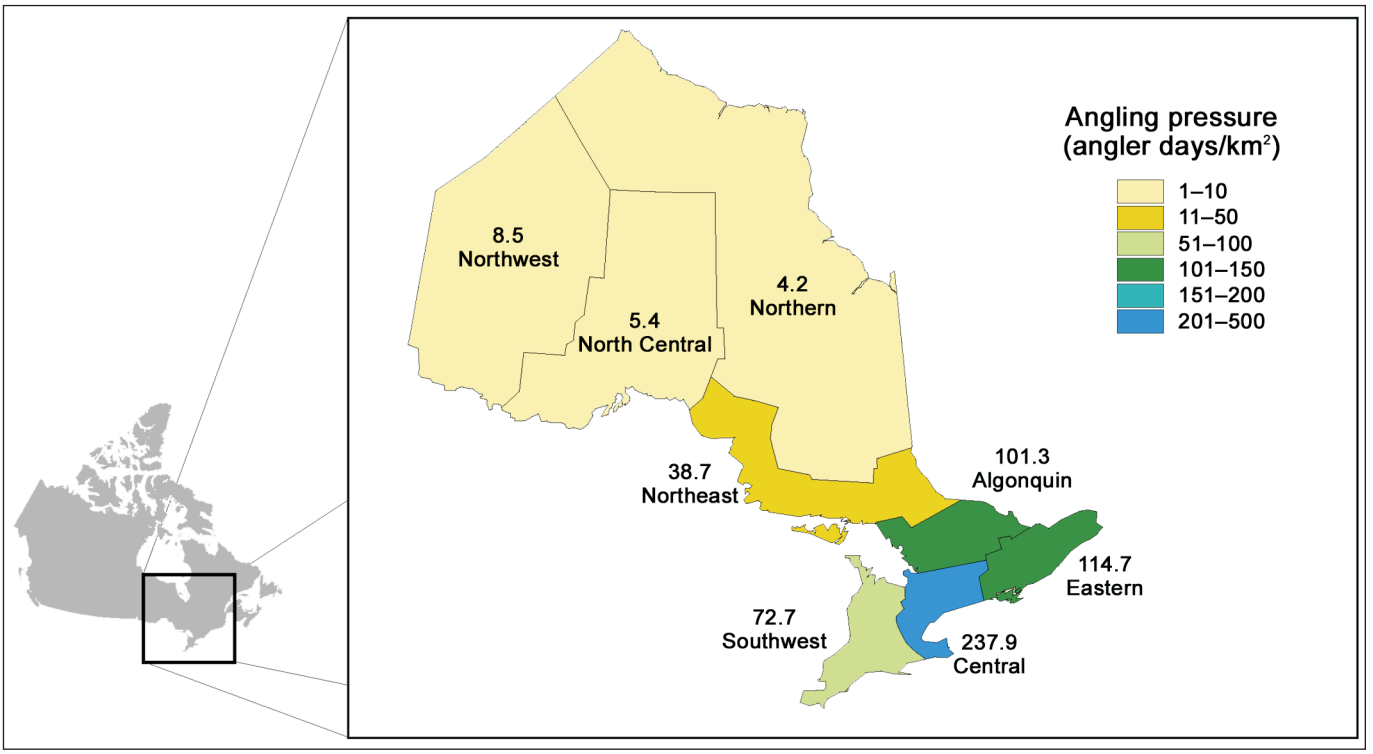
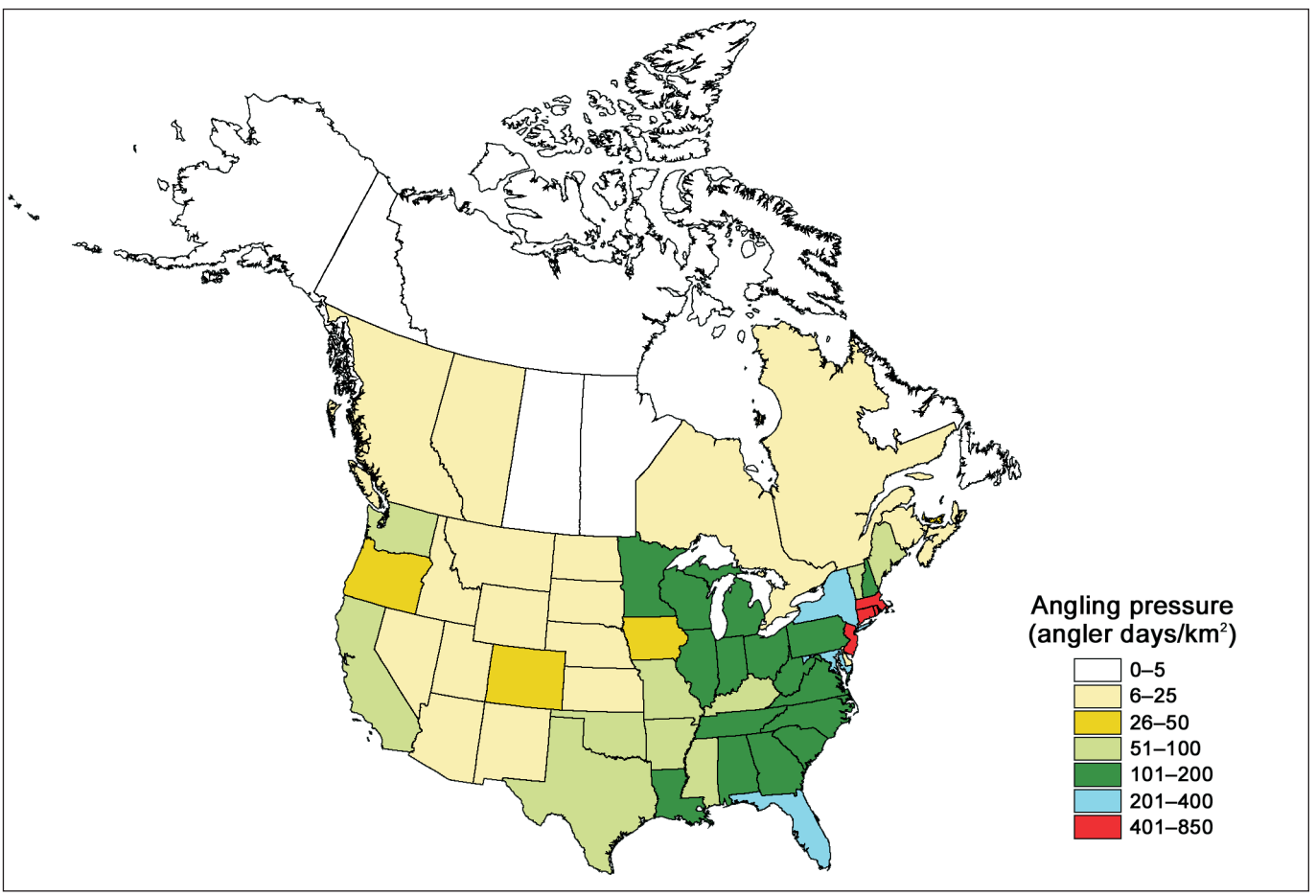


Figure 4
Recreational angling pressure by province in Canada and by state in the United States (data from DFO 1997; U.S. Department of the Interior 1997)



- Angling pressure in Canada ranges from <1 to 47 angler days per square kilometre at the provincial/territorial level and increases to over 230 angler days per square kilometre at the regional level in central Ontario.
- In 1996, 35 million U.S. anglers spent over 625 million days fishing in open water; about 18 days per angler.
- Average angling pressure is greater in the United States than in Canada and ranges from 7 to over 800 angler days per square kilometre at the state level.

2. Estimating the magnitude of lead sinker and jig use

2.1 Market demand

In 1995, Canadian anglers spent $2.5 billion, or an average of about $533 per angler, on goods and services directly related to recreational fishing (DFO 1997), a slightly higher estimate than that reported in the 1991 Wildlife Survey (Filion et al. 1993) and in Scheuhammer and Norris (1995). Over 80% of these expenditures were for food, lodging, and transportation/travel. Fishing supplies including bait, line, and fishing tackle represented 8% ($194 million) of anglers' expenses (Fig. 5), or about $42 per resident angler annually (DFO 1997). Using the estimated number of anglers in Canada (5.5 million) and the average yearly estimate of expenditure per angler on sinkers derived by Scheuhammer and Norris (1995) ($3.25 per year), it is estimated that Canadian anglers spend about $17.9 million per year buying fishing sinkers. Using the average retail cost of sinkers ($0.032 per gram of lead), the mass of lead sold as fishing sinkers annually in Canada is estimated to be about 559 tonnes. An undetermined additional amount of lead is sold in the form of jigs. The majority of this annual purchase of lead is destined to be deposited in the environment, with virtually no chance of recovery or recycling.

Regulations prohibiting the use of lead fishing sinkers and jigs within National Wildlife Areas and National Parks were initiated in 1997. These regulations were estimated to affect about 50 000 anglers and to reduce the use of lead fishing sinkers and jigs by about 4–5 tonnes annually. Local outreach efforts, including collection and exchange programs, combined with efforts to educate anglers on the hazards to wildlife from lead sinkers and jigs, have helped to reduce the demand for lead in some areas (e.g., Great Lakes 2000 Cleanup Fund 1995). However, together, these efforts have resulted in a reduction of only about 1% in the estimated annual purchase and use of lead sinkers and jigs (Fig. 6). Recreational anglers continue to purchase in excess of 500 tonnes of lead annually in the form of lead sinkers and jigs.

Comparatively, in the United States in 1996, anglers spent US$37.8 billion, or about US$1100 per angler, on goods and services directly related to fishing (U.S. Department of the Interior 1997). Over 70% of these expenditures were for food, lodging, transportation, and specialized equipment. Tackle, including hooks, sinkers, swivels, and other items attached to the fishing line, except lures and bait, represented 1% (US$376 million) of anglers' expenses, or about US$11 per angler per year. Using the estimated number of anglers in the United States (35 million) and the U.S. Environmental Protection Agency's (EPA) estimate of average annual expenditure per angler on sinkers (US$2.50 per year), U.S. anglers spend about US$87.5 million annually to buy lead fishing sinkers. Assuming a similar average retail cost for sinkers as determined by Scheuhammer and Norris (1995) for Canada ($0.032 per g of lead) and converting to U.S. currency (US$0.022 per g of lead), the mass of lead sold as fishing sinkers annually in the United States is about 3977 tonnes.

2.2 Import of lead fishing sinkers and jigs

Scheuhammer and Norris (1995) obtained import volume estimates for "fishing sinkers for sportsmen" from the International Trade Division of Statistics Canada for the period 1988–1994. For the present report, we requested information for the period 1988–1998. The "fishing sinkers for sportsmen" classification includes all shapes and sizes of small fishing sinkers used for recreational angling in Canada. This commodity does not include jigs; therefore, import of lead jigs would be in addition to amounts presented below. Reports do not specify "leaded" versus "lead-free" products, so it is possible that some imports may be lead-free, especially in recent years, in response to regulations prohibiting the use of leaded products in the United States, Britain, and Canada. However, based on the lack of broad-scale regulations in North America and the relatively limited availability of lead-free products at the retail level, it is likely that the vast majority of imported products are still being manufactured using lead.

Between 1988 and 1998, the wholesale value of imported sinkers ranged from $0.42 million in 1990 to $3.18 million in 1995 (Statistics Canada 1999). A considerable increase in the annual imports of these products began in 1995. Prior to 1995, the average annual import of sinkers was estimated to be $0.60 million. From 1995 to 1998, the average import rose to an estimated $2.67 million annually. Based on the estimated wholesale sinker value of about $0.027 per g of lead (Scheuhammer and Norris 1995), these values translate to an estimated 22.40 tonnes of lead in the form of lead fishing sinkers annually prior to 1995 and 98.91 tonnes annually after 1995 (Fig. 7).

Canada imported lead sinkers from 20 different countries between 1988 and 1998. The majority of sinkers

Figure 5
Estimated breakdown of the average Canadian angler's annual budget of approximately $600 (DFO 1997)

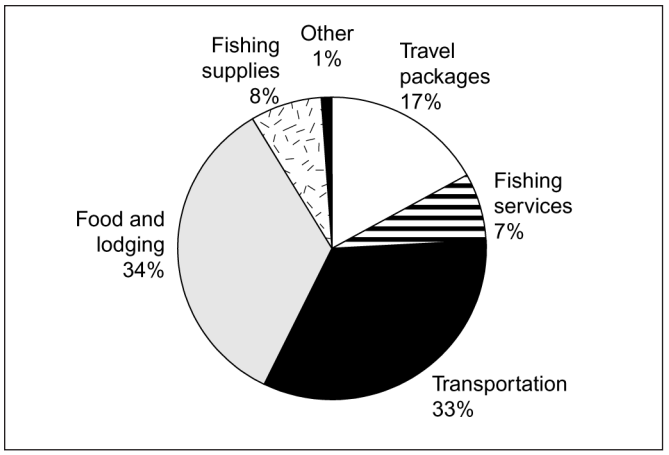
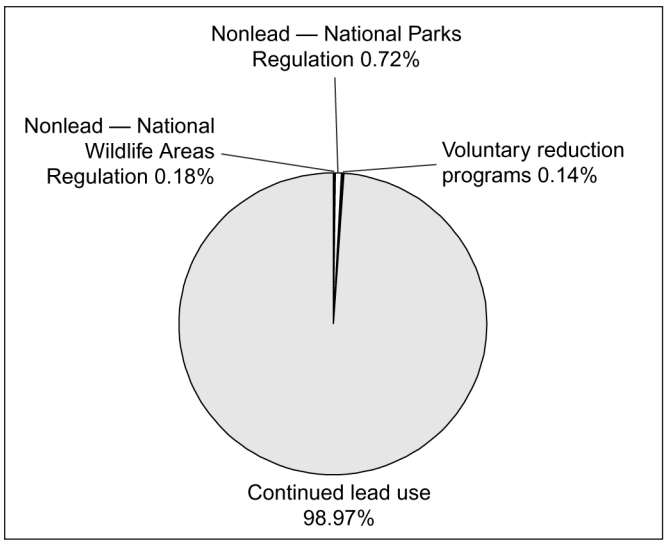


Figure 6
Estimated current proportional use of lead versus nonlead sinkers and jigs in Canada



(65%) imported since 1993 were from Europe, especially Ireland, Finland, and the United Kingdom (Fig. 8). Asian and U.S. producers provided an estimated 20% and 14%, respectively, of all sinkers imported into Canada. Sporadic imports from other countries combined (Mexico, Costa Rica, El Salvador, Sweden, Italy, Germany, Hungary, Egypt, Kenya, Japan, and Hong Kong) comprised 1% of the total import market value in Canada during this time. European imports, primarily from the United Kingdom and Finland, were highest in 1995 (Fig. 7). Annual imports from Ireland have continued to rise since 1995 and accounted for 70% (80 tonnes) of total Canadian imports in 1998. The reasons underlying these changing patterns of sinker import into Canada have not been assessed.

2.3 Domestic production of lead fishing sinkers and jigs

Domestic production of lead fishing sinkers in Canada was previously estimated to be about 40 tonnes

Figure 7
Estimated annual weight of sinkers imported into Canada from major geographical regions (imports from Africa, Mexico, and Central America were negligible: combined total of less than 1 tonne for the decade)

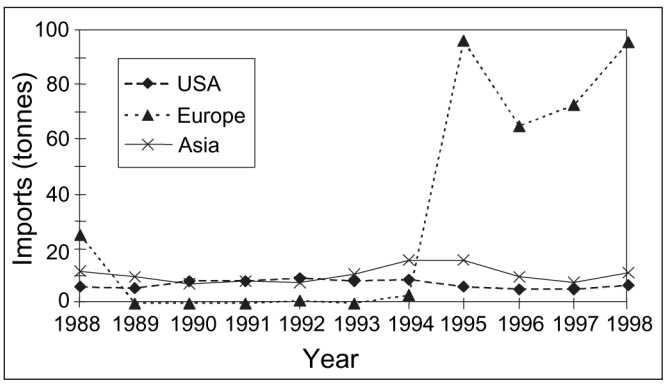
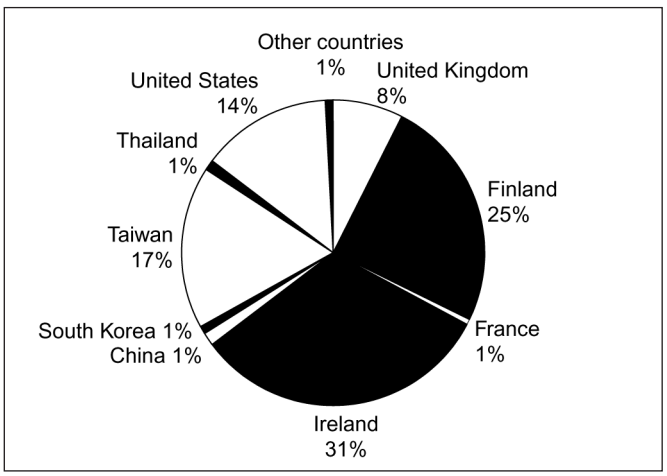


Figure 8
Proportions of total sinker imports (1988–1998) from various world regions (data from Statistics Canada 1999)



annually (Scheuhammer and Norris 1995), with an additional undetermined amount of lead used in the production of lead-headed jigs. Based on limited correspondence with some Canadian-based companies, we estimate that domestic production of lead sinkers has not changed substantially in recent years. However, some companies that previously produced only lead tackle have expanded their product lines to include lead-free sinkers and jigs. Some of these companies now produce alternative sinker products for sale and have contributed to sinker exchange programs in Canada and the United States.

Prior to 1995, lead sinker imports were estimated to account for a relatively small proportion (<5%) of the total market demand (Scheuhammer and Norris 1995). With the dramatic increase in foreign-made sinkers that began in 1995, it is estimated that imported products may now comprise as much as 18% of the current market, or almost 100 tonnes annually. The balance of the product used in Canada (~400 tonnes) is believed to originate from home and small business manufacture of lead sinkers that are then sold to individual anglers, tackle distributors, and retailers. Although we have no direct information about home production of sinkers in Canada, it is believed that such an industry must exist, because the estimated annual purchase of sinkers is substantively higher than the estimated import plus

Figure 9
Sources of lead fishing tackle comprising the Canadian fishing sinker market, 1995–1998

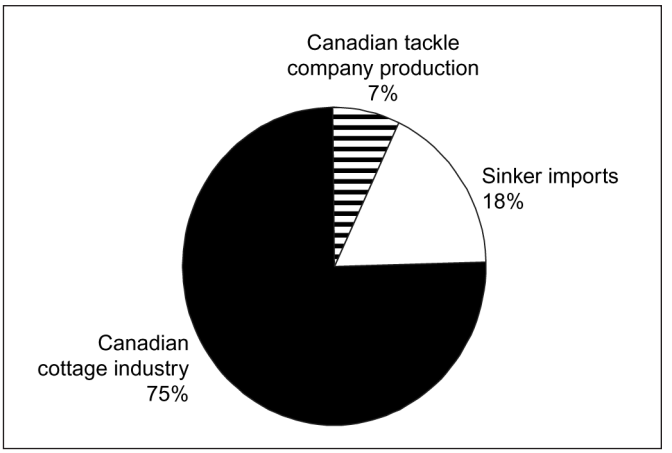
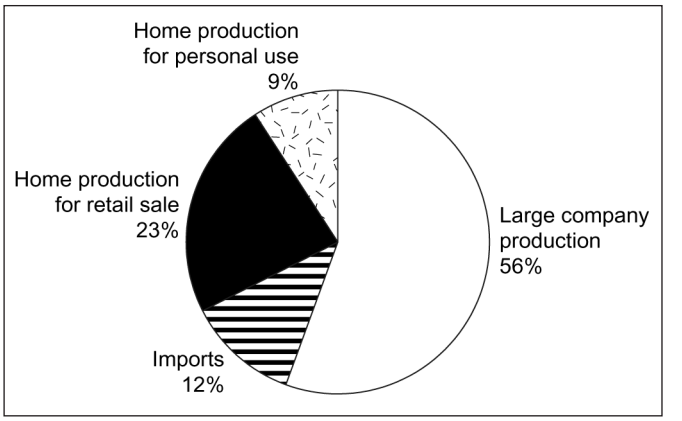


Figure 10
Sources of lead fishing tackle comprising the U.S. fishing sinker market, based on 1994 estimates (total volume of sales ~2600 tonnes) (Nussman 1994)



domestic production by relatively large tackle companies (Fig. 9).

In the United States, an estimated 480 million lead fishing sinkers (2500–2600 tonnes) are sold annually (U.S. EPA 1994; Nussman 1994). Domestic production of sinkers by fewer than 10 major manufacturing companies is estimated to be about 1500 tonnes annually. Sinker imports on average contribute only 320 tonnes to the market, while do-it-yourself home manufacture for retail and personal use together contribute about 875 tonnes (Nussman 1994) (Fig. 10).

2.4 Estimating environmental lead deposition from the annual loss of sinkers and jigs

Fishing sinkers and jigs may be accidentally dropped into the water or may be lost if the hook or line becomes entangled and the line breaks or is cut. Geographically, environmental deposition of lead in the form of lead fishing sinkers and jigs would be concentrated in areas of highest angling pressure. It is assumed that the majority of sinkers purchased annually are to replace those lost while fishing; thus, the magnitude of lead deposition from the loss of lead fishing weights can be estimated by monitoring annual production and sale. Angler questionnaires can contribute additional useful information.

During the summers of 1996 and 1997, researchers from the University of Arizona interviewed over 850 anglers from 12 U.S. states where previous wildlife surveys had documented elevated lead exposure and mortality in loons from lead sinkers and jigs or where angling pressure was high (Duerr 1999). Anglers were asked how long they had spent fishing on the survey day and whether or not they had lost any tackle that day. For all study sites combined, anglers each reported losing, on average, 0.18 sinkers per hour, 0.14 pieces of fishing line per hour, and 0.23 hooks and lures per hour. About 2% of anglers (16 of 859) reported releasing or losing fish with tackle attached. At this rate, each angler would have lost about 1 sinker for every 6 hours of fishing. Similarly, British anglers were reported to have lost or discarded an average of two or three sinkers per angling day (Bell et al. 1985). Another survey conducted in 1986 in the United States estimated that for every one split shot sinker used, four to six might be spilled and lost (Lichvar 1994). Split shot sinkers account for almost half of the total U.S. sinker production (U.S. EPA 1994). We are unaware of any Canadian surveys to determine rates of sinker loss by anglers; however, it is reasonable to assume that Canadian anglers probably experience loss rates comparable to those reported in the United States — about one sinker per angler per angling day. Given this assumption, approximately 66 million sinkers are lost annually in Canada. Annual angling budget expenditures estimated by the U.S. EPA (1994) and Scheuhammer and Norris (1995) suggested that the average angler purchases about 14 sinkers annually, or about one sinker per angling day, comparable to the number estimated lost. These studies indicate that annual sales of sinkers are driven primarily by sinker losses.

In a 1999 lead assessment report, the Minnesota Pollution Control Agency estimated that 49 tonnes of lead are used annually for fishing in Minnesota; however, no specific data were available to indicate the proportion of this lead that may be lost annually into lakes (Nankivel 1999). A study by the New Zealand Department of Conservation reported that retail outlets in the Lake Taupo, New Zealand, area had sold approximately 4 tonnes of lead fishing tackle over a 12-month period, providing an indication of the annual losses within that region. Investigations of the impacts of lead pollution in this region are continuing (Royal Forest and Bird Protection Society 1999). Similarly, areas of Canada that experience heavy angling pressure probably experience relatively high local lead contamination from loss of sinkers during angling.

Estimates of the local density of lost or discarded lead fishing sinkers and jigs in the environment have been made in the United Kingdom and the United States through visual inspections of soils and sediments (Bell et al. 1985; Forbes 1986; Sears 1988), wet sieving sediment samples (Cryer et al. 1987), drying sediments and hand sorting (Bell et al. 1985), or use of radiography (Sears 1988). More recently, Duerr and DeStefano (1999) described the use of a metal detector to estimate densities of sinkers on shorelines, shallow sediments, and lake bottoms. Lead sinker density in U.S. shoreline soil and sediments ranged from 0–0.01 sinkers per square metre in areas of low angling pressure to 0.47 sinkers per square metre in areas with high angling pressure (Duerr 1999). Lead sinker abundance was found to be higher

Table 2
Studies reporting densities of lost or discarded sinkers

Country/state	Location	Substrate	Sinkers/jigs per m^2	Reference
United Kingdom				
	River Thames	Sediments (<1 m) Shoreline	0.9–6.2 1.0–16.3	Sears 1988
	Woodstock Pool	Fishing platforms	105.2	Bell et al. 1985
	Llandindod Wells Lake	Shoreline Island	14.2 21.2	Forbes 1986
	South Wales	Shorelines with: heavy angling moderate angling light angling	 189.7 53.6 24.0	Cryer et al. 1987
United States				
Arizona	Arkansas Nuclear, Pope	Shoreline	0.07 HF[a]	Duerr 1999
Florida	One Canaveral, Brevard	Shoreline	0 HF	Duerr 1999
Florida	Merritt Island Refuge, Brevard/Volusa	Shoreline	0.02 HF 0 OT[b]	Duerr 1999
Idaho	Henry's Fork, Fremont	Shoreline	0.08 HF 0.01 OT	Duerr 1999
New Hampshire/ Maine	Umbagog Lake, Coos, New Hampshire/Oxford, Maine	Shoreline Lake bottom Snorkeling	0.05 HF 0.004 0.0 OT – 0.13 HF	Duerr 1999
Maine	Rangeley Lake Refuge, Oxford/Franklin, Maine	Lake bottom Snorkeling	0–0.008 HF/OT 0.4 OT – 0.0 HF	Duerr 1999
Michigan	Seney Refuge, Schoolcraft	Shoreline	0 HF 0.002 OT	Duerr 1999
North Carolina	Snake River, Mattamuskeet, Hyde	Shoreline	0.01 HF 0.004 OT	Duerr 1999
North Carolina	Pungo District of Pososin, Lakes Hyde/Washington	Shoreline	0 HF 0.003 OT	Duerr 1999
North Virginia	Ruby Lake, Refuge Pine	Shoreline	0.47 HF 0.01 OT	Duerr 1999
Wisconsin	Turtle Flambeau, Flowage Iron	Shoreline	0.0003 OT	Duerr 1999
Vermont	Missisquoi, Franklin	Shoreline	0.1 HF 0 OT	Duerr 1999

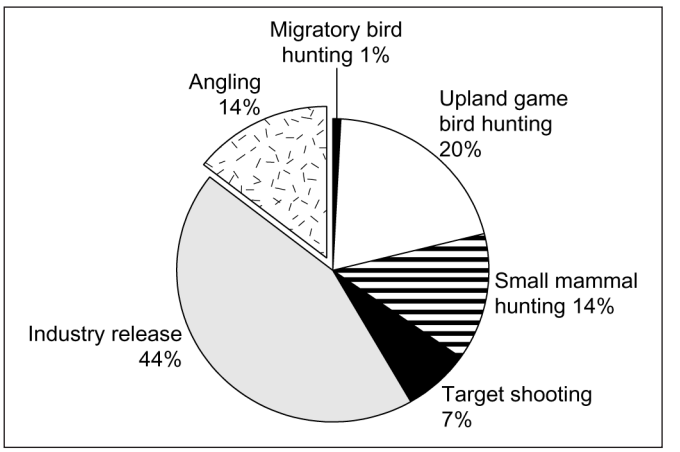
[a] HF = heavily fished (areas known to be subject to heavy angling pressure).
[b] OT = other (areas not subject to heavy angling pressure).

in United Kingdom surveys, ranging from 0.9–16 sinkers per square metre in River Thames sediments and shoreline (Sears 1988) to 24–190 sinkers per square metre of shoreline in South Wales (Cryer et al. 1987). Results of these studies are summarized in Table 2.

In 1996, Environment Canada's National Pollutant Release Inventory (NPRI) estimated that 1699 tonnes of lead were released into the Canadian environment by the industrial sector (NPRI 1996). Ninety-five percent of these releases were attributable to the primary metal mining and smelting industries. Releases to air and water by the mining industry have decreased substantially since the early 1990s, and further decreases are anticipated. Lead releases in the form of spent shotshell ammunition and the loss of fishing sinkers and jigs are typically not formally reported within the context of environmental lead emissions, such as the NPRI inventory; thus, relative contributions to overall lead releases from these sources are usually overlooked.

Spent lead shot from hunting and target shooting was estimated to contribute approximately 2400 tonnes of lead annually to the Canadian environment, prior to regulatory action beginning in the early 1990s (Scheuhammer and Norris 1995). By September 1999, the use of lead shot was prohibited for hunting most migratory game birds in all areas of Canada (Canada Gazette 1997a), presumably eliminating the contribution to environmental lead releases previously made by waterfowl hunting. In addition, the use of lead shot was prohibited for all hunting within National Wildlife Areas (Canada Gazette 1996). Together, these regulations should reduce the deposition of lead shot into the environment from hunting activities by about 800 tonnes annually. With these lead shot restrictions and continued industrial declines in emissions, recreational angling represents an increasing proportion (up to 14%, 559 tonnes) of the total amount of lead annually discharged into the Canadian environment (Fig. 11).

Figure 11
Estimated environmental lead releases in Canada from recreational angling and other sources (total estimated deposition ~3800 tonnes/year)



2.5 Summary

- Canadian anglers spend an estimated $17.9 million annually purchasing over 500 tonnes of lead in the form of fishing sinkers. An undetermined additional amount of lead is sold in the form of jigs.
- Regulations prohibiting the use of lead fishing sinkers and jigs within National Wildlife Areas and National Parks in 1997, together with local outreach efforts, have affected about 1% of anglers and have reduced lead sinker and jig demand by about 5 tonnes annually.
- Domestic production and import of sinkers in Canada averaged 140 tonnes annually between 1995 and 1998. The balance of the sinker market in Canada (approximately 400 tonnes) is believed to originate from home and other small-scale production.
- The U.S. sinker market was estimated to use 2500–2600 tonnes of lead annually, with home production contributing 875 tonnes (32%).
- Lead fishing sinkers and jigs may be accidentally dropped into the water or may be lost if the hook or line becomes entangled and the line breaks or is cut.
- U.S. anglers lose an estimated one sinker every 6 hours of fishing. Based on similar angling habits and methods in Canada and the United States, it is assumed that Canadian anglers experience a similar rate of sinker loss.
- Lost or discarded fishing sinkers and jigs introduce about 500 tonnes of metallic lead into the Canadian environment annually and represent up to 14% of all lead releases.

3. Lead sinker ingestion and poisoning in wildlife

3.1 Lead poisoning in wildlife

Elevated lead exposure and/or lead toxicosis in wildlife have traditionally been associated primarily with the ingestion of spent lead shot from hunting, as well as proximity to heavily travelled roads (i.e., lead from gasoline), base metal mining, smelting, and refining activities, and agricultural areas where lead arsenate pesticide was used (Eisler 1988). More recently, lead poisoning — primarily in Common Loons in North America (Pokras et al. 1993; Scheuhammer and Norris 1995) and swans in Great Britain (O'Halloran et al. 1988; Sears et al. 1989) — has been associated with the ingestion of lead fishing sinkers or jigs.

Ingested lead sinkers and jigs may become lodged in the gizzard, where metallic lead is oxidized and ionic lead released as a result of the grinding action of the gizzard and the acidic environment of the upper digestive tract. Ingestion of a single lead sinker or lead-headed jig, which may represent up to several grams of lead, is sufficient to expose a loon or other bird to a lethal dose of lead (Pokras et al. 1993). In such cases of mortality from acute lead poisoning, carcasses may appear to be in good body condition (Pokras et al. 1993).

Common Loons with ingested lead fishing weights present varying necropsy results, with no single finding predominating (Pokras et al. 1993). While proventricular distention, bile-stained gizzard lining, and green-stained vent feathers are commonly observed in waterfowl with lead poisoning from lead shot ingestion (Friend 1985), these signs are often not present in loons that have ingested lead fishing weights (Pokras et al. 1993). Absence of these signs, and the good body condition in which loons are usually found, suggested that mortality was relatively rapid due to acute lead poisoning.

Four loons that ingested small lead fishing weights in Atlantic Canada were found dead, in poor body condition, and with renal lead concentrations compatible with lead poisoning (Daoust et al. 1998). The gizzards of two of these birds had chronic traumatic lesions caused by penetration of a fishing hook. In these cases, both lead exposure and trauma caused by penetration of a fishing hook were judged to have contributed to the deteriorated body condition.

Diagnosis of lead shot or sinker ingestion by wild birds may be made through radiographic or fluoroscopic examination of the gizzard or by determination of tissue lead levels, most commonly in liver and/or kidney and occasionally blood. Blood lead levels in Common Loons without lead sinker ingestion are generally below 0.1 µg/mL. Loons confirmed to have ingested lead sinkers demonstrate highly elevated blood lead concentrations (Table 3), concurrent with inhibition of aminolevulinic acid dehydratase activity.

Although lead concentrations in liver, kidney, and bone tissue of most wildlife without lead shot or sinker ingestion are generally below 5 µg/g dry weight, tissue concentrations in Common Loons in Canada with confirmed lead sinker or jig ingestion were as high as 142 µg/g dry weight in liver and 726 µg/g dry weight in kidney (Table 3). Similar results have been reported in the United States. In a few cases, high lead exposure was found in individuals that did not show evidence of a lead sinker or jig in the gizzard at the time of examination, but this was uncommon (Franson and Cliplef 1993; Pokras et al. 1993; Scheuhammer and Norris 1995). Evidence gathered to date indicates that ingestion of lead sinkers and jigs is the only significant source of elevated lead exposure and lead toxicity for Common Loons.

3.2 Lead sinker and jig ingestion cases in wildlife

Among the earliest published reports of lead poisoning from sinker ingestion in North American wildlife are those of Locke and Young (1973) (a Whistling Swan *Olor columbianus*) and Locke et al. (1982) (three Common Loons). By the early 1990s, researchers had established networks of wildlife biologists, wildlife rehabilitators, and veterinarians to assist in reporting cases of lead poisoning in wildlife associated with the ingestion of lead fishing sinkers and jigs.

Scheuhammer and Norris (1995) summarized 46 instances of mortality from lead sinker and jig ingestion in Canada from eight wildlife species, including loons and various waterfowl and raptors. Since 1995, at least 26 additional cases of mortality from lead sinker and jig ingestion have been reported and include two new species: Herring Gull *Larus argentatus* and snapping turtle *Chelydra serpentina* (Table 4). Although lead sinkers were not directly implicated in its death, an endangered Whooping Crane *Grus americana* from western Canada was found to have elevated lead levels in blood, kidney, and liver and may have ingested a lead sinker (Snyder et al. 1991). Lead poisoning from ingestion of lead shot and lead fishing sinkers was identified

as an important factor limiting efforts to reintroduce the Trumpeter Swan *Cygnus buccinator* in Ontario (Hunter 1995). Radiographic analyses have also revealed lead fishing weights in dead Great Blue Herons *Ardea herodias*, Double-crested Cormorants *Phalacrocorax auritus*, Green Herons *Butorides virescens*, and unidentified gull and swan species submitted to a wildlife rehabilitator in southwestern Ontario (Twiss and Thomas 1998). Virtually all species of piscivorous bird, as well as species that feed in nearshore soils and sediments, are at risk of lead poisoning from inadvertent consumption of lost or discarded lead sinkers.

In the United States, Perry (1994) compiled over 300 cases of sinker ingestion in over 20 species of wildlife, including loons, swans, ducks, geese, cranes, pelicans, and cormorants. Most recently, two species of turtle have been found to ingest lead fishing sinkers. U.S. researchers have now documented lead sinker and jig ingestion in at least 371 individuals from 23 different wildlife species (Table 5).

Table 3
Summary of tissue lead concentrations for Common Loons with and without the presence of lead artifacts in their digestive systems

	Tissue concentrations (µg/g dry weight, except blood, µg/mL)				
	Blood	Liver	Kidney	Bone (radius)	Source[a]
Without lead artifact in gizzard					
Canada	NA	<5 (n = 28)	<5 (n = 28)	<5 (n = 28)	1
USA	0.01–0.05 (n = 16)	0.2–0.44 mean 0.24 (n = 10)	NA	NA	2
With lead artifact in gizzard					
Canada	NA	17–142 mean 59.2 (n = 12)	18.6–727 mean 218 (n = 25)	2.7–11 mean 5.7 (n = 8)	1, 3, 4
USA	0.78, 2.03 (n = 2)	20–72 mean 46 (n = 4)	NA	NA	2

[a] Sources are as follows:
1. A. Scheuhammer, Canadian Wildlife Service, National Wildlife Research Centre, Carleton University, Raven Road, Ottawa, Ontario, unpubl. data, 1999.
2. Pokras et al. (1992).
3. D. Campbell, Ontario Veterinary College, Guelph University, Guelph, Ontario, unpubl. data, 1999.
4. P.-Y. Daoust, Atlantic Veterinary College, Charlottetown, Prince Edward Island, unpubl. data, 1999.

Table 4
Wildlife species documented to have ingested lead fishing sinkers or jigs in Canada, 1964–1999

Species	n	Location	Date	Source[a]
Piscivorous birds				
Common Loon *Gavia immer*	1	British Columbia	1993	1
	1	Saskatchewan	1997	2
	36	Ontario	1987–1999	3, 4
	6	Quebec	1964–1994	5, 6, 7, 8
	4	New Brunswick	1993–1997	9
	6	Nova Scotia	1992–1996	
	1	Prince Edward Island	1990–1997	
Common Merganser *Mergus merganser*	1	Ontario	1990–1994	4
Herring Gull *Larus argentatus*	1	Quebec	1993	8
Bald Eagle *Haliaeetus leucocephalus*	1	British Columbia	1990–1994	1
Nonpiscivorous birds				
Trumpeter Swan *Cygnus buccinator*	1	British Columbia	1993	1
	1	Ontario	1995	4
Canada Goose *Branta canadensis*	4	Ontario	1992–1998	4, 10
	2	Quebec	unknown	11
	1	New Brunswick	unknown	1
	1	Prince Edward Island	1992	12
Mallard *Anas platyrhynchos*	1	Ontario	1993	9
Greater Scaup *Aythya marila*	1	Ontario	1994	9
White-winged Scoter *Melanitta fusca*	1	Ontario	1994	9
Other aquatic wildlife				
Snapping turtle *Chelydra serpentina*	1	Quebec	1994	11
Totals: 10 species	72	7 provinces	1964–1999	

[a] Sources are as follows:
1. Langelier (1994).
2. M. Wayland, Canadian Wildlife Service – Prairie and Northern Region, unpubl. data.
3. K. Chubb, Avian Care and Research Foundation, Verona, Ontario, unpubl. data.
4. D. Campbell, Ontario Veterinary College, Guelph University, Guelph, Ontario, unpubl. data.
5. Parks Canada, Quebec, unpubl. data.
6. R. Ouellet, ministère de l'Agriculture, des Pêcheries et de l'Alimentation, Quebec, unpubl. data, 1994.
7. R. Ouellet, ministère du Loisir, de la Chasse et de la Pêche, Quebec, unpubl. data, 1968.
8. J. Rodrigue and L. Champoux, Canadian Wildlife Service – Quebec Region, unpubl. data.
9. P.-Y. Daoust, Atlantic Veterinary College, Charlottetown, Prince Edward Island, unpubl. data.
10. H. Pittel, Avicare Rehabilitation Centre, Bowmanville, Ontario, unpubl. data.
11. M. Ouellet, Redpath Museum, McGill University, Montreal, Quebec, unpubl. data.
12. Barrow (1994).

Table 5
Wildlife species documented to have ingested lead fishing sinkers or jigs in the United States, 1976–1999

Species	n	Location	Year	Source[a]
Piscivorous birds				
Common Loon *Gavia immer*	1	Florida	1993	1
	34	Maine	1989–1999	2, 3
	3	Maine	1976–1991	4
	5	Massachusetts	1989–1999	2, 3
	3	Michigan	1988–1993	5
	5	Minnesota	1984–1990	6
	4	Minnesota	1976–1991	4
	64	New Hampshire	1989–1999	2, 3, 7
	1	New Hampshire	1976	8
	1	New Hampshire	1976–1991	4
	7	New York	1983, 1986, 1989	9
	12	New York	1994	10
	9	Vermont	1989–1999	2, 3
	1	Vermont	1976–1991	4
	1	Wisconsin	1980	8
	2	Wisconsin	1976–1991	4
American White Pelican *Pelecanus erythrorhynchos*	1	Washington	1993	11
Brown Pelican *Pelecanus occidentalis*	45	Florida	1991–1993	1
Double-crested Cormorant *Phalacrocorax auritus*	26	Florida	1991–1993	1
	1	Maine	1994	7
Snowy Egret *Egretta thula*	1	Florida	1991	1
Great Egret *Ardea alba*	3	Florida	1991	1
Great Blue Heron *Ardea herodias*	1	Massachusetts	1989–1993	7
	8	Florida	1991–1993	1
White Ibis *Eudocimus albus*	1	Florida	1993	1
Mississippi Sandhill Crane *Grus canadensis pulla*	3	Mississippi	?	12, 13
Red-breasted Merganser *Mergus serrator*	1	Florida	1993	1
Laughing Gull *Larus atricilla*	9	Florida	1991–1993	1
Herring Gull *Larus argentatus*	1	Florida	1993	1
Royal Tern *Sterna maxima*	7	Florida	?	1
Nonpiscivorous birds				
Whistling or Tundra Swan *Cygnus columbianus*	1	Michigan	1988–1993	5
	1	New York	1986	14
	1	Maryland	1993	15
	12	Minnesota	1983–1993	16
Trumpeter Swan *Cygnus buccinator*	15	Idaho		17
Mute Swan *Cygnus olor*	5	Michigan	1988–1993	5
	1	Michigan	?	8
	1	Michigan	?	18
	1	New York	1986	14
Canada Goose *Branta canadensis*	25	Minnesota	1983–1993	16
	1	Washington	1983–1993	11
Mallard *Anas platyrhynchos*	25	Minnesota	1983–1993	16
American Black Duck *Anas rubripes*	1	New York	1994	10
Wood Duck *Aix sponsa*	6	Minnesota	1983–1993	16
Redhead *Aythya americana*	10	New York	1994	10
	1	New York	1994	19
Other aquatic wildlife				
Snapping turtle *Chelydra serpentina*	2	Massachusetts	1990–1998	2, 20
Painted turtle *Chrysemys picta*	1	Massachusetts	1990–1998	2
Totals: 23 species	371	13 states	1976–1999	

[a] Sources are as follows:
1. B. Suto, Suncoast Seabird Sanctuary, Indian Shores, Florida, unpubl. clinic records, 1994.
2. M. Pokras, Tufts University, School of Veterinary Medicine, North Grafton, Massachusetts, unpubl. necropsy reports, 1999.
3. Pokras et al. (1993).
4. Franson and Cliplef (1993).
5. Michigan Department of Natural Resources, Rose Lake Wildlife Research Center, Wildlife Disease Laboratory, East Lansing, Michigan, unpubl. necropsy reports, 1993.
6. Ensor et al. (1992).
7. M. Pokras, Tufts University, School of Veterinary Medicine, unpubl. necropsy reports, 1994.

Continued

Table 5 (*continued*)
Wildlife species documented to have ingested lead fishing sinkers or jigs in the United States, 1976–1999

8. Locket et al. (1982).
9. W. Stone, New York State Department of Environmental Conservation, Wildlife Resource Centre, Avon, New York, unpubl. necropsy reports, 1990.
10. W. Stone, New York State Department of Environmental Conservation, Wildlife Resource Centre, Avon, New York, unpubl. necropsy reports, 1994.
11. R. Ralston, U.S. Fish and Wildlife Service, National Wildlife Forensics Laboratory, Ashland, Oregon, unpubl. necropsy reports, 1994.
12. U.S. Fish and Wildlife Service (1992).
13. Windingstad et al. (1984).
14. W. Stone, New York State Department of Environmental Conservation, Wildlife Resource Centre, Avon, New York, unpubl. necropsy reports, 1986.
15. Locke and Young (1973).
16. L. Wolf, University of Minnesota Wildlife Clinic, St. Paul, Minnesota, unpubl. clinic records, 1994.
17. Blus et al. (1989).
18. Gelston and Stuht (1975).
19. C. Franson, U.S. Geological Survey, National Wildlife Health Research Center, Madison, Wisconsin, unpubl. necropsy reports, 1993.
20. Borkowski (1997).

3.3 Types and sizes of lead fishing weights ingested by wildlife

Sinkers used for most freshwater angling range in weight from 0.3 to 230 g and in length and diameter from about 2 mm to 8 cm (Scheuhammer and Norris 1995). Lead fishing weights that weigh less than 50 g or are smaller than 2 cm in any dimension are generally the size found to be ingested by wildlife (Scheuhammer and Norris 1995), although larger sizes can be ingested by larger waterbirds, such as pelicans (C. Franson, U.S. Geological Survey, pers. commun.).

For Common Loons collected in Canada, for which the specific form of lead weight ingested was reported, (31 of 40) 77.5% were found to have ingested lead sinkers, 6 had ingested lead-headed jigs and 3 loons were found to have ingested both sinker and jig.

Lead weights ranging from 2-g split shot (4 mm^2) to 20-g homemade sinkers (5 mm × 22 mm) have been recovered from Common Loons in southern Ontario (K. Chubb, Avian Care and Research Foundation, pers. commun.). Ingested lead weights retrieved from loons in New England ranged in size from 0.6 mm to 38 mm and weighed as much as 23 g (M. Pokras, Tufts School of Veterinary Medicine, pers. commun.).

At least one-third of all sinker ingestion cases in loons from Ontario and New England have also included the presence of other fishing tackle, such as hooks, lines, etc. (K. Chubb, Avian Care and Research Foundation, pers. commun.; M. Pokras, Tufts School of Veterinary Medicine, pers. commun.). Small pebbles were often associated with the other two-thirds of the sinker ingestion cases, indicating that the sinkers may have been picked up from the bottom of the water body while the birds were seeking grit material. Franson et al. (2001) reported that small (2.4–9.5 mm) stones were recovered from the stomach contents of 78% of Common Loons found dead in New England and the southeastern United States; these data indicate the most probable size range of sinkers that loons might inadvertently ingest from lake bottoms.

3.4 Relative importance of lead sinker ingestion as a cause of Common Loon mortality

There are numerous causes of mortality in loons, the most common of which are drowning in commercial fishing nets, “trauma” from boat or other collisions or gunshot wounds, disease (especially botulism and aspergillosis), and lead poisoning from sinker ingestion.

Avian botulism is a paralytic condition occurring when birds consume a naturally occurring toxin produced by the bacterium *Clostridium botulinum*. Type E botulism, in particular, affects fish-eating birds and has, during a few years, reached epidemic levels in the Great Lakes region, causing sporadic outbreaks of mortality in Common Loons and other fish-eating birds (Brand et al. 1988); however, it has been diagnosed only infrequently elsewhere in the United States or Canada. A particularly severe and persistent outbreak of Type E botulism began on Lake Erie during 1999–2000 and continues in 2002, with hundreds or thousands of migrating Common Loons estimated to have died of the disease (Campbell and Barker 1999).

Overwintering Common Loons occasionally experience large-scale die-offs, sometimes numbering in the hundreds or thousands of individuals. In these cases, the majority of dead loons have been found to have succumbed to an emaciation syndrome characterized by loss of body fat, atrophy of pectoral muscle, and hemorrhagic enteritis; however, the etiology of this syndrome remains uncertain (Forrester et al. 1997). Spitzer (1995) discussed various sources of Common Loon mortality on marine wintering areas, including storms, food limitation, entanglement in fishing nets, and oil spills.

Aspergillosis, a fungal infection of the respiratory tract, has been commonly diagnosed in dead loons, but in most cases it is presumed to be secondary to other conditions, because only unhealthy or starving birds are thought to be unable to eliminate *Aspergella* spores from their lungs (Wobeser 1981). Aspergillosis is thus associated with immunosuppression, although the causes are generally uncertain.

In eastern Canada and the United States, several studies (primarily during the breeding season) have compared the relative importance of different causes of mortality and have concluded that lead sinker poisoning is often one of the leading causes of mortality for Common Loons. For example, in 105 Common Loon carcasses from

Figure 12
Relative importance of different causes of adult Common Loon mortality in New England between 1989 and 1992 (n = 60) (data from Pokras et al. 1993)

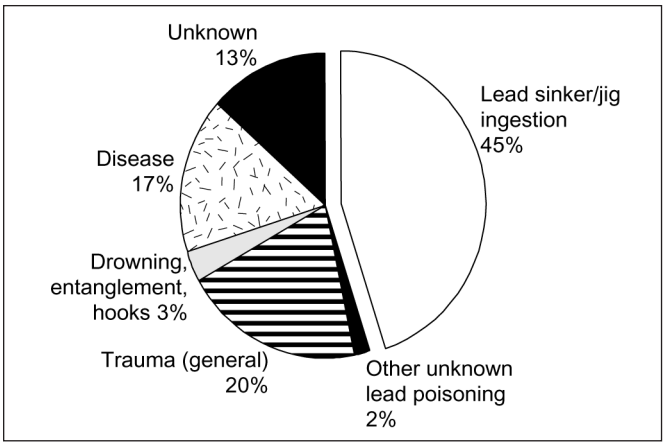


Figure 13
Relative importance of different causes of adult Common Loon mortality in Canada (based on cases reported in Ontario and Atlantic Canada between 1983 and 1995; n = 122)

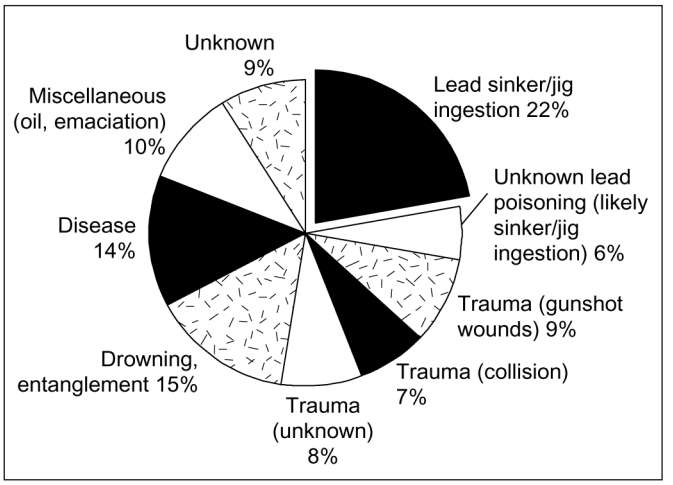


Table 6
Percentage of total reported mortality in adult Common Loons attributed to lead sinker/jig ingestion in Canadian and U.S. studies

Area of study	% mortality	Source[a]
Canada		
Saskatchewan	5 (1 of 21)	1
Ontario	18 (21 of 114)	2
Ontario	30 (19 of 63)	3
Quebec	6 incidental reports (6 of 6)	4, 5
New Brunswick, Nova Scotia, Prince Edward Island	17 (4 of 23)	6
New Brunswick, Nova Scotia, Prince Edward Island	22 (8 of 37)	7
United States		
18 states combined	5 (11 of 222)	8
Michigan	40 (15 of 38)	9
Minnesota	17 (37 of 221)	10
New England states	53 (98 of 185)	11

[a] Sources are as follows:
1. M. Wayland, Canadian Wildlife Service – Prairie and Northern Region, unpubl. data, 1997.
2. D. Campbell, Ontario Veterinary College, Guelph University, Guelph, Ontario, unpubl. data, 1991–1998.
3. K. Chubb, Avian Care and Research Foundation, Verona, Ontario, unpubl. data, 1983–1998.
4. Parks Canada, Quebec, unpubl. data, 1964–1988.
5. R. Ouellet, ministère de l'Agriculture, des Pêcheries et de l'Alimentation, Quebec, unpubl. data, 1994.
6. P.-Y. Daoust, Atlantic Veterinary College, Charlottetown, Prince Edward Island, unpubl. data, 1992–1994.
7. P.-Y. Daoust, Atlantic Veterinary College, Charlottetown, Prince Edward Island, unpubl. data, 1994–1998.
8. Franson and Cliplef (1993).
9. Poppenga et al. (1993).
10. Ensor et al. (1992).
11. M. Pokras, Tufts University, School of Veterinary Medicine, North Grafton, Massachusetts, unpubl. data, 1989–1998.

New York state examined between 1972 and 1999, lead poisoning and aspergillosis accounted for 21% and 23%, respectively, of overall mortality; no other single cause of mortality exceeded 10% (Stone and Okoniewski 2001). Similarly, in Atlantic Canada, 26% (8 of 31) of Common Loons found dead were diagnosed to have died from lead poisoning (Daoust et al. 1998). Lead poisoning from sinker or jig ingestion was the single most important cause of mortality for adult Common Loons in New England, accounting for 45% (27 of 60) of recorded adult Common Loon mortality in that region (Pokras et al. 1993) (Fig. 12). Excluding mortality in saltwater environments, lead sinker and jig ingestion accounts for over half (53%) of the reported adult loon deaths in the New England states (M. Pokras, Tufts School of Veterinary Medicine, unpubl. data).

In the early 1990s, surveys conducted in Ontario and Atlantic Canada demonstrated that lead poisoning from the ingestion of lead fishing sinkers and jigs was a leading cause of death in adult Common Loons, accounting for about 15–30% of reported mortality (Scheuhammer and Norris 1995). Drowning as a result of entanglement in fishing lines and disease, the other leading causes of death in adult loons, accounted for an additional 29% of reported deaths in Canada. In some areas (e.g., northern Quebec), Aboriginal harvest of loons probably accounts for a significant proportion of local or regional loon mortality (Coad 1994); however, in general, loons are not hunted in North America. Since the early 1990s, lead sinker/jig ingestion has accounted for approximately 22% (59 of 264) of the total reported mortality in adult Common Loons examined in Canada (Table 6). Figure 13 shows the relative importance of different mortality factors for Common Loons in Ontario and Atlantic Canada, the regions having the most data on loon mortality in Canada.

In Michigan, out of 180 loon carcasses examined between 1987 and 2001, 42 (23%) were judged to have died from lead poisoning, usually from sinker ingestion, whereas drowning accounted for 24% and "trauma" 21% of total mortality (Michigan Department of Natural Resources, unpubl. data). In Minnesota, lead poisoning accounted for between 5% (Pichner and Wolff 2000) and 17% of reported Common Loon mortality (Ensor et al. 1992).

Although much of the concern regarding lead sinker/jig ingestion by North American wildlife has focused on Common Loons, sinker and jig ingestion may be an important cause of death in other waterbirds as well. In a 4-year study of Mute Swans *Cygnus olor* along the River Thames in England, Sears et al. (1989) found ingestion of sinkers and jigs to be the single greatest cause of illness, accounting for 55% of moribund individuals. Lead sinker

ingestion has also been documented as an important cause of mortality in swans in Ireland (O'Halloran et al. 1988).

Recently, Franson and colleagues (C. Franson, U.S. Geological Survey, pers. commun.) examined carcasses of 2241 individual birds of 29 species from the United States that died in rehabilitation centres or were found dead in the field for the presence of ingested or entangled fishing tackle and evaluated radiographs of live birds trapped in the field for images consistent with fishing weights and other tackle. Ingested lead sinkers were found most frequently in Common Loons at a rate of 3.5% (n = 313) and in Brown Pelicans *Pelecanus occidentalis* at a rate of 2.7% (n = 365). Ingested lead fishing weights were also found in 1 of 81 Double-crested Cormorants and 1 of 11 Black-crowned Night-Herons *Nycticorax nycticorax*.

Documenting wildlife mortality from lead sinker ingestion has traditionally depended largely on the volunteer participation of cottagers, anglers, boaters, etc. who come across a carcass and notify an appropriate provincial or federal wildlife agency. Despite an increased awareness of the lead poisoning issue among wildlife researchers, veterinary colleges, wildlife rehabilitators, and others, it is unlikely that such a volunteer effort documents more than a small percentage of the total number of lead sinker poisoning cases.

3.5 Sinker ingestion by wildlife — Spatial distribution

Mortality associated with lead sinker or jig ingestion in wildlife has been reported in seven Canadian provinces and 13 U.S. states (Fig. 14). Common Loon mortality as a result of ingestion of lead sinkers or jigs has been reported in seven provinces and nine states. Reports of lead sinker ingestion by wildlife have come primarily from eastern Canada (especially Ontario and the Maritimes) and the northeastern United States (especially New England); despite occasional reports of lead sinker ingestion in other wildlife species, concerted research efforts have focused on Common Loons.

An average of six cases of wildlife mortality from sinker ingestion have been documented annually in Canada between 1987 and 1998, and about 20 cases have been reported annually in the United States during a similar time period (1983–1998). The frequency of sinker ingestion reports varies geographically. In Canada and the United States, the highest average number of annual ingestion cases originate from southern Ontario and New Hampshire, respectively. These are areas that experience relatively high recreational angling pressure and also a relatively high degree of research and monitoring effort.

3.6 Sinker ingestion by wildlife — Temporal trends

3.6.1 Annual trends

CWS, in partnership with veterinary colleges and wildlife rehabilitation centres, documented an average of 3–6 cases of sinker or jig ingestion and poisoning annually prior to 1995 (K. Chubb, Avian Care and Research Foundation, unpubl. data; D. Campbell, Ontario Veterinary College, unpubl. data; P.-Y. Daoust, Atlantic Veterinary College, unpubl. data). Since 1995, these organizations have each continued to report a few cases of lead sinker ingestion in Common Loons annually. However, few loon carcasses are found and submitted for pathological/toxicological analyses each year. It is probable that many more loons die of lead poisoning than are actually found and submitted for necropsy.

3.6.2 Seasonal trends

Sinker ingestion has been reported in Common Loons from April through December in Ontario and from May to December in New England. For Ontario and New England, 19% (5 of 26) and 31% (8 of 26), respectively, of all loons for which the month of ingestion is known have ingested their lead sinker or jig in August.

Lead poisoning rates of Mute Swans in the United Kingdom were a function of the abundance of lead sinkers in river sediments, but not of the abundance of sinkers on the shores of the river (Sears 1988), indicating that swans ingest sinkers mainly while foraging in sediments. Along the River Thames, reports of elevated blood lead levels and lead poisoning of Mute Swans decreased following the closure of the fishing season, suggesting that the availability of sinkers is greatest when anglers are most active. Lost tackle may settle deeper into sediments as time passes (Birkhead 1983; Sears 1988) and may thus become increasingly unavailable for ingestion. Indeed, since a ban on lead sinkers was established in Britain in 1986, the Mute Swan population decline has dramatically reversed (Kirby et al. 1994).

3.7 Angling pressure and the incidence of sinker ingestion in wildlife

Angling pressure in Canada ranges from <1 to 47 angler days per square kilometre at the provincial/territorial level and increases to over 230 angler days per square kilometre at the regional level in central Ontario (see section 1). The highest frequency of reports of lead sinker poisoning in wildlife was reported in southern Ontario (n = 45), an area where angling pressure can exceed 100 angler days per square kilometre. Regional/local angling pressure estimates for other provinces, particularly those in which cases of sinker poisoning were reported, were not available. This information would be beneficial in identifying other locations where loons and other waterbirds are at significant risk of sinker/jig ingestion.

Wildlife mortality associated with lead sinker and jig ingestion occurs within regions of Canada and the United States where considerable recreational angling pressure occurs (Figs. 15 and 16; Table 7). Angling pressure is often greater in the United States than in Canada and can exceed 500 angler days per square kilometre per year at the state level. In the United States, sinker and jig ingestion has been reported mostly in states with angling pressures exceeding 100 angling days per square kilometre annually. The greatest numbers of wildlife mortality cases have been reported in Florida (n = 103), New Hampshire (n = 78), Maine (n = 37), and New York (n = 21).

3.8 Summary

- Wildlife, primarily waterbirds, ingest fishing sinkers and jigs during feeding, either by mistaking them for food

Figure 14
Distribution of reported incidents of wildlife mortality associated with lead fishing sinker and jig ingestion in Canada and the United States. Numbers of different species per province or state are also indicated.

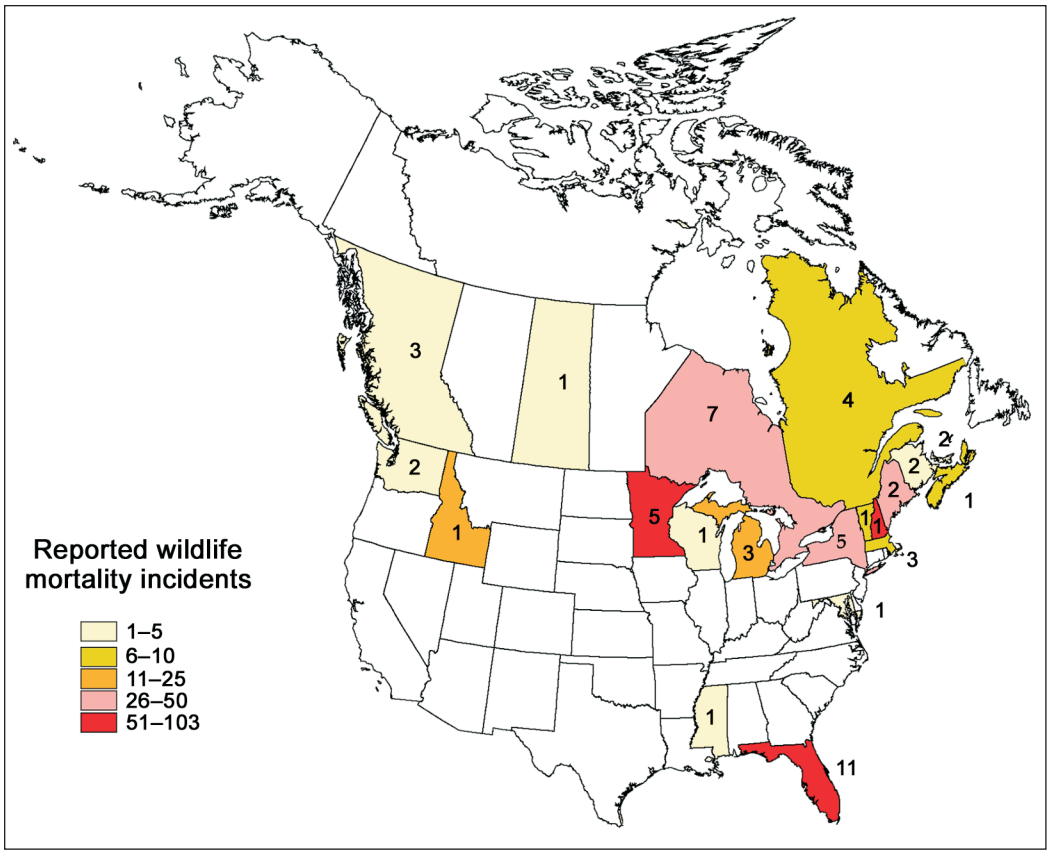


Figure 15
Recreational angling pressure in Canada and the United States, with an indication of those provinces and states where wildlife mortality from lead sinker or jig ingestion has been reported

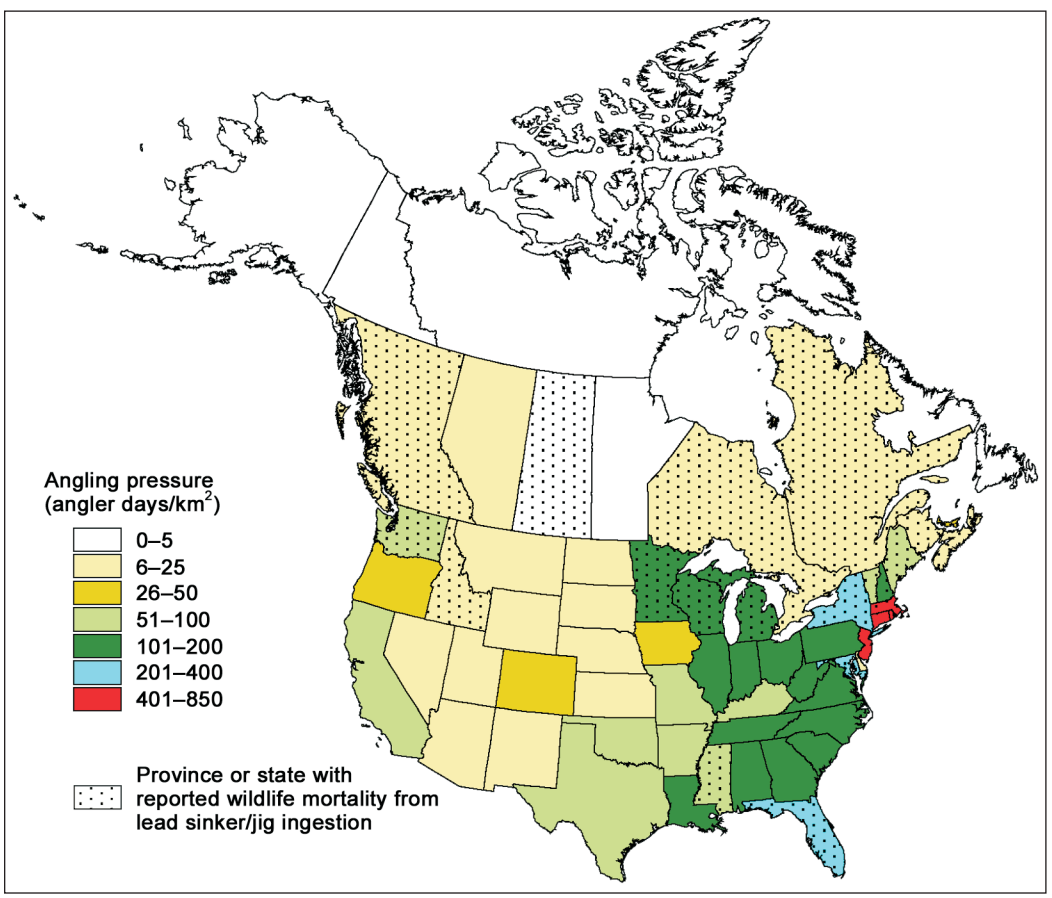


Figure 16
Regions of eastern Canada and the United States with high recreational angling pressure, overlain with locations of known incidents of Common Loon mortality (black circles) from ingestion of lead fishing sinkers or jigs

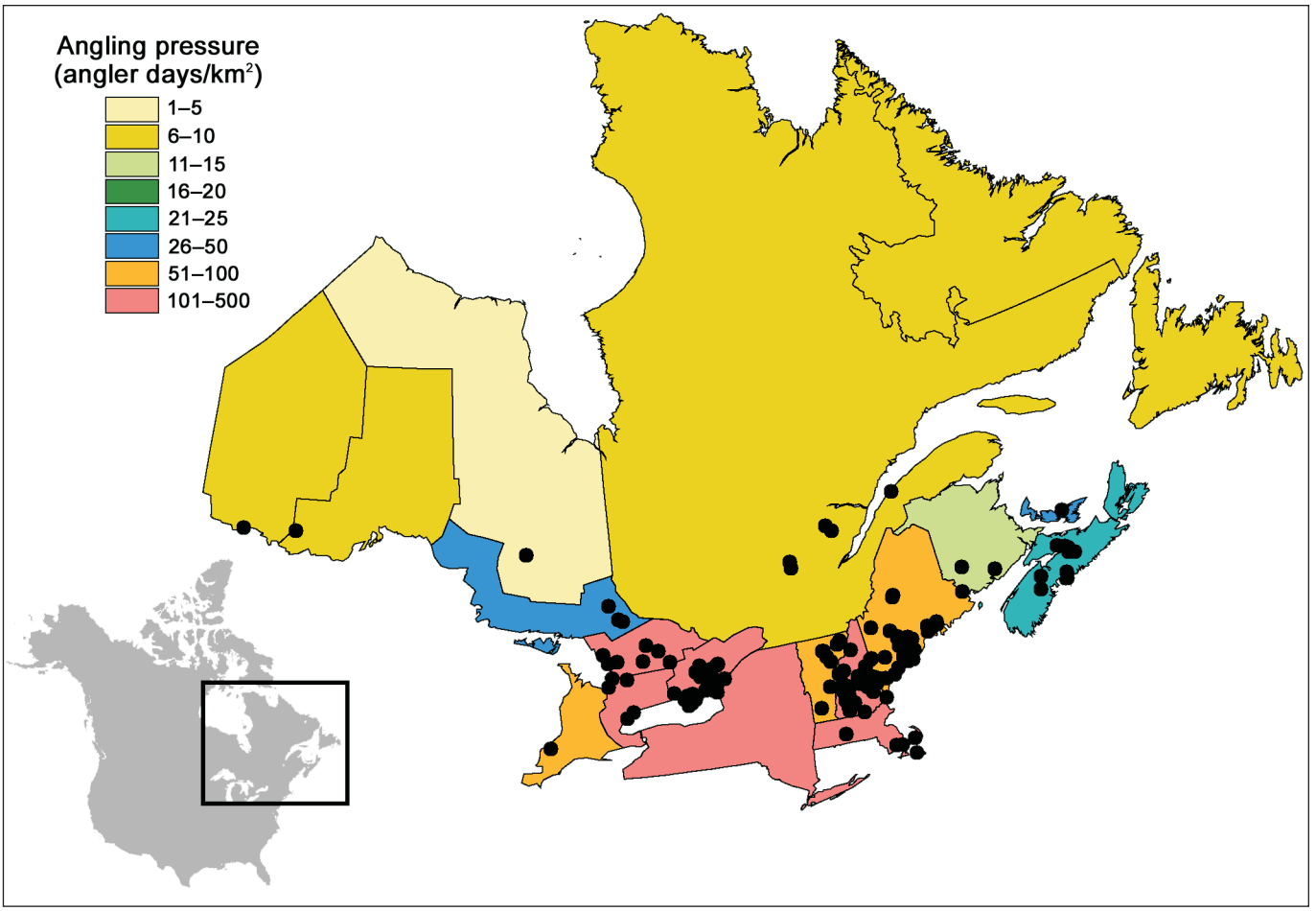


Table 7
Angling effort in areas of North America where sinker/jig ingestion has been reported in Common Loons and other wildlife (data from OMNR 1993; DFO 1997; U.S. Department of the Interior 1997)

Area	Number of reported cases of sinker/jig poisoning	Number of recreational anglers (millions)	Number of days per year spent angling (millions)	Average annual number of days per angler	Geographic area (km²)	Angling pressure (angler days/km²)
Canada	**72 individuals (10 species)**	**4.63**	**55.5**	**13**	**9 922 385**	**5.6**
British Columbia	3	0.71	8.34	11.7	948 595	8.8
Saskatchewan	1	0.18	2.23	13	651 900	3.4
Ontario (total)	45	2.62	31.4	11.8	1 067 582	29.3
Northwest Region	3	0.25	1.93	7.6	228 310	8.5
North Central Region	4	0.13	1.14	9.1	211 826	5.4
Northern Region	3	0.12	1.45	12.0	346 281	4.2
Northeast Region	4	0.35	4.10	11.9	105 994	38.7
Algonquin Region	7	0.42	4.37	10.5	43 182	101.3
Eastern Region	17	0.34	3.77	11.2	32 842	114.7
Central Region	5	0.55	8.77	15.9	36 879	237.9
Southwest Region	1	0.34	4.53	13.5	62 268	72.7
Quebec	10	1.08	10.9	10.1	15 406 803	7.1
New Brunswick	5	0.09	0.91	10.5	73 435	12.4
Nova Scotia	6	0.06	1.15	18.1	55 490	20.8
Prince Edward Island	2	0.01	0.27	19.7	5 655	47.1

Continued next page

Table 7 (*cont'd*)
Angling effort in areas of North America where sinker/jig ingestion has been reported in Common Loons and other wildlife (data from OMNR 1993; DFO 1997; U.S. Department of the Interior 1997)

Area	Number of reported cases of sinker/jig poisoning	Number of recreational anglers (millions)	Number of days per year spent angling (millions)	Average annual number of days per angler	Geographic area (km^2)	Angling pressure (angler days/km^2)
United States	**153 (23 species)**	**35.3**	**628.2**	**17.8**	**9 381 920**	**66.9**
Florida	103	2.9	45.5	15.9	140 255	324.2
Idaho	15	0.48	4.4	9.1	213 445	20.7
Maine	37	0.36	5.1	14.4	80 275	63.7
Maryland	10	0.72	10.2	14.4	25 480	500.1
Massachusetts	9	0.70	10.1	14.4	20 265	400.1
Michigan	11	1.82	28.7	15.7	147 510	194.6
Minnesota	68	1.54	27.0	17.6	206 030	131.1
Mississippi	3	0.58	9.7	16.8	122 335	79.6
New Hampshire	78	0.27	3.5	13.3	23 290	152.0
New York	21	1.71	29.4	17.2	122 705	239.3
Vermont	10	0.19	2.0	10.4	24 015	81.2
Washington	2	1.01	12.9	12.8	172 265	74.7
Wisconsin	3	1.47	17.1	11.6	140 965	121.5

items or by consuming lost bait fish with the line and sinker still attached.

- Ingestion of a single lead sinker or lead-headed jig, which may represent up to several grams of lead, is usually sufficient to expose a loon or other bird to a lethal dose of lead.
- Evidence gathered to date indicates that lead sinker and jig ingestion is the only significant source of elevated lead exposure and lead toxicity for Common Loons.
- Lead sinker and jig ingestion has been documented in 10 wildlife species in Canada, including fish-eating birds (Common Loon, Common Merganser *Mergus merganser*, Herring Gull), waterfowl (Trumpeter Swan, Canada Goose *Branta canadensis*, Mallard *Anas platyrhynchos*, Greater Scaup *Aythya marila*, White-winged Scoter *Melanitta fusca*), raptors (Bald Eagle *Haliaeetus leucocephalus*), and snapping turtles; in the United States, ingestion of lead sinkers and jigs by over 20 species of wildlife, including swans, cranes, pelicans, and cormorants, has been documented.
- Lead fishing weights that weigh less than 50 g or are smaller than 2 cm in any dimension are generally the size that has been found to be ingested by wildlife.
- Lead sinker or jig ingestion is the single most important cause of death of adult Common Loons reported in Canada and the United States, commonly exceeding deaths associated with entanglement in fishing gear, trauma, and disease.
- Documented cases of wildlife mortality from lead sinker ingestion have come largely from serendipitous discovery of carcasses by cottagers, anglers, boaters, and others. Consequently, the total number of loons or other wildlife that die of lead poisoning from sinker ingestion cannot be confidently estimated.
- Lead sinker and jig ingestion in wildlife has been reported in seven provinces and 13 U.S. states.
- In Canada, about six cases of sinker ingestion and poisoning in wildlife (mostly Common Loons) are reported annually, accounting for between 17 and 30% of reported adult loon mortality in Ontario and Atlantic Canada. In the United States, an average of 20 cases of sinker and jig ingestion are reported annually.
- Common Loons have been found to ingest sinkers from April through December in Ontario, with the majority of cases reported in August.
- In Canada, the greatest frequency of wildlife mortality associated with lead sinker or jig ingestion occurs in southern Ontario, where angling pressure can exceed 100 angler days per square kilometre in some areas.
- In the United States, wildlife mortality associated with lead sinker or jig ingestion was reported most frequently in Florida and New England, where state-level angling pressures ranged from 60 to over 300 angler days per square kilometre.

4. Can population-level effects of lead sinker poisoning be determined for Common Loons in Canada?

4.1 Background

In eastern North America, various anthropogenic factors have been shown to adversely influence the health and/or reproductive success of Common Loons. These include direct effects — such as mercury poisoning from point source contamination of watersheds (Barr 1986); lead poisoning from sinker/jig ingestion (Scheuhammer and Norris 1995; Twiss and Thomas 1998); drowning in fishing gear and fatal trauma caused by collision with motorboats and personal watercraft (Miconi et al. 2000) — and indirect effects — from lake acidification, which both diminishes food supply for loons and is associated with increased mercury concentrations in fish (Alvo et al. 1988; Scheuhammer and Blancher 1994; McNicol et al. 1995a); and cottage development, which may remove shoreline breeding habitat and increase disturbance by human activity (Heimberger et al. 1983).

While the effects of these environmental stressors have been documented — and there is no doubt that they can cause direct mortality, increased susceptibility to other stressors, and/or reduced reproductive success in loons — relatively little is known about their population-level effects. In areas where the relative importance of different mortality factors has been assessed, poisoning from ingestion of lead sinkers or jigs has often been demonstrated to be a major cause of death for adult loons on their breeding grounds (Pokras et al. 1993; Scheuhammer and Norris 1995). Relatively long-term studies indicate that 22–53% of reported adult loon mortality in eastern North America is attributable to lead sinker and jig ingestion (section 3). Indeed, in New England, on the southern edge of the Common Loon's breeding range, loon population numbers are low and lead poisoning incidence is particularly high; adult mortality from lead poisoning is suspected to be a potential contributing factor limiting population growth (D. Major, U.S. Fish and Wildlife Service, pers. commun.).

One obstacle to the goal of evaluating population effects of lead poisoning or other environmental stressors in Common Loons is a lack of understanding of the appropriate spatial and temporal scales at which loon populations are regulated. What defines a Common Loon "population" genetically is undetermined, but research to elucidate this is ongoing, using DNA markers (D. Evers, BioDiversity Research Institute, pers. commun.). Once known, such information, coupled with information on dispersal and reproductive parameters using individually marked birds, would allow us to better define populations; to assess the degree of interchange among populations; and to determine the importance of source–sink effects, where individuals from healthy populations with high productivity (sources) disperse into less optimal habitats (sinks) in which productivity does not compensate for adult mortality (Pulliam 1988; Bernstein et al. 1991; Rodenhouse et al. 1997).

Because loons are long-lived, with low rates of mortality and reproduction (they breed every year, lay two or very rarely three eggs, and live up to 25 years; McIntyre and Barr 1997), there can be a longer lag time between environmental change and response in loon population size than for many other birds, making early detection of stressor effects difficult. However, measures of productivity have been used to provide an indication of the relative health of local or regional loon populations (Barr 1986; Kerekes and Masse 2000). Productivity is a function of a number of variables, including the age (experience) of breeders, health of breeding pairs (including the influence of contaminant exposure), habitat quality (availability of nesting sites, food supply, etc.), weather conditions during breeding (e.g., high water levels can flood nests), and mortality rates during migration and overwintering (if productivity is measured on an annual basis). Determining the effects of environmental stressors on loon populations thus requires spatially extensive, long-term data.

Some research to model the effects of environmental stressors on Common Loon populations has been attempted in the northern United States, at the southern edge of the species' breeding range (Evers et al. 2001). In Canada, long-term monitoring has evaluated the effects of lake acidification on loon reproductive success under the Long-Range Transport of Air Pollutants program (McNicol et al. 1987; Wayland and McNicol 1990), and models have been generated to predict the effects of changing sulphate deposition patterns (Blancher et al. 1992; McNicol 1999). Meanwhile, research in Atlantic Canada has focused on studying the associations between methylmercury exposure and loon productivity (Burgess et al. 1998a,b; Kerekes and Masse 2000). However, little is known about the dispersal, philopatry, site fidelity, mortality rates, and long-term reproductive success of loons anywhere in Canada.

In this section, we report on the spatial distribution and abundance of Common Loons in different regions of Canada; provide estimates of national and provincial

population sizes of loons in Canada; report population trends in different geographical regions using available data sources; and, finally, explore some possibilities for modelling loon populations and predicting population effects of environmental stressors. Specifically, we examine the following questions:

- Which areas (provinces/territories/ecozones) hold the greatest numbers of loons? Where do loons occur at highest density?
- What is the temporal variation in loon populations from these different geographical areas? Is there evidence that loon populations are stable, increasing, or decreasing?
- What level of philopatry do loons show? What are the patterns (distances, directions) of dispersal from natal sites? What is the extent of movement between adjacent lakes? What migration routes do Canadian loons use, and where do loons from specific breeding locations spend the nonbreeding season? (The latter is critical to evaluate mortality factors that act outside the breeding season and to investigate cumulative contaminant loads incurred during the winter.)
- How does the spatial distribution of the loon population match geographical patterns of exposure to known anthropogenic stressors? For example, what proportion of the population is at significant risk from lead sinker/jig ingestion?

Finally, we propose 1) a matrix population modelling approach to determine the effect of individual stressors on Common Loon populations and 2) a spatially extensive approach that uses geographic information systems (GIS) to overlay data on known loon populations and their productivity with the distribution of multiple anthropogenic stressors (e.g., acidified environments, environments with elevated mercury concentrations in fish, and environments that experience high recreational angling pressure).

4.2 Spatial patterns of abundance

Wetlands International has estimated the global Common Loon population at 500 000–700 000 individuals (Rose and Scott 1997), and latest population estimates for Canada indicate a minimum of 544 562 individuals (239 401 territorial pairs; Table 8). Approximately 82% of the North American range and 81% of the western hemispheric range of the Common Loon is in Canada (A. Couturier, Bird Studies Canada, pers. commun.). This means that Canada has by far the greatest responsibility of the world's nations for Common Loon conservation and management. The majority of the Canadian population occurs in two provinces (Ontario and Quebec) and the Northwest Territories (Fig. 17). An additional 30 000–35 000 adults reside in the United States, and a few hundred to 2000 individuals reside in Greenland and Iceland (Evers 2000).

In Canada, Common Loons breed from the tree line south to the Canada–U.S. border. Highest population densities are in the Mixedwood Plains and Boreal Plains ecozones. Although Common Loons do occur in the Arctic (e.g., southern Baffin Island), their densities are much lower in far northern areas than in southern parts of Canada. Much of their range overlaps the Canadian Shield, where there are many large, deep water, oligotrophic lakes with large populations of small fish (Vogel 1997). Loons prefer to breed on large lakes (>5 ha); where lakes are small, multi-lake territories may occur (Evers et al. 2000). In some areas, high densities of breeding loons overlap with populations of humans in the "cottage country" of southern Canada and the northern United States. It is for these regional loon "populations" that the threat of sinker ingestion and poisoning is highest.

Table 8
Estimated number of Common Loon territorial pairs and individuals in Canada (modified from Evers [2000] to include additional information)

Province/territory	Number of territorial pairs	Number of individual adults	Source[a]
Ontario[b]	97 000	232 800	1
Quebec[b]	50 000	120 000	2
Northwest Territories[b]	45 000	108 000	3
British Columbia[b]	25 000	60 000	4
Manitoba[c]	10 000–12 000	28 800	5
Nunavut[b]	5 000	12 000	3
Saskatchewan	1 500–2 000	4 800	6
Nova Scotia[d]	1 200	2 880	7
New Brunswick[d]	1 000	2 400	7
Alberta	1 000	2 400	8
Yukon	200	480	9
Prince Edward Island	1	2	10
Newfoundland and Labrador	Not available		
Estimated total	236 901 – 239 401	574 562	

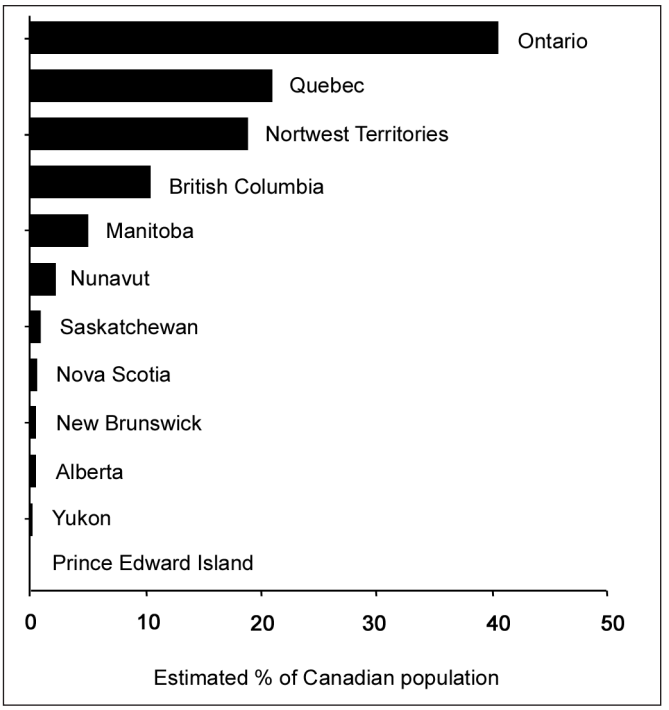
[a] Sources are as follows:
1. Wayland and McNicol (1990).
2. D. Bordage, Canadian Wildlife Service – Quebec Region, pers. commun.
3. J. Hines, Canadian Wildlife Service – Prairie and Northern Region, pers. commun.
4. A. Breault, Canadian Wildlife Service – Pacific and Yukon Region, pers. commun.
5. B. Koonz, Manitoba Department of Natural Resources, pers. commun.
6. A. Brazda, U.S. Fish and Wildlife Service, pers. commun.
7. Erskine (1992).
8. Federation of Alberta Naturalists, pers. commun.
9. Evers (2000).
10. J. Kerekes, pers. commun. to Evers (2000).

[b] Numbers were extrapolated mainly from aerial waterfowl survey data, in which "territorial pairs" of birds were determined during spring and summer. Loons are usually counted incidentally in these surveys and may tend to be underestimated. Because only 80% of the adult loon population actually occupies territories as breeding pairs, a 20% "buffer population" has been added to these estimates (Evers 2000).
In Quebec, annual Eastern Black Duck Joint Venture (EBDJV) helicopter surveys covering an area of 500 000 km^2 counted 25 000 breeding pairs in 1997. Because the survey focused on dabbling ducks, it is likely that Common Loons were underestimated; therefore, the number of loons was doubled to give 50 000 pairs (D. Bordage, Canadian Wildlife Service – Quebec Region, pers. commun.). This estimate was within the range reported by Evers (2000) (75 000 pairs) and DesGranges and Laporte (1979) (35 000 pairs) for Quebec.
In Ontario, survey plots (2 × 2 km) were located systematically (25 plots to a block with 100-m sides) using a Universal Transverse Mercator (UTM) mapping grid comprising nine block across most of northeastern Ontario between latitude 45° and 48°N (McNicol et al. 1987; Ross 1987; Ross and Fillman 1990).

[c] A coarse estimate based on 30 years of birding experience in the province. (Extrapolations from lake area and sampled loon densities are not appropriate for Manitoba because of the many large lakes with no loons.)

[d] Estimates for Nova Scotia were based on assuming 1–2 pairs of loons in each 10 × 10 km^2 square surveyed that had confirmed or probable breeding evidence during the Maritime Breeding Bird Atlas survey, plus a small fraction added to account for unsampled squares. Prince Edward Island had no breeding evidence before 1992, but there was one brood record (sent to the Nest Records Scheme) a year later (A.J. Erskine, Canadian Wildlife Service – Atlantic Region, pers. commun.).

Figure 17
Percentage of Canadian Common Loon population in different provinces/territories



According to the Breeding Bird Survey (BBS), the highest loon densities in Canada are found in northwestern Ontario, west-central Manitoba, and central British Columbia (Fig. 18). These data must be viewed with some caution, because the BBS has limited coverage for many species (including loons) due to the low density of routes in northern and other remote areas. Common Loon abundance is probably underestimated through the BBS due to generally poor BBS coverage of the boreal forest.

During the Maritimes Breeding Bird Atlas (1986–1990), Common Loons were recorded in 506 (33.1%) of 1529 surveyed squares (10 × 10 km squares); flightless young were recorded in 185 (12.1%) squares, and nests in 47 squares (3.1%) (Erskine 1992). Erskine (1992) indicated that loons were less frequent in areas where the underlying rock was sedimentary (eastern New Brunswick, northern Nova Scotia, Prince Edward Island) because of the absence or scarcity of lakes suitable for them.

In southern Quebec, Common Loons were reported in 998 (40.5%) of 2464 atlas squares; of these records, 451 (45.2%) were of possible breeding, 350 (35.1%) of probable breeding, and 197 (19.7%) of confirmed breeding (Alvo 1995). Most loons occur in the Laurentians, Abitibi Uplands, and Anticosti Island, where large water bodies are abundant (Alvo 1995). Aerial surveys conducted in 1990–1992 indicated that there were 2–16 pairs/100 km^2 in the Laurentian and Abitibi areas (D. Bordage, CWS – Quebec Region, unpubl. data). Densities of breeding pairs in Quebec were estimated by DesGranges and Laporte (1979) to be 10 times greater south than north of the 50th parallel in the Laurentians and Appalachians. In northern areas of Quebec, loon populations are limited by the high proportion of lakes lacking fish and/or having high turbidity (DesGranges 1989).

In Ontario, loons are abundant within the Canadian Shield of the Great Lakes–St. Lawrence basin and boreal forest regions (Dunn 1987). Within the BBS coverage area, they reach their highest abundance in the Lake of the Woods area (Fig. 18). Loons are now essentially absent from the Carolinian forest zone where they previously bred (Dunn 1987). During the Ontario Breeding Bird Atlas (1981–1985), Common Loons were reported in 1007 (55%) of 1824 survey squares. Breeding was possible in 21% of squares, probable in 34%, and confirmed in 45%. In northeastern and central Ontario, the highest densities of loons, determined from waterfowl surveys, were in the Chapleau area (22 pairs/100 km^2) and Gogama area (21 pairs/100 km^2) and the lowest in the Sault Ste. Marie area (4 pairs/100 km^2), with an overall average density of 13 pairs/100 km^2 (McNicol et al. 1987). A footnote to Table 8 gives a summary of survey methodology.

Loons are abundant in the area around Flin Flon, Cranberry Portage, Snow Lake, and La Pas in Manitoba, where lakes are underlain by limestone and were covered by Glacial Lake Agassiz. These lakes have relatively high pH, numerous islands and beach ridges, and large fish populations (see Yonge 1981). Yonge (1981) found an average population of about 100 pairs, or 1 pair/40 ha, at Hanson Lake, Saskatchewan (80 km west of Flin Flon), and considered this typical of lakes in the region. In addition, large numbers of nonbreeding loons apparently flock in these areas and would probably be counted on the BBS. However, loons are notably absent from many large lakes in Saskatchewan, because 1) lakes are too large and shallow with excessive wave action, lake-level fluctuations, or turbidity; 2) the limestone bedrock does not provide sites with suitable elevation or vegetation cover for nests; or 3) lakes in this region may have poor fish stocks or fish populations inaccessible to loons (B. Koonz, Wildlife Branch, Manitoba Department of Natural Resources, pers. commun.).

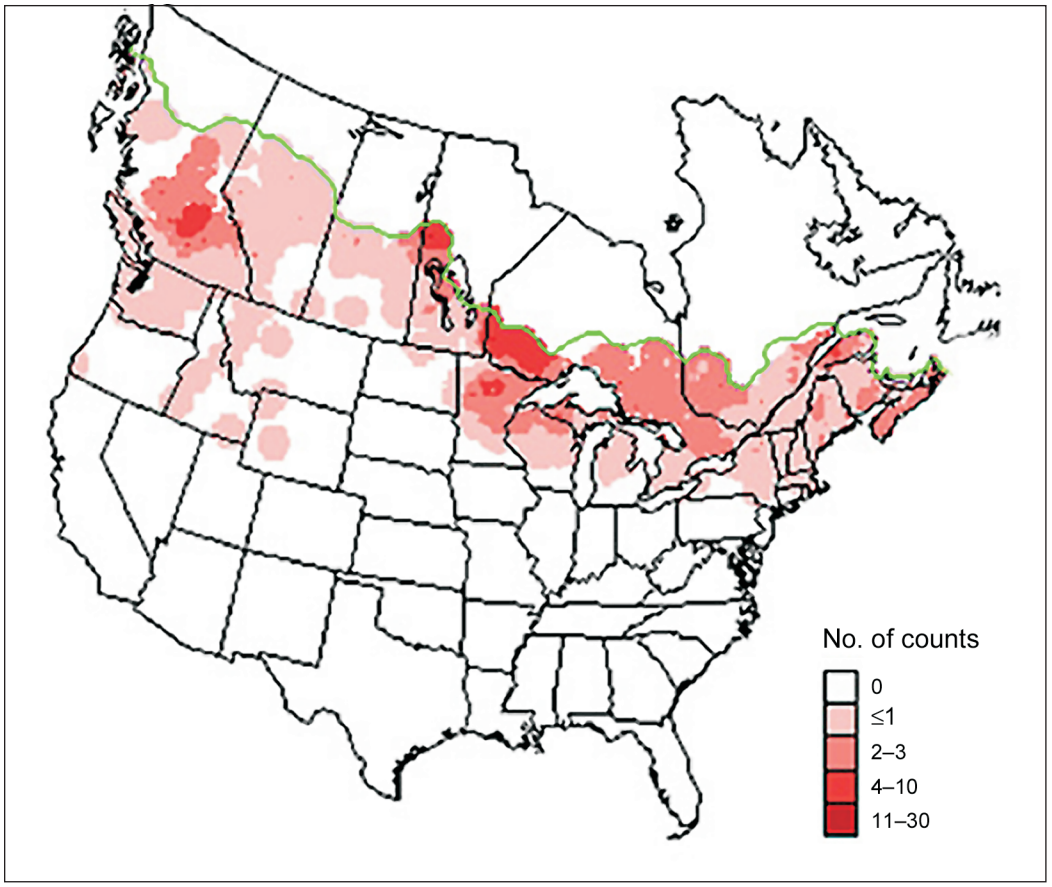
Similarly, Common Loons are absent or extremely rare and localized in the prairie potholes of southern Manitoba, Saskatchewan, and Alberta (as shown by BBS; Sauer et al. 2000) because of the shallow depth of water bodies, poor fish stocks, and high intensity of human/agricultural activity in this area (Vogel 1997; D. Nieman, CWS – Prairie and Northern Region, pers. commun.).

In Saskatchewan, loons were recorded as breeding in 249 of 724 (34%) surveyed squares; breeding was possible in 16%, probable in 12%, and confirmed in 6% of squares. The species is a "common summer resident of northern Saskatchewan, south to Redberry Lake, the Yorkton region, Nickle Lake and Moose Mountain" (Smith 1996). The Saskatchewan atlas, however, was based on historical records and some field observations and is not comparable to the intense surveys conducted over 5- to 10-year periods in some of the other provinces.

In Alberta, most records of breeding Common Loons were in the boreal forest and parkland; very few records were from the grassland zone (Semenchuk 1992). During the Breeding Bird Atlas, loons were reported in 760 of 2206 squares surveyed (34.5%) in Alberta. Breeding was recorded in 26.7% of squares; of these, breeding was possible in 29.8%, probable in 30.3%, and confirmed in 40.0% of squares.

In British Columbia, the highest abundance of Common Loons, according to the BBS and breeding records, is in the Thompson–Okanagan and Fraser plateaus and the Fraser River basin (Campbell et al. 1990). In the Yukon,

Figure 18
Geographical distribution and relative abundance (number of counts per 50-stop route) of Common Loons during summer, 1982–1996, based on Breeding Bird Survey (BBS) data (Sauer et al. 2000). Green line indicates northern limit of survey.



Common Loons are most abundant in the south, where confirmed breeding has occurred at the head of the Stewart River, in the Chapman Lake area in central Yukon, and in Old Crow Flats in the north. Scoby Lakes near the Coal River in southeast Yukon had a high number of nests in June 1986 (Birds of the Yukon database, CWS Pacific and Yukon Region, unpub. data).

The Christmas Bird Count (CBC), a volunteer program run in North America since 1959, indicates that most loons winter on the Atlantic, Pacific, and Gulf coasts of the United States (Fig. 19).

4.3 Population trends

Some evidence exists to suggest that Common Loons have retreated from parts of their former range, particularly their southern breeding limits (McIntyre and Barr 1997). In Ontario, loons have been extirpated south of 43°30'N latitude (Peck and James 1983), largely because of agriculture and urbanization (Dunn 1987). Although most of their historic range is occupied in the Maritimes, Erskine (1992) estimated that loon numbers have been reduced by 33–55% since pre-European settlement times.

Relatively few data exist on long-term trends for Common Loon abundance or reproductive parameters in Canada. Sources of data on loon population parameters include the Canadian Lakes Loon Survey (CLLS) (McNicol et al. 1995a; http://www.bsc-eoc.org/cllsmain.html); aerial waterfowl surveys, especially as part of the North American Waterfowl Management Plan joint ventures (various dates); the BBS 1967–1998 (Dunn et al. 2000); the CBC (Sauer et al. 1996); and Étude des Populations d'Oiseaux du Québec (ÉPOQ) (J. Larivée and A. Cyr, l'Association québécoise des groupes d'ornithologues, pers. commun.). In addition, a few long-term studies have evaluated population parameters and trends in specific locations, including Kejimkujik National Park, Nova Scotia; the Lepreau area, New Brunswick; and La Mauricie National Park, Quebec (Burgess et al. 1998a,b; Kerekes and Masse 2000).

For the Atlantic provinces, spring helicopter survey data are available for breeding loon pairs in New Brunswick, Nova Scotia, and Newfoundland and Labrador. Using the data 1990–1999; no significant trend in loon breeding densities was observed in New Brunswick, but there was a significant decline in adult loon numbers in Nova Scotia (N. Burgess, CWS – Atlantic Region, pers. commun.). In Kejimkujik National Park, Nova Scotia, no appreciable trend is apparent. Numbers of adults are unchanged, and, although fledging success declined for a few years (Kerekes and Masse 2000), it has recently returned to the level observed in the 1980s, an appreciably lower rate (~0.3 large young per resident pair) than the average for eastern North America (~0.5–0.6 large young per resident pair) (J. Kerekes, CWS – Atlantic Region, pers. commun.).

According to the latest ÉPOQ analyses (J. Larivée, l'Association québécoise des groupes d'ornithologues, pers.

Figure 19
Winter distribution and abundance (number of counts per 50-stop route) of Common Loons in North America, estimated through the Christmas Bird Count (CBC) (Sauer et al. 1996)

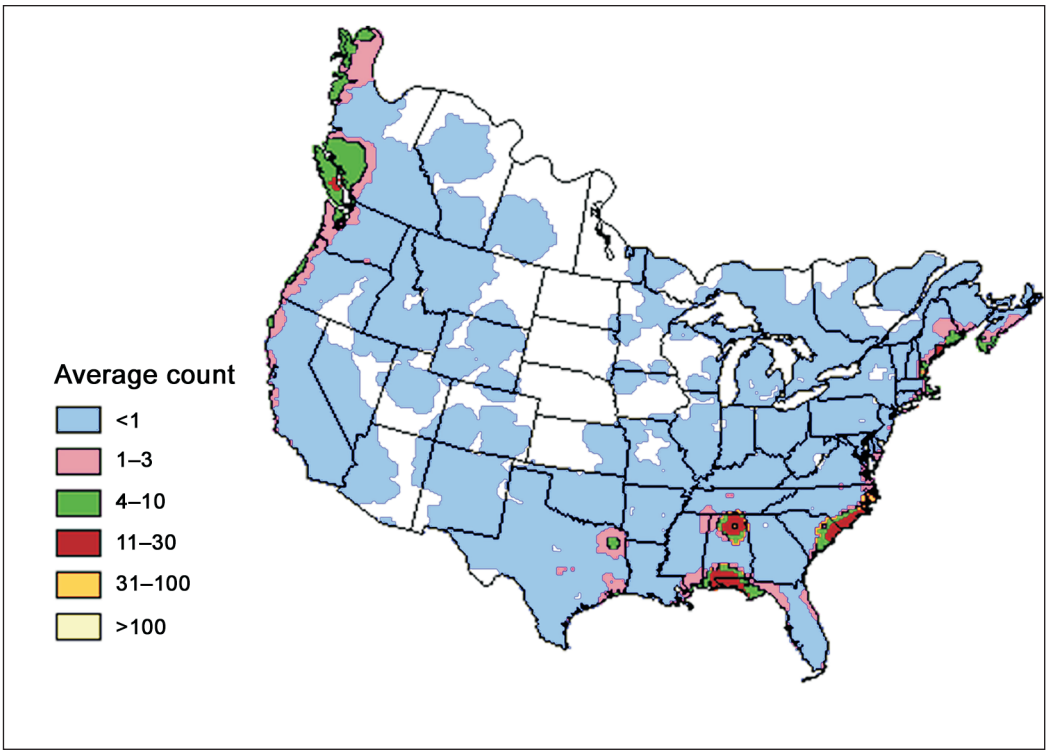
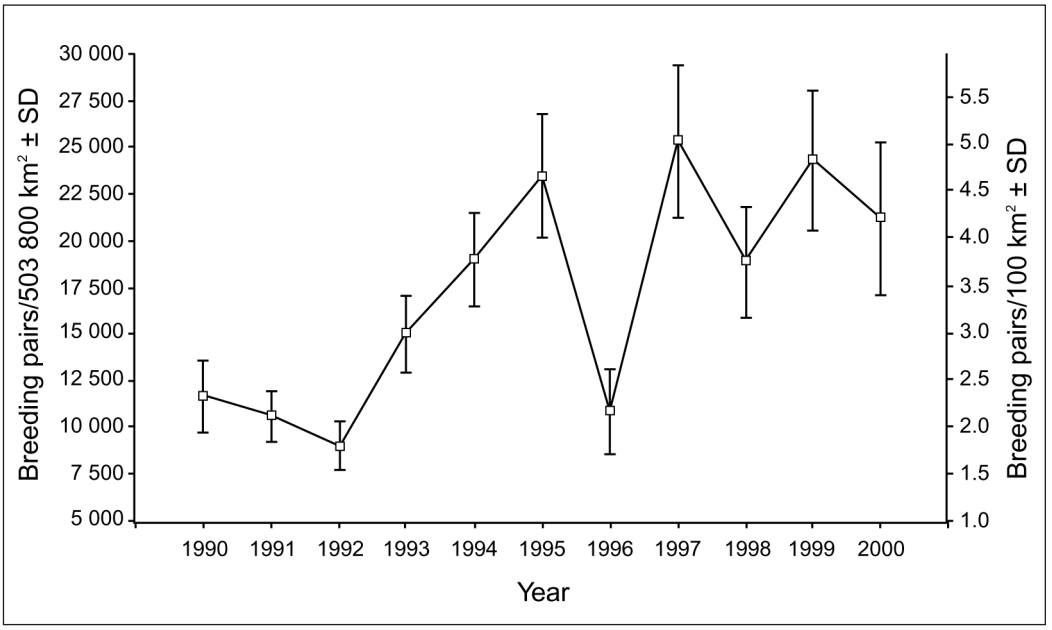


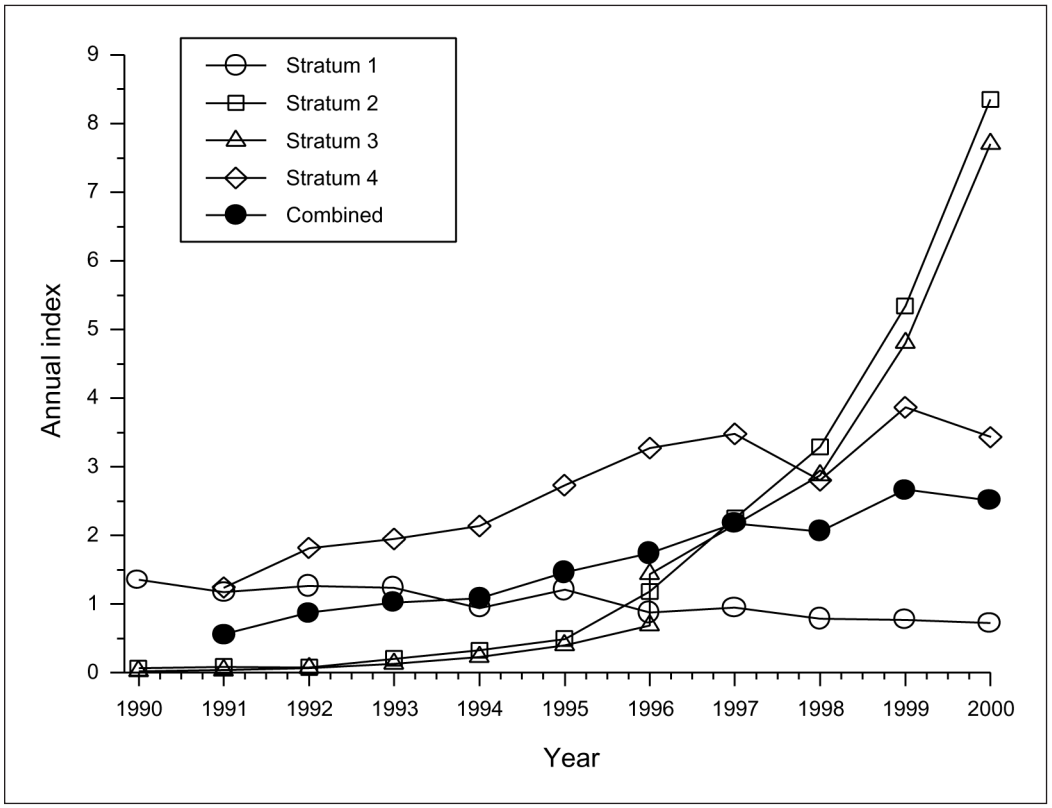
Figure 20
Counts of Common Loons in Quebec, 1990–2000, during Black Duck Joint Venture helicopter surveys



commun.), loon populations in Quebec were stable over the period from 1980 to 2000. This conclusion is substantiated by counts of loons made during helicopter surveys of breeding Black Ducks in Quebec, which indicate stable or increasing loon numbers (LePage and Bordage 1998; L. Champoux, CWS – Quebec Region, pers. commun.; Fig. 20).

In Ontario, loon numbers appear to be stable or increasing overall, based on data from waterfowl surveys (Ross 2002). However, analysis of temporal patterns in Common Loon breeding productivity indicates a different trend (Jeffries et al. 2003). Loon breeding success data were collected by CWS staff or CLLS volunteers from 292 lakes in three CWS Acid Rain Biomonitoring Program study areas in central Ontario (Jeffries et al. 2003) between 1987 and 1999. When the effects of lake area and pH were taken into account, there was a significant negative trend over this

Figure 21
Estimated Common Loon population trends in eastern Canada, from annual surveys of indicated breeding pairs (Collins 2000). Stratum 1 = Atlantic highlands (Nova Scotia, New Brunswick, southeast shore of Quebec); Stratum 2 = Eastern boreal (Newfoundland, southern Labrador, northeast shore of Quebec); Stratum 3 = Central boreal (eastern Quebec boreal forest); Stratum 4 = Western boreal (western Quebec boreal forest plus Ontario boreal forest) (adapted from Collins 2000).



period in the proportion of observed pairs with at least one chick estimated to have fledged. Although these results confirmed the important influence of lake pH on loon breeding success, temporal trends in breeding productivity were similar among all lake pH classes, indicating that factors other than pH alone were involved in the decline in breeding success. Additional information, particularly documentation of the interactions between important population attributes (e.g., demographics, dispersal patterns) and acidity-related and other stressors (e.g., mercury, weather, and climate), is needed to reconcile current observations of stable or increasing adult populations and declining breeding productivity. The analysis of CLLS data indicates that even 10-year periods may not be sufficiently long to allow a confident identification of trends in loon reproductive parameters and that there can be rather large yearly variation in loon productivity.

Recent analysis of breeding survey data for all of eastern Canada (Fig. 21) indicates an overall increasing trend in the number of loon pairs over the period 1990–2000, with an overall rate of increase of 16.6% per annum ($P < 0.05$; Collins 2000). Significant increases were also found within most individual geographical strata (Stratum 2 [Eastern boreal (Newfoundland, southern Labrador, northeast shore of Quebec)], 67.5% annual increase; Stratum 3 [Central boreal (eastern Quebec boreal forest)], 78.0% annual increase; and Stratum 4 [Western boreal (western Quebec boreal forest plus Ontario boreal forest)], 11.9% annual increase; Fig. 21). Only a single stratum (Stratum 1 [Atlantic highlands (Nova Scotia, New Brunswick, southeast shore of Quebec)]) showed a significant decreasing population trend (5.9% annual decrease; Fig. 21). Overall, the average breeding density (number of indicated breeding pairs per 100 km^2) for Common Loons in eastern Canadian surveys ranged from a low of 6.4 in 1991 to a high of 15.4 in 1997 (Collins 2000). Analogous data for western Canada are scarce; apart from incidental counts of loons made during waterfowl surveys, relatively little is known about loon population numbers or trends in western Canada.

Another source of data from which loon population trends can be derived is the BBS, although again it must be cautioned that the BBS has limitations for monitoring trends in aquatic species such as loons. Nevertheless, all results from this survey, whether at the national or at the ecozone level, indicate that Canadian Common Loon populations are stable or increasing (Table 9; Fig. 22). For Canada overall, there was a marginally significant increase ($P < 0.15$) over the long term and a significant increase ($P < 0.05$) over the last 10 years (Table 9; Dunn et al. 2000). There was no evidence of declining populations within any ecozone. Marginally significant increases occurred in the Boreal Plains ecozone (1989–1998), and a significant increase occurred in the Pacific Maritime ecozone from 1967 to 1998.

The CBC (Sauer et al. 1996) cannot provide trend estimates for Common Loons, because most individuals winter offshore and are not recorded by land-based observers. In Canada, the highest concentrations of loons on

CBCs are in western British Columbia and in southern New Brunswick and Nova Scotia.

Loons are increasing in some northeastern parts of the United States where they were formerly locally extirpated (e.g., areas in New England such as southern New Hampshire and Vermont) and in the northwest (parts of northeastern Washington and the Idaho Panhandle). In some other states, previously documented declines have continued (e.g., some areas of Michigan; Evers 2000).

In summary, there is little evidence to suggest that loon populations are generally declining in Canada. However, available data are not sufficiently robust to state this with confidence, especially given that very little adequate population monitoring of Common Loons has been undertaken. Regional or local reductions in productivity and/or declining breeding densities have been documented in some areas of eastern Canada.

Table 9
Population trends in different Canadian ecozones derived from the Breeding Bird Survey (Dunn et al. 2000)

	1967–1998		1989–1998	
Ecozone	Trend[a]	n[b]	Trend[a]	n[b]
All of Canada	1.7**	289	5.0*	198
Boreal Shield	0.3	109	2.4	70
Atlantic Maritime	3.3	54	10.6	34
Mixedwood Plains	12.6	17	14.7	15
Boreal Plains	4.3	39	8.8**	32
Pacific Maritime	13.4*	16		
Montane Cordillera	3.6	41	3.0	30

[a] % annual change; ** indicates significant at $P < 0.15$; * indicates significant at $P < 0.05$.
[b] n = total number of BBS routes used to calculate trend.

4.4 Population modelling

Wildlife population modelling has become extremely popular and sophisticated over the last two decades (Clobert and Lebreton 1991; Lebreton et al. 1992; McDonald and Caswell 1993), and there are now numerous software programs specifically designed for modelling and decision-making (e.g., for American Black Duck *Anas rubripes* management: http://fisher.forestry.uga.edu/blackduck/software.html). Matrix modelling involves complex algebra and will be only briefly outlined here; for a full review, see Caswell (1989) and McDonald and Caswell (1993). Many different versions of models exist, and these may be unstructured, stage-structured, or age-structured. A well-known age-structured model suited to many avian species is the Leslie matrix (McDonald and Caswell 1993). Alternatively, rather than simply classifying individuals by age, stage-structured models allow the researcher to define classifications by many variables of biological interest, including spatial location, social status, habitat quality, and stage of development. Matrix models include sensitivity analysis — that is, they allow the investigator to determine objectively which life history parameters are most important from an ecological or evolutionary perspective; field research can then be appropriately focused on these parameters (McDonald and Caswell 1993). Matrix models can be constructed using the life cycle graph and include stochastic variation and density-dependent nonlinearities necessary for simulation modelling.

Simulation modelling has been carried out for many avian species — in North America, perhaps most notably for the Northern Spotted Owl *Strix occidentalis* (Franklin et al. 1996; Raphael et al. 1996) and Florida Scrub Jay *Aphelocoma coerulescens* (Fitzpatrick and Woolfenden 1989), as well as for several endangered species (see McDonald and Caswell 1993). For long-lived species, such

Figure 22
Trends (mean percent change per year over the period 1966–1996) in Common Loon numbers estimated from Breeding Bird Survey data (Sauer et al. 2000). Data indicate regional population increases over most of the loon's Canadian range. Green line indicates northern limit of survey.

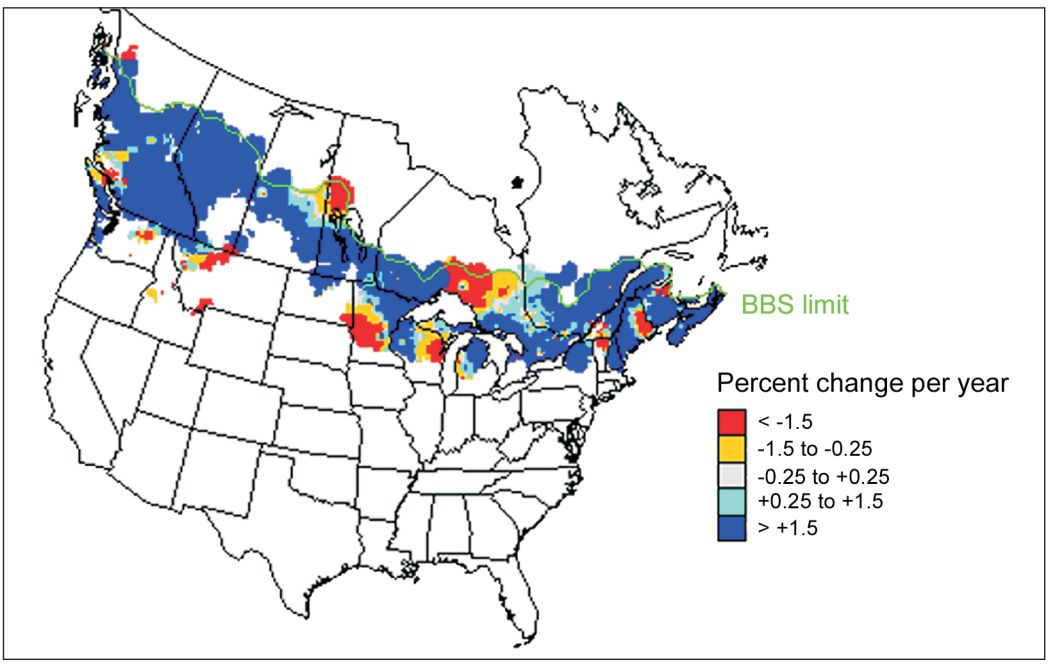
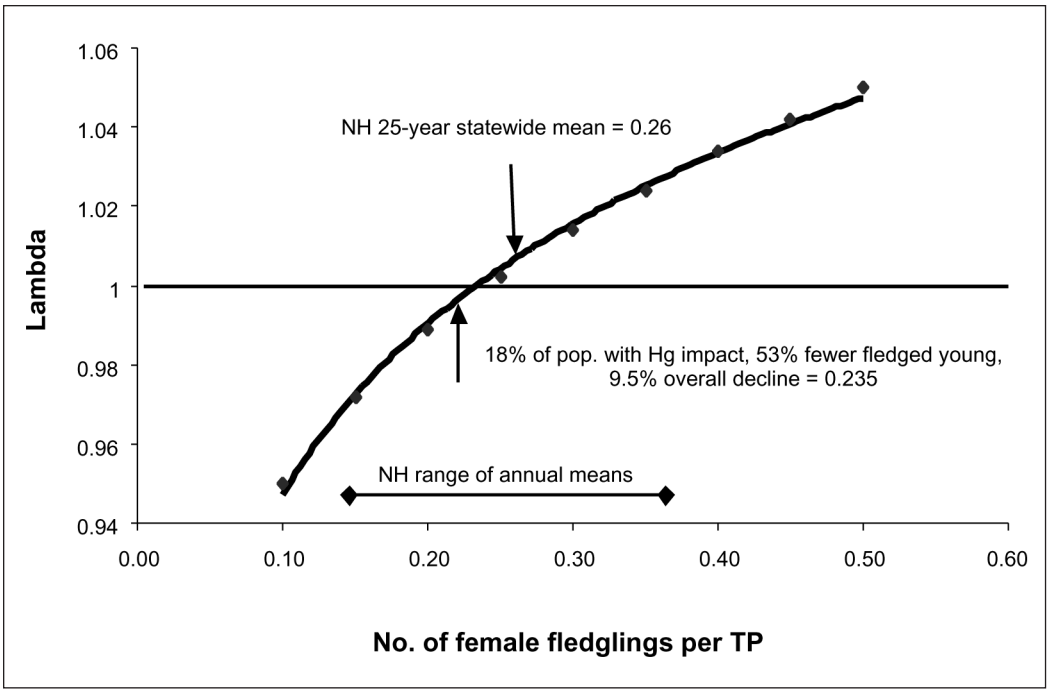


Figure 23
Population growth rate (lambda) versus reproductive success in Common Loons in New Hampshire (from Evers et al. 2001). TP = territorial pair



as the Common Loon, an age-structured model is most appropriate and was used by Evers et al. (2001) to model the effects of mercury on loon populations in New England. Incorporating the effects of environmental contaminant exposure (e.g., lead sinker ingestion) into such models is essentially no different from examining the effects of any density-independent mortality factor (e.g., effects of hunting mortality on game bird populations; Johnson and Williams 1999).

Recent attempts have been made to model the effects of contaminant stressors from the level of the individual to the population level for a few avian species (Walker et al. 1996). For example, Sibly et al. (2000) examined the effect of the cyclodiene pesticide, dieldrin, on population growth rates in Eurasian Sparrowhawks *Accipiter nisus* using life history data from two areas in Britain and were able to demonstrate that acute dieldrin poisoning increased the instantaneous mortality rate by 0.20/year (from 0.48/year — the "natural" mortality rate — to 0.68/year) causily a population decline of 20%/year (λ = 0.82/year; Scenario 1). In Scenario 2, which combined both lethal and sublethal effects of dieldrin, a decline of 60%/year was indicated. In the latter scenario, density-dependent effects (population growth rate increased linearly as density decreased) would function to maintain the population at 64% of its previous level. Sibly et al. (2000) also asked what long-term dieldrin-induced reductions in fecundity or survival could be sustained by an increasing population and concluded that the population could withstand a halving of fecundity (from 2.07 to 1.04) and a reduction in adult survival from 0.74 to 0.55.

Clearly, there are too many unknowns, in the case of loons in Canada, to do the kind of analysis that Sibly et al. (2000) have done for Eurasian Sparrowhawks in Britain. The proportion of the British sparrowhawk population having highly elevated (>9 µg/g) levels of dieldrin in the liver could be confidently estimated, because birds were being actively sampled over many years and their dieldrin concentrations were being measured and recorded. Similar monitoring for elevated lead exposure in loons has not been undertaken so that the proportion of the Canadian loon "population" having elevated lead exposure is unknown. Even if blood lead monitoring were being done, the data would not be useful for determining lead sinker ingestion rates, because lead sinker poisoning is acutely toxic. Either none of a sample of loons will have ingested a sinker, and thus all will have uniformly low blood lead concentrations, or a number of individuals will have ingested a sinker and rapidly perished, almost certainly not being included in blood sampling surveys. There are no confident estimates of the total number of loons that die annually of lead sinker poisoning or of the rates of lead sinker ingestion for any loon population in Canada.

Evers et al. (2001) modelled the effects of dietary mercury exposure on loons in New Hampshire using 25 years of population data collected on 1276 adult and 955 juvenile loons and found that only 133 lakes out of more than 700 suitable lakes were occupied by loons in New Hampshire. One factor limiting population expansion may be mercury exposure. Loons using habitats where they were at high risk for elevated mercury exposure had 15% fewer nests, 51% fewer eggs, and 45% fewer young than those using "low-risk" habitats. Based on three main population parameter estimates (annual adult survival, 95%; annual juvenile survival, 60%; and average age at first breeding, 7 years), Evers et al. (2001) used Leslie population matrices to model the effects of mercury on New Hampshire loon populations (Fig. 23). They also used elasticity analysis to simulate how population growth rate might respond to a proportional change in a particular life history trait. This modelling indicated that once 18% of the loon population was affected by mercury, 53% fewer young would survive,

and the overall population would decline by 9.5%/year (Fig. 23). Modelling was performed using RAMAS software (Akçakaya et al. 1999).

In order to model the response of a local or regional loon population to a specific environmental stressor, such as lead sinker poisoning, at least five population parameters, and the effects of the stressor on each, need to be ascertained: 1) fertility (reproductive success), 2) the age-specific proportion of breeders, 3) immigration and emigration (dispersal) rates, 4) adult/juvenile survival rates, and 5) population density (Clobert and Lebreton 1991).

Reproductive success (the average number of fledged young per territorial pair) can be estimated based on several studies of varying duration in Canada and the United States. For example, the average number of chicks per territorial pair was 0.57 (range 0.20–1.00) in La Mauricie National Park in Quebec (Kerekes and Masse 2000), 0.54 in the Hanson Lake area in Saskatchewan (Yonge 1981), 0.40 in north-central Alberta (Gingras and Paszkowski 1999), and 0.28 in Kejimkujik National Park in Nova Scotia (Kerekes et al. 1995). In New Hampshire, an average of 68% of territorial pairs attempt nesting, and there were 0.52 young fledged per territorial pair, based on a 22-year standardized, statewide database (Taylor and Vogel 2000). For northern Michigan, Wisconsin, and Minnesota sites, reproductive success ranged from 0.51 to 0.79 per territorial pair (Evers et al. 2000).

Little information exists on the age-specific proportion of breeding loons, even for intensively studied sites in New England; therefore, for modelling purposes, the probability that an adult that has not bred previously will breed in a given year must be estimated (Nur and Sydeman 1999). In U.S. studies, age at first breeding ranged from 4 to 11 years, with a mean of 7 (based on 32 marked loons). It is also important to determine the proportion of the potential breeding population that are nonbreeders (Nur and Sydeman 1999). In northwestern Ontario, Croskery (1990) found that the number of young fledged annually from 254 active territories ranged from 57 to 67 (22–26%) (1983–1986), with >80% of successful reproduction coming from only 76 territories, indicating that a large proportion of the breeding population was typically unsuccessful (Croskery 1990). Similarly, Taylor and Vogel (2000) estimated that unsuccessful territorial or nonterritorial adults may comprise up to 46% of the summer loon population.

Dispersal data have been collected for loons in some U.S. sites. Dispersal distances were 1–11 km for breeding adults and 13–96 km for juveniles; and returning adults banded as juveniles established territories 1–64 km from their natal lakes (Evers et al. 2000, 2001). Territorial fidelity (another measure of immigration/emigration) averaged >80% for U.S. (northern Michigan, Wisconsin, Minnesota) loons, but varied according to whether territories were partial-lake, multi-lake, or whole lake (72%, 76%, and 84% fidelity, respectively). The degree to which loons use different lakes can affect their probability of exposure to contaminants, including lead sinkers, because even adjacent lakes can differ widely in water chemistry (Piper et al. 1997) and in degree of human use, including recreational angling intensity. Data on dispersal provide information on source–sink effects (Pulliam 1988) and the relevance of metapopulation dynamics (Hanski 1994), which is critical for demographic modelling. For example, it is known that emigration of loons from populations in British Columbia and Montana helped natural restoration of a breeding population in Idaho (Evers 2000). However, the degree of interchange and mixing that occurs at different geographical scales is generally not well known.

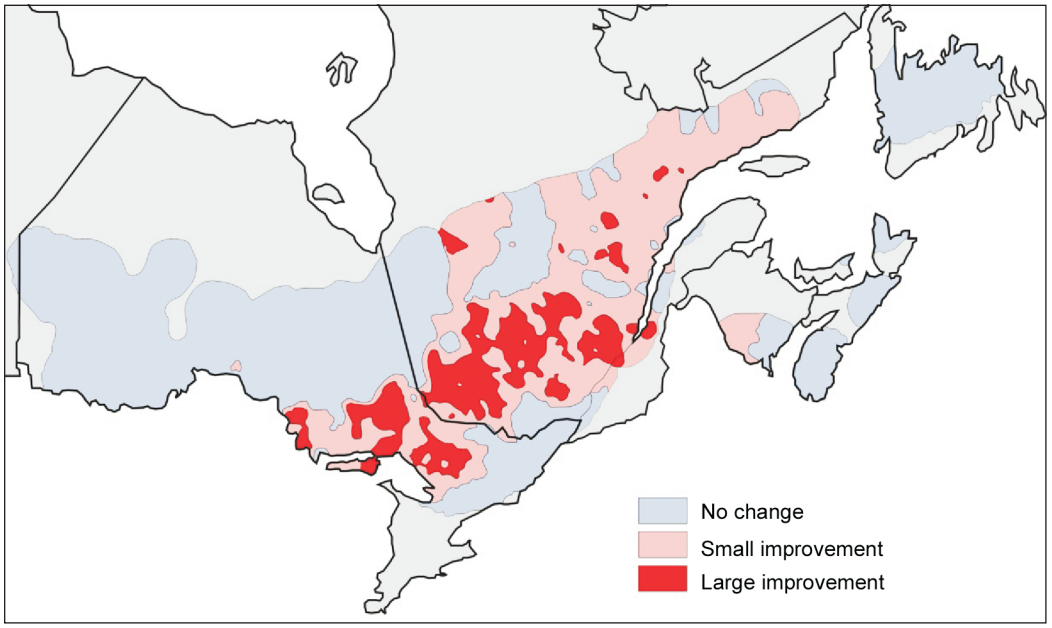
Survival (or mortality) rates can be estimated from recaptures (or resightings) of marked birds and recoveries of banded birds and can be analyzed with software such as SURVIV or MARK (Clobert and Lebreton 1991; White and Burnham 1999). Based on observations of over 600 banded loons in the upper Great Lakes region from 1989 to 1998, Evers et al. (2000) estimated an annual adult mortality rate of 3–4%, which is extremely low compared with those of almost any other bird species.

Lead sinker ingestion is extremely toxic and virtually always results in acute mortality in loons. Sublethal effects have not been reported and are apparently rare (due to the lethal nature of most ingestion incidents), but might predispose any surviving loons to other mortality events, such as collisions with boats or drowning in fishing gear (Miconi et al. 2000). Because sinker ingestion generally has an acutely toxic effect, collecting data on reproductive parameters, such as numbers of young fledged per territorial pair in areas where loons are exposed to lead, would not be particularly helpful; rather, it would be necessary to construct a matrix population model that included lead poisoning as a mortality factor. To accomplish this, accurate estimates of lead-induced mortality, as well as other causes of mortality, are necessary. This requires accurate long-term data on adult survival rates, which can be obtained only through long-term surveys of banded individuals. Concurrently, comprehensive records of all lead sinker (and other) mortality events would need to be kept. If lead poisoning mortality were additive to the natural mortality rate, then annual survival, S, would be:

$$S = e^{-(L+N)}$$

where L is the lead sinker-induced and N the natural instantaneous mortality rate. Additional terms describing mortality rates from other important anthropogenic sources (e.g., gunshot wounds, collisions with boats, oiling, etc.) would then need to be incorporated into this equation. With current data, it is not feasible to accurately estimate mortality from lead sinker ingestion, or any other specific cause, in Common Loons in Canada. To fill this information gap, studies to intensively monitor mortality/survival rates in groups of marked birds, coupled with accurate documentation of the relative importance of different causes of death in loons in specific areas identified for research in Canada, would be needed. Researchers should focus on gathering the necessary population parameters under a few different environmental conditions; for example, a healthy loon population in an area with little or no exposure to anthropogenic stressors (e.g., parts of northwestern Ontario) might be compared with a population using otherwise similar lakes that are exposed to high recreational angling pressure. Furthermore, it may not be sufficient to determine overall loon productivity at various study sites, because some adult loons may contribute disproportionately to annual productivity (Croskery 1990). These birds may need to be identified, because if certain environmental stressors selectively affect these individuals, substantially greater population effects may result than if mainly nonproductive loons were being impacted. Such information is unavailable

Figure 24
Predicted changes in the proportion of habitat suitable across eastern Canada for nesting piscivorous birds (Common Loon plus Common Merganser) between 1982 and 2010, after current sulphate emission control targets are achieved (McNicol 1999)



over most of the Common Loon's range in Canada. Estimating (or modelling) the influence of lead sinker mortality on loon populations requires comprehensive demographic studies to determine needed population parameters.

If more intensive loon population research were to be undertaken in Canada, it might be prudent to focus on areas where long-term data on loon productivity and lake water chemistry and information on some environmental stressors are already available: Kejimkujik National Park, Nova Scotia (Kerekes et al. 1995; Burgess et al. 1998a,b); La Mauricie National Park and Gatineau Park, Quebec (Lane et al. 2000); Algoma, Sudbury, and Muskoka in Ontario (McNicol et al. 1995a,b); and some areas of Alberta (Gingras and Paszkowski 1999). Individual loons have already been marked in some of these areas (Lane et al. 2000), but follow-up studies have generally not taken place. Whether these regions provide an appropriate range of recreational angling pressure would need to be evaluated. Areas that experience little or no recreational angling would be immune from potential impacts of lead sinker ingestion.

4.5 Matching the spatial distribution of loons with geographical patterns of angling pressure — using GIS to model regional effects

A complementary approach to modelling populations is to use a GIS to overlay life history data of interest (e.g., numbers of breeding pairs, reproductive success, mortality) with information on known physical and chemical stressors, such as lake pH; fish mercury concentrations; physical lake attributes, such as surface area, depth, and turbidity; indices of human activity, such as level of cottage development, intensity of recreational angling, and level of motorboat and jet ski use; and pertinent data on weather and fish abundance. Relationships between these various potential stressors and loon population parameters could then be determined. Then, by holding other stressors constant in the model, different scenarios of increased or decreased exposure to a particular stressor (e.g., angling pressure) and its effect on reproductive success or population growth could be simulated.

An analogous approach has recently been used to predict improvements in the suitability of lakes to support Common Loon and Common Merganser nesting pairs following recovery of acidified lakes in southeastern Canada after 2010 sulphate emission control targets are achieved (Fig. 24; Blancher et al. 1992; McNicol 1999). In principle, a data layer for lead sinker density could be integrated into a similar spatial model. If data are not available for the actual distribution and abundance of lead sinkers in lakes, then a layer indicating angling intensity could perhaps be used as a surrogate, assuming a direct relationship between angling intensity and lead sinker abundance in lakes. To develop the model, a contour map representing possible exposure of loons to lead via sinkers could be interpolated and the resulting values spatially associated with data on loon productivity in locations where such data were available.

Good predictive models require a clear understanding of the relationships between changing stressor intensity and population response. Currently, such a level of understanding does not exist for lead sinker distribution or angling pressure (or for most other potential environmental stressors) and loon population dynamics. Perhaps, as a first step, a GIS approach could identify the most "at-risk" areas, where multiple stressors overlap to create habitats in which loons are most likely to be adversely affected. For example, geo-referenced data on recreational angling intensity could be overlain with data on lake acidity, fish mercury concentrations, and other appropriate stressors to which loons are susceptible, to identify environments where loons are most likely to be exposed to hazardous conditions. Field research to study loon population dynamics could then be directed to those locations.

4.6 Summary and priorities for future research

Substantially improved estimates of critical model parameters are needed before the population effects of lead-induced mortality in Common Loons can be assessed with any confidence. To address this issue, long-term monitoring to determine essential demographic data from a few selected study populations in Canada, representing a range of differing scenarios of exposure to lead sinkers and other environmental stressors, would have to be undertaken. Following marked individuals through time is essential to derive accurate population parameters for different subpopulations. Once such needed data were forthcoming, matrix and metapopulation modelling could be carried out using software such as RAMAS (see Akçakaya et al. 1999). Alternatively, accurate loon abundance and productivity data, once gathered and mapped, could be overlain, in a GIS, with accurate measures of recreational angling intensity, to determine if negative relationships exist over time between sinker use and loon abundance or productivity. It is for wildlife managers to decide whether such expensive long-term studies are required before regulatory or other controls on the manufacture, sale, or use of lead sinkers and jigs are undertaken, considering that lead sinker poisoning is a clearly documented, major (and preventable) cause of mortality for breeding loons in southeastern Canada, where recreational angling activity is relatively intense (section 3).

Important analyses that are needed in order to confidently analyze the population effects of lead sinker poisoning in loons includes:

- life history data for loons in Canada, including the most "at-risk" loon subpopulations, using individually marked birds to derive important population parameters;
- DNA analyses to better define "populations" and to better understand source–sink effects and metapopulation structure for Common Loons in Canada; and
- integration of multiple environmental stressors into a large-scale spatial analysis using GIS.

5. Initiatives to reduce the impacts of lead sinkers and jigs on wildlife and the environment

5.1 Development of lead-free fishing sinkers and jigs

Numerous nontoxic alternative materials exist for the manufacture of fishing sinkers and jigs, including tin, steel, bismuth, tungsten, rubber, and clay. Each of these alternative materials differs with respect to the types of uses for which it is appropriate, based on various factors, including malleability and density. Steel, tin, and bismuth sinkers were among the first lead-free alternative fishing weights available in the United States and Canada; however, a wider variety of nontoxic sinkers and jigs is now being produced. Most of the nontoxic sinker manufacturers are located in the United States, but there are several companies in Canada that have indicated that they produce nonlead sinkers and jigs (Table 10).

Tin, steel, and bismuth sinkers are perhaps the most common alternatives to lead; bismuth is also a relatively common alternative for jig and spinnerbait production. Zinc and brass sinkers are also available; however, metallic forms of these materials are known to be toxic to waterfowl and other birds, although they are less toxic than lead (Grandy et al. 1968; U.S. EPA 1994; Zdziarski et al. 1994). A limited selection of alternative sinkers and jigs is currently available through large retail chain stores and tackle and sporting goods stores in Canada.

Many of the available alternative products are currently more expensive than lead, as a result of higher costs of the raw materials and more complicated manufacturing processes. Table 11 summarizes the approximate retail prices of various fishing weights and gives the estimated cost that Canadian anglers may face when purchasing lead-free products.

5.2 Regulatory and awareness initiatives in Canada

In 1997, Environment Canada and Parks Canada, in a parallel regulatory initiative, prohibited the possession of lead fishing sinkers and lead jigs weighing less than 50 g by anglers fishing in National Wildlife Areas and National Parks, under the *Canada Wildlife Act* and the *National Parks Act*, respectively (Canada Gazette 1997b; Parks Canada 1997). These two regulations are of limited geographic scope (<3% of Canada's land mass; Fig. 6) and are anticipated to affect only about 50 000 out of the estimated 5.5 million recreational anglers in Canada.

Since 1995, various governmental and nongovernmental organizations have conducted lead sinker and jig collection and exchange programs, have distributed literature on the toxic effects of lead tackle, and have promoted the voluntary use of alternative materials. Environment Canada supported several of the early sinker exchanges with funding through its Great Lakes 2000 program. The Ontario Ministry of the Environment's Bay of Quinte Remedial Action Plan's "Take a little lead out" program was a pilot project upon which other organizations have modelled their own sinker and jig exchanges. Several cottage associations have declared their lakes to be "lead-free." In addition, Parks Canada has conducted educational campaigns and established lead tackle collection and exchange sites at many of their National Parks.

In spring 1999, a Private Members' Bill (C-403) requesting Environment Canada to use the *Canadian Environmental Protection Act* to ban the import, manufacture, sale, offer for sale, and use of lead fishing sinkers or jigs weighing less than 50 g nationwide was tabled for discussion in the House of Commons (Bonwick 1999), reiterating the recommendation previously made by the Canadian House of Commons Standing Committee on Environment and Sustainable Development (Caccia 1995). In response to this Bill, Environment Canada agreed to develop a national communications strategy and to work with partners to establish a voluntary cooperative lead sinker phaseout program founded on education and public awareness. As part of this strategy, Environment Canada has developed information items, including a pamphlet and a national web site (http://www.cws-scf.ec.gc.ca/fishing/open_e.html), providing the public with information about the negative effects of lead sinkers and jigs and about various lead-free alternatives.

Little information is available on the level of compliance with the 1997 regulation banning lead sinkers and jigs in National Parks and National Wildlife Areas. However, compliance within Canada's National Parks is thought to be high because of the outreach efforts within the parks and the requirement to purchase a separate angling permit to fish within park boundaries. The large number of anglers in Canada (5.5 million or about 1 in 5 Canadians), coupled with the relatively large proportion of the sinker and jig market arising from the home production of lead fishing sinkers and jigs, provides considerable obstacles for user-based prohibitions. Results of a mini-survey of principal stakeholders

Table 10
Manufacturers of nonlead fishing weights

Company	Location	Type of fishing weight
Canada		
D&D Lures	Windsor, Ontario	Bismuth and tin sinkers and jigs
JackFish Lures Non-Toxic Tackle Co.	Edmonton, Alberta	Bismuth sinkers, jigs, and spinner baits
Jr's Environmental Friendly Clay Sinkers	Collingwood, Ontario	Clay sinkers
Lucky Strike Bait Works	Peterborough, Ontario	Tin sinkers and bismuth jigs
Sourdough Bay Fishing Supplies	Medicine Hat, Alberta	Bismuth sinkers and jigs
Tucker Tackle	Ailsa Craig, Ontario	Bismuth sinkers
United States		
Belvoirdale/Dinsmores	Wyncote, Pennsylvania	Tin sinkers
BIO-CAST	Fruita, Colorado	Ceramic weights
Bullet Weights	Alda, Nebraska	Steel and tin sinkers
Du-Co Ceramics	Saxonburg, Pennsylvania	Ceramic sinkers
Jadico	Camdenton, Missouri	Bismuth jigs
Loon Outdoors	Boise, Idaho	Putty sinkers
Luhr-Jensen and Sons Inc.	Hood River, Oregon	Rubber sinkers
ORVIS Company	Roanoke, Virginia	Tungsten beads, putty weights
Owner Hooks	VonKarman, California	Bismuth sinkers and jigs
SafeCasters	Pasco, Washington	Granite sinkers
Water Gremlin	White Bear Lake, Minnesota	Tin/plastic composite/steel resin sinkers
Britain		
Dinsmores	Walsall, U.K.	Tin sinkers
Sweden		
Eco Weight	Stockholm, Sweden	Magnetite/concrete sinkers

Table 11
Estimated consumer costs associated with the use of lead and alternative sinker products for sport fishing in Canada[a]

Weight type (size)	Approximate average price per lead weight[b] ($)	Approximate average price per lead-free weight[c] ($)	Average yearly expenditure per angler[d] ($)		Average increase per angler per year ($)	Increase in yearly angling supplies budget (%)[e]
			lead	lead-free		
Split shot sinkers						
(assortment)	0.03	0.04 (tin)	0.42	0.56	0.14	0.3
(# 5)	0.05	0.17 (tin)	0.70	2.38	1.68	3.6
(# 7)	0.04	0.11 (tin)	0.56	1.54	0.98	2.1
Bell sinkers						
(¼ ounce)	0.22	0.39 (bismuth)	3.08	5.46	2.38	5.0
Painted jigs						
(¼ ounce)	0.54	0.58 (bismuth)	3.78	4.06	0.28	0.7
(⅜ ounce)	0.47	0.58 (bismuth)	3.29	4.06	0.77	1.8

[a] Where direct comparisons were possible in large retail stores.
[b] Retail price in Canadian dollars.
[c] In addition to the retail prices listed above, steel bass casters were available in large retail chain stores and sold for about $0.18 per sinker. Current retail prices for other alternative fishing weights were not available; however, promotional materials suggest that weights made of clay, steel, and rubber fall within the range of prices listed above for tin and bismuth.
[d] Based on an estimated average of 14 sinkers and 7 jigs purchased per angler per year.
[e] Percent increase in fishing supplies budget of $42.00 per year. The average angler purchases about 14 sinkers per year, at a total current cost of about $3.25 (Scheuhammer and Norris 1995). Based on retail prices of common types of weights, the average angler may be expected to spend an additional $2.00 per year to use nonlead fishing weights.

regarding awareness of the issue and control options supported an approach coupling educational programs with national regulation (Twiss and Thomas 1998). Limited educational initiatives have already been developed and implemented by Environment Canada, including a Hinterland Who's Who publication on lead poisoning (CWS 1996), a pamphlet for distribution at outdoor shows, as well as support for some of the Province of Ontario's sinker exchange programs. However, efforts have been limited, and the majority of anglers continue to use lead.

5.3 Regulatory and awareness initiatives in other countries

In 1987, the United Kingdom banned lead fishing sinkers weighing less than 28.35 g (1 ounce) because of widespread mortality of swans from ingestion of lead sinkers (Birkhead 1982, 1983; Government of the United Kingdom 1986).

The U.S. Fish and Wildlife Service has banned the use of lead sinkers and jigs weighing less than 28.35 g (1 ounce) in three National Wildlife Refuges — namely, Red Rock Lakes, Montana; National Elk Refuge, Wyoming; and Seney, Michigan — under the *National Wildlife Refuge System Administration Act* and has also moved to create lead-free areas at 13 additional wildlife refuges in nine states where loons and anglers coexist. In addition, the U.S. National Parks Service, under the *National Park Services Act*, has banned the use of lead sinkers and jigs weighing less than 28.35 g (1 ounce) in Yellowstone National Park, Wyoming.

In 1994, the U.S. EPA published, in the Federal Register, a proposed rule under the *Toxic Substances Control Act* to prohibit the manufacture, processing, and sale, within the United States, of sinkers containing lead, zinc, or brass that are 2.54 cm (1 inch) or less in any dimension (U.S. EPA 1994). The regulation as initially drafted was not enacted, because many states and angling groups, including the American Sportfishing Association, argued that there was insufficient evidence to warrant a national ban on lead fishing sinkers. The American Sportfishing Association, among others, recommended that regional measures be taken to address this issue — specifically, that the U.S. EPA and U.S. Fish and Wildlife Service evaluate the extent of the problem nationwide, promulgate regulations in areas where there was evidence of a threat, pursue regulations in National Parks and National Wildlife Refuges where a problem has been documented, and initiate educational programs to inform people about the dangers associated with improper in-home production of these products.

During the discussion of the U.S. EPA's proposal to ban lead fishing sinkers and jigs, the Environmental Defense Fund suggested that there was a potential risk to human health from the exposure to lead fumes during home manufacturing of sinkers and from lead ingestion as a result of biting split shot sinkers to crimp them onto fishing lines (U.S. EPA 1994). It was estimated that 875 tonnes of lead may be used annually in the home production of fishing weights in the United States. In the United States, over 40 cases of lead toxicosis in humans as a result of home production and use of lead sinkers have occurred in New York, New Hampshire, North Carolina, and Iowa. In New York state, seven cases of high lead exposure from sinker production were reported between 1988 and 1993. Blood lead levels in all seven individuals exceeded 25 µg/dL, with three individuals having blood lead levels exceeding 60 µg/dL, a level that typically causes noticeable symptoms of lead poisoning in adults (U.S. EPA 1994). Three small children in New Hampshire were exposed in a home where lead sinkers were made and had blood lead concentrations ranging between 27 and 53 µg/dL. Adult family members producing the lead sinkers and cleaning the work areas also had elevated blood lead concentrations. Following transfer of home ownership, the subsequent family's three children also experienced high lead exposure, with blood lead levels ranging between 29 and 42 µg/dL (U.S. EPA 1994). In North Carolina, an outdoor cauldron where lead was melted for sinker production resulted in severe contamination (450 000 µg/g) of soils in the area. U.S. EPA guidelines recommend that children not be allowed access to areas with soil lead levels exceeding 2000 µg/g. Soil lead levels in this instance were more than 200 times the allowable level, and lead levels in dust on outdoor patio areas around the home where the pot was situated were roughly 50 times the level allowed following lead paint abatement. At least 26 children and adults were exposed to lead at this sinker production site (U.S. EPA 1994). In Iowa, two children were found to have elevated blood lead levels associated with biting lead split shot to attach it to fishing line (New Hampshire Department of Fish and Game 1998). In the published medical literature, one case of lead poisoning from sinker ingestion was reported in an 8-year-old child from Ohio. The boy had an elevated blood lead level of 2.6 µmol/L (~54 µg/dL) following ingestion of 20–25 lead sinkers (Mowad et al. 1998). Following removal of the sinkers and chelation therapy, blood lead levels returned to normal.

It has been argued that human exposure to lead from the home production of fishing sinkers and jigs is an issue only if lead is handled or used improperly; however, concerns have been raised about the quality of information available with home sinker production kits. The Environmental Defense Fund argues that safety information provided with do-it-yourself kits generally warns of the risk of burning when in contact with hot objects, but only rarely indicates health risks associated with lead or the need to take preventative precautions that in an industrial setting would be mandated by law (U.S. EPA 1994). We are unaware of any Canadian studies of lead exposure in people through the home production of sinkers and jigs; however, Health Canada has been informed of the cases referenced in the present report.

In the wake of the debate over the U.S. EPA's proposal to ban lead fishing sinkers and jigs, and following the recommendations for regional action, a few northeastern states developed cooperative outreach programs and pursued regulatory measures within their jurisdictions to address the concerns associated with lead fishing sinkers and jigs.

The New Hampshire Department of Fish and Game ratified a state regulation in July 1998 that prohibits the use of lead sinkers and jigs weighing 28.35 g (1 ounce) or less in all freshwater lakes and ponds statewide beginning 1 January 2000 (State of New Hampshire 1998). This action was taken to insure the Common Loon's continued success in the state. The New Hampshire Department of Fish and Game, in cooperation with the U.S. Fish and Wildlife Service and the New Hampshire Loon Preservation Committee, launched a series

of communication and educational initiatives to urge anglers to switch to lead-free sinkers and jigs.

Maine will prohibit the sale of lead sinkers and jigs weighing 14.2 g (½ ounce) or less beginning in the year 2002 (State of Maine 1999) and in the meantime is conducting an educational campaign to inform anglers of the hazards of lead sinkers and has instituted sinker exchange programs.

Massachusetts has instituted state regulations prohibiting the use of lead fishing gear on the small number of lakes where loons are known to nest (M. Tisa, Massachusetts Division of Fisheries and Wildlife, pers. commun.).

In February 1999, a bill (S02592) was introduced in the New York legislature to amend the environmental conservation law to prohibit the sale and use of lead sinkers. This bill was recently passed by the state legislature, and New York will prohibit the sale of most lead fishing sinkers by 2004. The measure also seeks to end the hazardous practice of biting down on lead split shot weights to affix the sinker on fishing line.

In March 1999, a bill was introduced in Minnesota to authorize grants to develop alternatives to lead fishing sinkers and lead jigs (Minnesota House of Representatives 1999). The bill was read and forwarded to the Environmental and Natural Resources Policy Committee for discussion. A cooperative education campaign group comprising wildlife researchers, rehabilitators, veterinarians, and manufacturers and retailers of lead-free tackle has been established (MOEA 1999). A similar cooperative education and sinker exchange program is ongoing in Vermont (M. Pokras, Tufts School of Veterinary Medicine, pers. commun.).

At the federal level, the U.S. Fish and Wildlife Service is currently considering a proposed 2-year phase-in on a ban of lead sinkers and jigs in National Wildlife Refuges across the country where loons and Trumpeter Swans breed (L. Morse, U.S. Fish and Wildlife Service, pers. commun.)

In Sweden, research and awareness programs began in the early 1990s in cooperation with the Swedish Anglers' Association and the National Chemicals Inspectorate to encourage the use of lead-free fishing weights (OECD 1999). A study of the dissolution of lead weights lost when fishing was launched by the National Chemicals Inspectorate in 1994, and a policy was established to phase out the use of lead; however, it is believed that the use of lead sinkers for angling is probably the same as when the policy for phaseout was established, as there is a lack of competitive alternatives on the market. In 1999, the National Chemicals Inspectorate also launched a campaign aimed at minimizing the use of certain sinkers for salmon fishing during spring and autumn in fast-flowing water, as this kind of fishing is considered to result in the highest losses of lead fishing weights. The National Chemicals Inspectorate continues to work with stakeholders to encourage a phaseout of lead fishing weights in Sweden.

The 1996 OECD Declaration on Risk Reduction for Lead (OECD 1996) included a recommendation that member countries restrict the use of lead shot in wetlands and promote the use of alternatives to lead sinkers in shallow waters.

5.4 Summary

- Numerous viable alternative materials exist for use in the production of fishing sinkers and jigs, including tin, steel, bismuth, tungsten, rubber, ceramic, and clay. Tin, steel, and bismuth sinkers and bismuth jigs are the most readily available alternatives in Canada.
- Many of the alternative products on the market are currently more expensive than lead, due to a higher price of the raw materials and more difficult manufacturing processes. Based on available direct retail price comparisons of common types of fishing weights, the average angler is estimated to spend up to an additional $2.00 annually to buy nonlead sinker and jig products.
- Environment Canada and Parks Canada prohibited the possession of lead fishing sinkers/jigs weighing less than 50 g by anglers fishing in National Wildlife Areas and National Parks, under the *Canada Wildlife Act* and the *National Parks Act*, respectively, in 1997. These two regulations are of limited geographic scope (<3% of Canada's land mass) and are anticipated to affect only 50 000 of the estimated 5.5 million recreational anglers in Canada.
- In 1987, Britain banned the use of lead fishing sinkers weighing less than 28.35 g.
- The U.S. Fish and Wildlife Service banned the use of lead sinkers and jigs in three National Wildlife Refuges and in Yellowstone National Park. The states of New Hampshire, Maine, and New York have ratified statewide regulations beginning in 2000, 2002, and 2004, respectively.

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