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# Stress in seabirds: causes, consequences and diagnostic value

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## Abstract

We describe a range of anthropogenic stressors that impact seabirds, review the effects of these stressors on individuals and populations and discuss the role and value of seabirds as monitors of marine ecosystem health. Stressors described are restricted to those which affect seabirds directly or indirectly through the marine environment; we have not dealt with terrestrially based stressors such as introduced mammalian predators or loss of habitat, which can potentially affect seabirds whilst breeding. We discuss three broad categories of stress in seabirds. Marine pollutants (including biologically non-essential heavy metals, oil, organic pesticides and polychlorinated biphenyls (PCBs), and plastics), industrial fisheries (further divided into the effects of depletion of prey stocks and direct mortality), and climate change. Additionally we highlight the role of seabirds as monitors of marine ecosystem health, taking the example of long-term mercury contamination as a case study. We conclude that seabirds are exposed to an increasing array of potential stressors, and that the impact of a particular source of stress on seabirds varies markedly between species in relation to foraging and breeding ecology. The most serious threat to seabirds is direct mortality of adults resulting from industrial and commercial fishing activities. In some cases this is a significant threat to individual populations or even entire species.

## Introduction

Seabirds are a conspicuous and high-profile component of marine ecosystems, feeding at a wide range of trophic levels and in a broad spectrum of marine habitats, from the littoral to the pelagic, throughout the world's marine environment. Their position at or near the apex of many marine food chains make seabirds ideal sentinel organisms with which to monitor changes within marine ecosystems. For example, seabirds have been used extensively to monitor marine pollution and to provide insights into the health of fish and other prey stocks. However, anthropogenic influences, for example through increased emissions of pollutants or through increased exploitation of marine resources, place seabirds under ever-increasing stress.

The suggestion that seabirds suffer a wide range of detrimental anthropogenic effects is not, of course, novel. Indeed, the literature provides numerous documented examples of direct exploitation of seabirds and of alteration or destruction of breeding habitat over many centuries. For example, in a recent review of the past and present status of seabirds in the Azores archipelago in the central, warm temperate Atlantic, Monteiro et al. (1996) described that following human colonisation in the late fifteenth century, 14 alien mammal species were introduced, some of which impacted directly on seabird populations through predation (for example, cat (*Felis catus*), ferret (*Mustela furo*), weasel (*M. nivalis*), and two species of rat (*Rattus* spp.)). Direct human exploitation of seabirds was also pronounced; 10,000 Manx shearwater (*Puffinus puffinus*) were taken for food, oil and feathers nightly, with the result that this species was almost eliminated during the sixteenth century. Today, many seabirds in the Azores are confined to small offshore islands and stacks, their populations are relatively small compared to historical levels and the threat from introduced mammals and from increased development, particularly in the littoral zone, is considerable (Monteiro et al., 1996).

Whilst clearly of immense importance to the survival of many seabird populations in particular, and island ecosystem health in general, the negative effects of introduced terrestrial predators to oceanic islands are beyond the scope of this review of marine stressors on seabirds. For further information on this subject, the reader is directed towards Burger and Gochfeld (1994).

In this review we examine the effects and importance of a range of potential stressors on seabirds, including pollutants (heavy metals, oil, synthetic organic contaminants and plastics), industrial fisheries and climate change. We evaluate the role of seabirds as biomonitors of certain stressors within the marine environment. There is no commonly accepted definition of which avian groups or species fall within the category 'seabirds'. For the purposes of this review, we define seabirds as members of the Procellariiformes (albatrosses, shearwaters, petrels, and prions), the Spheniscidae (penguins), the Pelecaniformes (cormorants, shags, gannets, boobies, pelicans, tropicbirds and frigatebirds), the Alcidae (auks), the Laridae (gulls, terns and noddies) and the Stercorariidae (skuas). Albatross taxonomy follows that proposed by Roberston and Nunn (1997), and skua taxonomy follows that proposed by Cohen et al. (1997).

## Pollution

### *Heavy metals*

Concentrations of heavy metals in seabirds, in particular the non-essential and highly toxic metals cadmium, lead and mercury, have been reported extensively. However, detecting deleterious effects of heavy metals in seabirds has proved difficult. Many seabirds exhibit relatively high concentrations of cadmium and mercury, in particular, with apparently little or no effect. Many concentrations of mercury and cadmium reported in seabirds are far in excess of those known to be harmful to terrestrial birds. A selection of heavy metal concentrations in seabirds is presented in Table 1. This is by no means an exhaustive list of published information, but serves to provide examples of concentrations of mercury, cadmium and lead in a range of species, tissues and locations. Metal data are presented on a dry weight basis. Where published on a wet weight basis, data have been converted to dry weight assuming a tissue water content of 70%.

Relatively high cadmium concentrations have been reported in a wide range of seabirds (Muirhead & Furness, 1988; Honda et al., 1990; Lock et al., 1992; Elliott & Scheuhammer, 1997; Stewart et al., 1999, Table 1). For example, Muirhead and Furness (1988) measured exceptional cadmium concentrations in excess of 330 $\mu$g g$^{-1}$ in kidney tissue from rockhopper penguins (*Eudyptes chrysocome*), Tristan albatrosses (*Diomedea dabbenena*) and Atlantic petrels (*Pterodroma incerta*) from Gough Island, south Atlantic ocean. Elliott and Scheuhammer (1997) reported mean cadmium concentrations up to 306 $\mu$g g$^{-1}$ in kidney tissue from Leach's storm-petrels (*Oceanodroma leucorhoa*) from Canada, and Stewart et al. (1999) reported mean cadmium concentrations as high as 190 $\mu$g g$^{-1}$ in kidney tissue from grey petrels (*Procellaria cinerea*) from New Zealand.

Nicholson and Osborn (1983) reported renal lesions in northern fulmars (*Fulmarus glacialis*), Manx shearwaters and Atlantic puffins (*Fratercula arctica*) from the UK with kidney cadmium concentrations in the range ca. 33–400 $\mu$g g$^{-1}$. However, no evidence of kidney damage was found in similar species with relatively high cadmium concentrations from the north-west Atlantic (Elliott et al., 1992). Furness (1996a), in a review of the toxicological effects of cadmium in birds, suggested that cadmium concentrations in kidney tissue in excess of ca. 330 $\mu$g g$^{-1}$ might be deleterious, although he noted that in seabirds this threshold may be higher. Indeed, with the exception of the study by Nicholson and Osborn (1983), sub-lethal toxicological effects of cadmium in seabirds have proved extremely difficult to detect.

Lead and mercury exhibit relatively high values for the ratio of anthropogenic to natural emissions to the environment. Consequently these metals are considered to pose relatively large pollution threats (Lantzy & Mackenzie, 1979). Mortality of seabirds due to lead toxicity on the Mersey estuary, north-west England was reported by Bull et al. (1983). Among five species of gulls (*Larus* spp.) found dead, black-headed gulls (*L. ridibundus*) were the most commonly recorded species. Lead concentrations in liver tissue from dead gulls were generally in the range ca. 3–33 $\mu$g g$^{-1}$. The authors concluded that the birds had died from alkyl lead poisoning after eating prey contaminated with industrial effluent from petrochemical plants. In contrast, apparently healthy laughing gulls (*L. atricilla*), which were collected from a lead-contaminated area from the Texas coast, had mean lead concentrations in liver tissue of ca. 13–17 $\mu$g g$^{-1}$ and mean kidney concentrations of ca. 7 $\mu$g g$^{-1}$; Munoz et al., 1976; Hulse et al., 1980).

Lead concentrations tend to be relatively low in pelagic seabirds. Elliott and Scheuhammer (1997) reported mean lead concentrations in liver tissue of 0.28 $\mu$g g$^{-1}$ in both Cassin's auklets (*Ptychoramphus aleuticus*) and rhinoceros auklets (*Cerorhinca monocerata*), and up to 0.51 $\mu$g g$^{-1}$ in Leach's storm-petrels. However, a small proportion (27% and 10% in 1993 and 1994, respectively) of Laysan albatross (*Phoebastria immutabilis*) chicks in Hawaii exhibited lead concentrations in liver of over ca. 0.8 $\mu$g g$^{-1}$, and up to ca. 236 (1993) and 167 (1994) $\mu$g g$^{-1}$, respectively, due to ingestion of lead-containing paint chips from buildings. Additionally, blood lead levels were sufficiently high to suppress enzyme activity in some individuals. Conversely, adult Laysan albatrosses exhibited no or minimal exposure to lead (Work & Smith, 1996). Further work is required in order to determine threshold lead concentrations causing toxic effects for seabirds, but for most pelagic species the risk from lead would appear to be relatively low.

Mercury pollution has resulted in widespread fatalities of wildlife, including birds, and in some cases even humans (Fujiki et al., 1972). However, mercury concentrations in seabirds are often difficult to interpret from a toxicological perspective, since values markedly above those known to have deleterious effects (even death) in non-marine birds are frequently measured in apparently healthy individuals showing no adverse effects or reduced reproductive output. In a recent review of the toxic effects of mercury in birds, Thompson (1996) suggested that in non-marine birds, mercury concentrations between ca. 67–100 $\mu$g g$^{-1}$ in liver or kidney tissue were likely to cause toxic effects or death. In contrast, Muirhead and Furness (1988) reported mercury concentrations in liver tissue of Tristan albatross from Gough Island of up to ca. 903 $\mu$g g$^{-1}$. Similarly high mercury concentrations have been reported in albatrosses and other procellariiform seabirds, in particular, from a wide range of locations (Honda et al., 1990; Lock et al., 1992; Kim et al., 1996; Stewart et al., 1999).

It appears that pelagic seabirds are able to detoxify ingested organic, methyl mercury (the predominant form of mercury in prey (Monteiro et al., 1998)), and store the less toxic, inorganic form. This mechanism seems to be particularly important in large, long-lived and slow-moulting species (albatrosses, for example), which can potentially accumulate large amounts of mercury over their life-spans, and which have relatively little opportunity to excrete mercury via the feathers following moult (Thompson & Furness, 1989; Honda et al., 1990; Kim et al., 1996). High mercury concentrations in tissues of pelagic seabirds (see above) may reflect accumulation of the less toxic inorganic form, following demethylation.

There is little, if any, unequivocal evidence of toxic effects, even at the sub-lethal level, of mercury on seabirds. For example, in a study using feathers as a monitoring tissue, Thompson et al. (1991) found no relationship between mercury concentration and a wide range of reproductive parameters or survival in great skuas (*Catharacta skua*) thought to be exposed to considerable stress at the time of the study through acute food shortage (Hamer et al., 1991). However, at least in the northern hemisphere, mercury concentrations have increased during the last century in many pelagic seabird species (Thompson et al., 1992b, 1993a; Monteiro & Furness, 1997; Thompson et al., 1998; see Seabirds as monitors of marine ecosystem health, this paper), and seabirds which feed on mercury-rich mesopelagic prey appear particularly prone to accumulate relatively high mercury burdens (Monteiro & Furness, 1997; Monteiro et al., 1998). The possibility that sub-lethal toxic effects of mercury in seabirds might develop in the future cannot be ruled out.

### *Oil*

Seabirds are particularly conspicuous casualties of oil pollution. Oil floats on the sea surface where it comes directly into contact with swimming birds. Oil can soak birds' plumage, lowering or destroying its insulation and buoyancy such that birds become chilled and sink lower in the water (Holmes & Cronshaw, 1977). Increased effort to try and maintain body temperature and position in the water uses energy reserves. Birds can be poisoned directly through oil ingestion (Peakall et al., 1982; Fry & Lowenstine, 1985). Exposure of avian eggs to oil can cause embryo mortality (Albers, 1978) and dosing experiments with Cassin's auklets revealed that oil caused reduced egg production and hatching (Ainley et al., 1981).

Species which spend relatively large amounts of time swimming and diving for food, auks for example, are especially vulnerable to oil pollution, at least in numerical terms. This pattern of high auk mortality

*Table 1.* Selected cadmium, mercury and lead concentrations in adult seabirds

| Species | Location | Pollutant | Tissue (number analysed) | Mean concentration ($\mu$g g$^{-1}$ dry weight) | Reference |
|---|---|---|---|---|---|
| Rockhopper penguin<br>*Eudyptes chrysocome* | Gough Island | Cadmium | Liver (12) | 46.2* | 1 |
| | | | Kidney (12) | 237.6* | |
| | | Mercury | Liver (12) | 7.6* | |
| Macaroni penguin<br>*Eudyptes chrysolophus* | Antarctica | Cadmium | Liver (9) | 29.7* | 2 |
| | | | Kidney (9) | 161.7* | |
| | | Mercury | Liver (9) | 3.0* | |
| Chinstrap penguin<br>*Pygoscelis antarctica* | Antarctica | Cadmium | Liver (13) | 6.6* | 2 |
| | | | Kidney (13) | 62.7* | |
| | | Mercury | Liver (13) | 1.7* | |
| Tristan albatross<br>*Diomedea dabbenena* | Gough Island | Cadmium | Liver (2) | 105.6* | 1 |
| | | | Kidney (2) | 452.1* | |
| | | Mercury | Liver (2) | 884.4* | |
| Atlantic yellow-nosed albatross<br>*Thalassarche chlororhynchos* | Gough Island | Cadmium | Liver (9) | 29.7* | 1 |
| | | | Kidney (9) | 82.5* | |
| | | Mercury | Liver (9) | 25.4* | |
| Shy albatross<br>*Thalassarche cauta* | New Zealand region | Cadmium | Liver (42) | 7.7 | 3 |
| | | | Kidney (42) | 74.2 | |
| | | Mercury | Liver (42) | 35.0 | |
| | | | Kidney (42) | 5.1 | |
| Buller’s albatross<br>*Thallassarche bulleri* | New Zealand region | Cadmium | Liver (26) | 13.4 | 3 |
| | | | Kidney (26) | 120.5 | |
| | | Mercury | Liver (26) | 22.2 | |
| | | | Kidney (26) | 4.8 | |
| Sooty albatross<br>*Phoebetria fusca* | Gough Island | Cadmium | Liver (8) | 85.8* | 1 |
| | | | Kidney (8) | 250.8* | |
| | | Mercury | Liver (8) | 465.3* | |
| Atlantic petrel<br>*Pterodroma incerta* | Gough Island | Cadmium | Liver (13) | 62.7* | 1 |
| | | | Kidney (13) | 201.3* | |
| | | Mercury | Liver (13) | 92.4* | |
| Broad-billed prion<br>*Pachyptila vittata* | Gough Island | Cadmium | Liver (31) | 52.8* | 1 |
| | | | Kidney (31) | 108.9* | |
| | | Mercury | Liver (31) | 1.3* | |
| White-chinned petrel<br>*Procellaria aequinoctialis* | New Zealand region | Cadmium | Liver (27) | 20.7 | 3 |
| | | | Kidney (27) | 87.5 | |
| | | Mercury | Liver (27) | 79.4 | |
| | | | Kidney (27) | 12.6 | |
| Grey petrel<br>*Procellaria cinerea* | New Zealand region | Cadmium | Liver (6) | 44.9 | 3 |
| | | | Kidney (6) | 190.0 | |
| | | Mercury | Liver (6) | 142.8 | |
| | | | Kidney (6) | 18.9 | |
| Bulwer’s petrel<br>*Bulweria bulwerii* | Eastern Azores | Mercury | Body feathers (66) | 22.6 | 4 |

*Continued on p. 95*

*Table 1.* Continued

| Species | Location | Pollutant | Tissue (number analysed) | Mean concentration ($\mu$g g$^{-1}$ dry weight) | Reference |
|---|---|---|---|---|---|
| Cory's shearwater *Calonectris diomedea* | Eastern Azores | Mercury | Body feathers (263) | 5.3 | 4 |
| Great shearwater *Puffinus gravis* | Gough Island | Cadmium | Liver (12) | 49.5* | 1 |
| | | | Kidney (12) | 244.2* | |
| | | Mercury | Liver (12) | 6.6* | |
| Little shearwater *Puffinus assimilis* | Gough Island | Cadmium | Liver (13) | 46.2* | 1 |
| | | | Kidney (13) | 141.9* | |
| | | Mercury | Liver (13) | 4.0* | |
| Sooty shearwater *Puffinus griseus* | New Zealand region | Cadmium | Liver (7) | 28.2 | 3 |
| | | | Kidney (7) | 151.1 | |
| | | Mercury | Liver (7) | 2.5 | |
| | | | Kidney (7) | 1.7 | |
| Grey-backed storm petrel *Garrodia nereis* | Gough Island | Cadmium | Liver (8) | 39.6* | 1 |
| | | | Kidney (8) | 75.9* | |
| Fork-tailed storm petrel *Oceanodroma furcata* | Hippa Island | Cadmium | Liver (7–9) | 141.6 | 5 |
| | | | Kidney (7–9) | 242.0 | |
| | | Mercury | Liver (7–9) | 5.7 | |
| | | Lead | Liver (8–9) | 0.3 | |
| | | | Bone (8–9) | 6.2 | |
| Leach's storm petrel *Oceanodroma leucorhoa* | Hippa Island | Cadmium | Liver (7–9) | 99.9 | 5 |
| | | | Kidney (7–9) | 306.2 | |
| | | Mercury | Liver (7–9) | 3.3 | |
| | | Lead | Liver (8–9) | 0.4 | |
| | | | Bone (8–9) | 6.1 | |
| Common diving petrel *Pelecanoides urinatrix* | Gough Island | Cadmium | Liver (17) | 23.1* | 1 |
| | | | Kidney (17) | 105.6* | |
| | | Mercury | Liver (17) | 1.8* | |
| South polar skua *Catharacta maccormicki* | Antarctica | Cadmium | Liver (8) | 16.5* | 2 |
| | | | Kidney (8) | 82.5* | |
| | | Mercury | Liver (8) | 8.9* | |
| Tristan skua *Catharacta hamiltoni* | Gough Island | Cadmium | Liver (13) | 9.9* | 1 |
| | | | Kidney (13) | 85.8* | |
| | | Mercury | Liver (13) | 24.4* | |
| Great skua *Catharacta skua* | Foula | Mercury | Liver (30) | 11.6 | 6 |
| | | | Kidney (33) | 9.7 | |
| | | | Body feathers (197) | 7.0 | |
| Arctic skua *Stercorarius parasiticus* | Foula | Mercury | Body feathers (28) | 2.5 | 7 |
| Black-legged kittiwake *Rissa tridactyla* | Iceland | Mercury | Body feathers (36) | 5.5 | 8 |
| | Firth of Forth | | Body feathers (46) | 3.8 | |
| | Foula | | Body feathers (42) | 2.9 | |

*Continued on p. 96*

*Table 1.* Continued

| Species | Location | Pollutant | Tissue (number analysed) | Mean concentration ($\mu$g g$^{-1}$ dry weight) | Reference |
|---|---|---|---|---|---|
| | N-W Norway | | Body feathers (34) | 4.2 | |
| | N-E Norway | | Body feathers (60) | 3.1 | |
| Glaucous gull | Spitsbergen | Cadmium | Liver (11) | 13.2* | 2 |
| *Larus hyperboreus* | | | Kidney (11) | 75.9* | |
| | | Mercury | Liver (11) | 5.3* | |
| Laughing gull | Galveston Bay | Cadmium | Liver (5) | 1.2* | 9 |
| *Larus atricilla* | | | Kidney (5) | 7.8* | |
| | | Lead | Liver (5) | 3.5* | |
| | | | Kidney (5) | 4.6* | |
| | | | Bone (5) | 28.1* | |
| Yellow-legged gull | Central Azores | Mercury | Body feathers (26) | 5.5 | 4 |
| *Larus cachinnans* | | | | | |
| Arctic tern | Foula | Mercury | Body feathers (23) | 0.9 | 7 |
| *Sterna paradisaea* | | | | | |
| Ancient murrelet | Hippa Island | Cadmium | Liver (7–9) | 6.3 | 5 |
| *Synthliboramphus antiquus* | | | Kidney (7–9) | 36.9 | |
| | | Mercury | Liver (7–9) | 3.0 | |
| | | Lead | Liver (8–9) | 0.3 | |
| | | | Bone (8–9) | 6.2 | |
| Cassin's auklet | Hippa Island | Cadmium | Liver (7–9) | 5.5 | 5 |
| *Ptychoramphus aleuticus* | | | Kidney (7–9) | 72.7 | |
| | | Mercury | Liver (7–9) | 1.0 | |
| | | Lead | Liver (8–9) | 0.3 | |
| | | | Bone (8–9) | 5.5 | |
| Little auk | Spitsbergen | Cadmium | Liver (9) | 13.2* | 2 |
| *Alle alle* | | | Kidney (9) | 69.3* | |
| | | Mercury | Liver (9) | 1.7* | |
| Brünnich's guillemot | Spitsbergen | Cadmium | Liver (9) | 13.2* | 2 |
| *Uria lomvia* | | | Kidney (9) | 52.8* | |
| | | Mercury | Liver (9) | 2.0* | |
| Common guillemot | Foula | Mercury | Body feathers (34) | 1.07 | 7 |
| *Uria aalge* | | | | | |
| Atlantic puffin | Iceland | Mercury | Body feathers (37) | 4.8 | 8 |
| *Fratercula arctica* | St Kilda | | Body feathers (24) | 5.1 | |
| | Firth of Forth | | Body feathers (30) | 3.2 | |
| | Foula | | Body feathers (46) | 3.7 | |
| | N-W Norway | | Body feathers (16) | 3.0 | |
| | N-E Norway | | Body feathers (31) | 1.0 | |

*References* – 1: Muirhead and Furness (1988); 2: Norheim (1987); 3: Stewart et al. (1999); 4: Monteiro et al. (1999); 5: Elliott and Scheuhammer (1997); 6: Thompson et al. (1991); 7: Stewart et al. (1997); 8: Thompson et al. (1992a); 9: Reid and Hacker (1982).

* Data originally expressed on a wet weight basis. Converted to dry weight assuming tissue water content of 70%.

through oil pollution is evident in the review of oiling incidents in the north Atlantic between 1966 and 1983 compiled by Evans and Nettleship (1985). Similarly, of approximately 30,000 dead birds of 90 species recovered following the ‘Exxon Valdez’ oil spill in March 1989, 74% of those identified were guillemots (*Uria* spp.) The total number of birds killed was estimated at 100,000–300,000 (Piatt et al., 1990).

The number and species of seabirds killed by oil spills depends upon a wide range of variables. For example: species vulnerability to oiling, location of spill, time of year when spill occurred, weather conditions at time of spill and type of oil released. Species vulnerability to oil pollution is in itself dependent upon a number of factors. Williams et al. (1995) determined Oil Vulnerability Indices (OVIs) for seabirds in the North Sea, which could be used as a management tool to assess potential risk of marine birds to oil pollution. Developing on earlier vulnerability indices (for example, King & Sanger, 1979), the OVIs were based on the risk of an individual of a given species being killed by oil, population size, potential for a species to recover following a reduction numbers and the reliance of a given species on the marine environment. Hence, those species which suffer high oiling rates, mainly through spending a relatively large proportion of time on the sea’s surface, have small populations, have low potential to recover from a reduction in population and which rely exclusively on the marine environment were assigned relatively high OVIs. Species in this category included black guillemot (*Cepphus grylle*), great skua, shag (*Phalacrocorax aristotelis*), Arctic skua (*Stercorarius parasiticus*), little gull (*Larus minutus*) and razorbill (*Alca torda*) (Williams et al., 1995).

Seabird mortality, resulting from acute oil pollution incidents, has often been dramatic and on a relatively large scale, as in the case of the ‘Exxon Valdez’ spill in 1989 (for example, Piatt et al., 1990). Even relatively small oil spills have resulted in considerable seabird deaths (Barrett, 1979). However, chronic oil pollution, from sources including operational discharges of oil by ships at sea, relatively small spills, oil leakage from offshore drilling operations, inputs from rivers and natural seepage, accounts for at least as many seabird deaths (Clark, 1984).

However, perhaps most importantly from a biological perspective, there is little, if any, evidence to indicate that oil pollution has been responsible for sustained and lasting population declines of seabirds (Clark, 1984; Dunnet, 1987), and mortality from this pollution source probably accounts for only a small proportion of total annual mortality of seabirds (Dunnet, 1982). Even in the vicinity of extreme oil pollution incidents, seabird populations appear to recover to pre-spill numbers in a matter of a few years (Kuletz, 1996; Piatt & Anderson, 1996; Wiens et al., 1996). Furthermore, reductions in seabird population sizes and breeding performance associated with the ‘Exxon Valdez’ oil spill, whilst certainly in part due to oil, could also be the result of long-term changes in food availability (Piatt & Anderson, 1996).

### *Organic pesticides, PCBs and dioxins*

A summary of organic contaminant concentrations in seabirds is presented in Table 2. Perhaps the most infamous of organic pesticides is dichlorodiphenyltrichloroethane (DDT), synthesised in 1874, and first used extensively for pest control during World War II. The seminal paper by Ratcliffe (1967) documented the link between DDT and eggshell thinning, although it is the principal metabolite DDE which is associated most closely with eggshell thinning and subsequent reproductive failure.

Eggshell thinning in seabirds has been reported for several species (see Blus, 1996 for a recent review). The extent of thinning varied in response to DDE concentration and also on an inter-specific basis. For example, in brown pelicans (*Pelecanus occidentalis*), mean DDE residues in eggs of between 1–3 $\mu$g g$^{-1}$ (all organic pollutant data are presented on a wet weight basis) resulted in 5–16% thinning (Blus et al., 1974; King et al., 1977; Blus et al., 1979), whilst residues of 25–66 $\mu$g g$^{-1}$ produced 44–47% thinning (Risebrough, 1972; Jehl, 1973). In contrast, double-crested cormorant (*Phalacrocorax auritus*) eggs from California with a mean DDE level of 32 $\mu$g g$^{-1}$ exhibited 11% thinning, and those from Baja California with a mean DDE level of 24 $\mu$g g$^{-1}$, 30% thinning (Gress et al., 1973). Blus (1996) concluded that eggshell thinning of around 18% was likely to result in population decline. Brown pelicans appear particularly sensitive to DDE; Blus (1982) noted reproductive failure when egg residues exceed 3.7 $\mu$g g$^{-1}$. Recently, Elliott et al. (1997) concluded that there had been little change in DDE residue levels in eggs of pelagic seabirds from the Queen Charlotte Islands over a 20-year period, and that relatively high DDT levels in fork-tailed storm-petrels (*Oceanodroma furcata*) indicated recent exposure. Ludwig et al. (1997) reported that DDT-group residues were

*Table 2.* Selected organic contaminant concentrations in adult seabirds

| Species | Location | Pollutant | Tissue (number analysed) | Mean concentration ($\mu$g g$^{-1}$ wet weight) | Reference |
|---|---|---|---|---|---|
| Adelie penguin | Antarctica | ΣDDT | Egg (27) | 0.007a | 1 |
| *Pygoscelis adeliea* | | | Liver (8) | 0.002 | |
| | | ΣPCB | Egg (27) | 0.009a | |
| | | | Liver (8) | 0.040 | |
| Jackass penguin | Eastern Cape, South Africa | DDE | Egg (21) | 0.060 | 2 |
| *Spheniscus demersus* | | Dieldrin | Egg (21) | ND | |
| Leach's storm petrel | Hippa Island, Canada | DDE | Egg (5) | 0.894a | 3 |
| *Oceanodroma leucorhoa* | | DDT | Egg (5) | 0.008a | |
| | | Dieldrin | Egg (5) | 0.010a | |
| | | ΣPCB | Egg (5) | 0.756a | |
| Fork-tailed storm petrel | Hippa Island, Canada | DDE | Egg (5) | 1.520a | 3 |
| *Oceanodroma furcata* | | DDT | Egg (5) | 0.291a | |
| | | Dieldrin | Egg (5) | 0.024a | |
| | | ΣPCB | Egg (5) | 1.470a | |
| South polar skua | Antarctica | ΣDDT | Egg (46) | 0.100a | 1 |
| *Catharacta maccormicki* | | | Liver (4) | 0.023 | |
| | | ΣPCB | Egg (46) | 0.214a | |
| | | | Liver (4) | 0.190 | |
| Herring gull | Hornøy, Norway | ΣDDT | Egg (5) | 0.260 | 4 |
| *Larus argentatus* | | ΣPCB | Egg (5) | 0.830 | |
| Kelp gull | Eastern Cape, South Africa | DDE | Egg (15) | 0.300 | 2 |
| *Larus dominicanus* | | Dieldrin | Egg (4b) | 0.022 | |
| Black-legged kittiwake | Hornøy, Norway | ΣPCB | Egg (22) | 0.590 | 4 |
| *Rissa tridactyla* | | | | | |
| Roseate tern | Eastern Cape, South Africa | DDE | Egg (3) | 0.040 | 2 |
| *Sterna dougallii* | | Dieldrin | Egg (3) | ND | |
| Shag | Hornøy, Norway | ΣDDT | Egg (5) | 0.180 | 4 |
| *Phalacrocorax aristotelis* | | ΣPCB | Egg (5) | 0.710 | |
| Cape cormorant | Eastern Cape, South Africa | DDE | Egg (7) | 0.050 | 2 |
| *Phalacrocorax capensis* | | Dieldrin | Egg (7) ND | | |
| Great cormorant | Eastern Cape, South Africa | DDE | Egg (9) | 0.170 | 2 |
| *Phalacrocorax carbo* | | Dieldrin | Egg (1b) | 0.060 | |
| Brandt's cormorant | Gulf of Farallones, USA | ΣDDT | Egg (7b) | 0.795a | 5 |
| *Phalacrocorax penicillatus* | | ΣPCB | Egg (7b) | 0.750a | |
| Cape gannet | Eastern Cape, South Africa | DDE | Egg (4) | 0.040 | 2 |
| *Morus capensis* | | Dieldrin | Egg (3b) | 0.002 | |
| Common guillemot | Gulf of Farallones, USA | Σ DDT | Egg (13b) | 2.296a | 5 |
| *Uria aalge* | | ΣPCB | Egg (13b) | 1.652a | |
| | Hornøy, Norway | ΣDDT | Egg (5) | 0.290 | 4 |
| | | ΣPCB | Egg (5) | 0.480 | |

*Continued on p. 99*

*Table 2.* Continued

| Species | Location | Pollutant | Tissue (number analysed) | Mean concentration ($\mu$g g$^{-1}$ wet weight) | Reference |
|---|---|---|---|---|---|
| Brünnich's guillemot | Hornøy, Norway | ΣDDT | Egg (5) | 0.340 | 4 |
| *Uria lomvia* | | ΣPCB | Egg (5) | 0.530 | |
| Razorbill | Hornøy, Norway | ΣDDT | Egg (5) | 0.730 | 4 |
| *Alca torda* | | ΣPCB | Egg (5) | 2.150 | |
| Atlantic puffin | Hornøy, Norway | ΣDDT | Egg (5) | 0.380 | 4 |
| *Fratercula arctica* | | ΣPCB | Egg (5) | 1.060 | |
| Pigeon guillemot | Gulf of Farallones, USA | ΣDDT | Egg (11b) | 0.225a | 5 |
| *Cepphus columba* | | ΣPCB | Egg (11b) | 0.145a | |
| Ancient murrelet | Hippa Island, Canada | DDE | Egg (6) | 1.300a | 3 |
| *Synthliboramphus antiquus* | | DDT | Egg (6) | 0.002a | |
| | | Dieldrin | Egg (6) | 0.016a | |
| | | ΣPCB | Egg (6) | 0.398a | |
| Cassin's auklet | Hippa Island, Canada | DDE | Egg (5) | 0.539a | 3 |
| *Ptychoramphus aleuticus* | | DDT | Egg (5) | 0.002a | |
| | | Dieldrin | Egg (5) | 0.020a | |
| | | ΣPCB | Egg (5) | 0.186a | |
| Rhinoceros auklet | Gulf of Farallones, USA | ΣDDT | Egg (5b) | 0.570a | 5 |
| *Cerorhinca monocerata* | | ΣPCB | Egg (5b) | 0.360a | |
| | Helgesen Island, Canada | DDE | Egg (6) | 0.195a | |
| | | DDT | Egg (6) | 0.001a | |
| | | Dieldrin | Egg (6) | 0.009a | |
| | | ΣPCB | Egg (6) | 0.172a | |

*References* – 1: Court et al. (1997); 2: de Kock and Randall (1984); 3: Elliott et al. (1997); 4: Barrett et al. (1996); 5: Jarman et al. (1996).
a: Geometric mean.
b: Number of samples in which pollutant detected.
ND: Not detected.

sufficient to cause eggshell thinning and contributed to reduced productivity by 2–3% in black-footed albatrosses (*Phoebastria nigripes*) at Midway Atoll. Although the use of DDT in western countries has been banned for many years and there is evidence that concentrations in seabirds have correspondingly declined (for example, Nisbet & Reynolds, 1984), it has been used extensively elsewhere. Detrimental impacts on seabirds from other sources may be more significant (see Industrial fisheries section, this paper), but continued exposure of pelagic seabirds to DDT-group compounds is cause for concern and should be monitored closely.

There is relatively little direct evidence of detrimental effects of the 'drin' group of pesticides (aldrin, dieldrin, endrin, isodrin and telodrin) on seabirds. In the UK for example, aldrin and dieldrin were applied as seed dressings, and dieldrin was additionally used in sheep dips, limiting the exposure of seabirds to these contaminants. Unlike DDT and its metabolites which affect birds through eggshell thinning, the drins cause direct mortality when present in sufficient quantity. In the 1960s, Sandwich terns (*Sterna sandvicensis*) at a large colony in the Dutch Wadden Sea were recorded dying in convulsions and tremors (Koeman et al., 1968). Analyses of tern tissues revealed lethal concentrations of dieldrin, endrin and telodrin, sourced to an insecticide-producing plant near the mouth of the river Rhine. The Sandwich tern colony declined from 20,000 pairs to less than 1,000 pairs over a few years (Koeman et al., 1968).

There is a relatively large body of work on the detrimental effects of polychlorinated biphenyls (PCBs) and polychlorinated dibenzodioxins (PCDDs) in gulls, terns (*Sterna* spp.) and cormorants (*Phalacrocorax* spp.) from the Great Lakes. Whilst outside the scope of this marine review, the reader is directed to the comprehensive summary of PCBs and dioxins in this region provided by Hoffman et al. (1996). There is some evidence that PCB levels in seabirds have declined over recent decades (for example, Nisbet & Reynolds, 1984; Savinova et al., 1995), although levels have remained stable in Leach's storm-petrel eggs from the Queen Charlotte Islands over a twenty year period to 1991 (Elliott et al., 1997). PCB and dioxin residues were considered sufficiently high in black-footed albatrosses at Midway Atoll to contribute to reduced productivity in this species (Ludwig et al., 1997). Further monitoring of PCBs and PCDDs in pelagic seabirds would seem prudent.

### *Plastics*

Large-scale production of plastic began during the 1940s and has increased considerably since then. Plastics can be divided into two broad categories; user-plastics, such as bottles, packaging materials and bags, and industrial plastic particles from which user-plastic products are manufactured. User-plastics can effect seabirds in a number of ways. Discarded or lost fishing line and fishing nets or net fragments can cause entanglement of seabirds (Laist, 1987), small fragments or whole pieces of user-plastic may be ingested causing cuts and ulceration to the intestine (Fry et al., 1987) and discarded plastic products may be incorporated by seabirds into their nests (Montevecchi, 1991) increasing the likelihood of entanglement by breeding adults or by developing chicks. However, although often unsightly, discarded user-plastics pose a relatively minor threat to most seabirds.

Most plastic found in seabird stomachs is industrial plastic particles, and in general, procellariiforms appear to be particularly prone to ingest this form of plastic (Azzarello & van Vleet, 1987; Fry et al., 1987; Ryan, 1987a; van Franeker & Bell, 1988; Moser & Lee, 1992), although Spear et al. (1995), in a study of plastics in seabirds from the tropical Pacific over an eight year period, recorded relatively high incidence of fragments of user-plastics and found that non-procellariiforms species were as likely to contain plastic as procellariiform species. Whilst experimental work with chickens (*Gallus* spp.) indicated that the presence of plastic particles in the gizzard reduced food consumption and energy assimilation (Ryan, 1988), and plastic may increase uptake of PCBs in great shearwaters (*Puffinus gravis*) (Ryan et al., 1988), evidence for a detrimental effect of plastic particles on digestive ability or body condition of seabirds is at best weak (Furness, 1985a, b; Ryan, 1987b; Ryan & Jackson, 1987; Moser & Lee, 1992). However, Pettit et al. (1981) recorded degradation of the digestive tract as a result of ingested plastic in Laysan albatrosses, Auman et al. (1997) concluded that ingested plastic increased physiological stress in Laysan albatross chicks due to satiation and mechanical blockage and Spear et al. (1995) reported a significant negative relationship between number of plastic particles and body weight in five species of procellariiforms.

## Industrial fisheries

Industrial fisheries affect seabirds both directly and indirectly and can have both beneficial and detrimental effects. For scavenging seabirds such as fulmars (*Fulmarus* spp.), gannets (*Morus* spp.), great skuas and some gulls, fisheries provide offal and discards of large demersal fish that would not otherwise be available. This may be a particularly important source of food for some species during the winter months (Camphuysen et al., 1995). According to one recent estimate (Alverson et al., 1994), this material comprises about 25 million tonnes annually, or about one third of current global fish production. This has allowed an increase in the populations and geographic distributions of some species of seabird, notably northern fulmars (Thompson et al., 1995) and Audouin's gull (*Larus audouinii*) (Oro & Ruiz, 1997). Furness (1992) suggested that populations of scavenging seabirds in the North Sea may now be limited by the quantity of material discarded annually, although this is unlikely to be the case at least for northern fulmars, which include a high proportion of small epipelagic fish and zooplankton in the diet in addition to discards (Phillips et al., 1999).

In several ecosystems, the stocks of fish that seabirds feed upon have increased in response to large-scale fishing of their predators and competitors. In the North Sea, reductions in stocks of demersal predatory fish such as Atlantic cod (*Gadus morhua*) in the early years of this century, followed by massive reductions in stocks of herring (*Clupea harengus*) and mackerel (*Scomber scombrus*) in the 1950s, 1960s and 1970s led to increases in stocks of sandeels (Ammodytidae), which most seabirds in the region feed upon (Sherman et al., 1981; Furness, 1982). In the Southern Ocean, reductions in populations of baleen whales have made large quantities of krill (*Euphausia superba*) available for a variety of seabird species (Croxall, 1992). However in most parts of the world, exploitation of pelagic shoaling species is intense and industrial fisheries are in direct competition with seabirds (Blaxter & Hunter, 1982; Furness & Ainley, 1984).

Industrially-fished stocks tend to have an age-structure heavily skewed towards younger age-classes of a size taken by seabirds. This can increase food availability at least in the short-term, although such an age-structure can also lead to low and variable recruitment, with eventual detrimental effects on spawning stock biomass. Fluctuations in recruitment are correlated with reproductive lifespan, so that long-lived species such as herring and mackerel or more vulnerable to these effects than are short-lived species such as sandeels and anchovies (*Engraulis* spp.) that recruit into the breeding stock at an early age (Furness & Ainley, 1984).

The two main detrimental effects of fisheries on seabirds are depletion of prey stocks and direct mortality. These are each considered in more detail below.

### *Depletion of prey stocks*

Major changes in marine ecosystems have coincided with intensive industrial fisheries in a number of regions including the North Sea and north Atlantic (Sherman et al., 1981; Mehlum & Bakken, 1994) the coasts of California and northern Chile (MacCall, 1982) and the Benguela current system off southern Africa (Furness & Cooper, 1982). In some cases these changes have been linked directly to industrial fishing and have had pronounced effects on seabirds. For instance repeated breeding failure of Atlantic puffins on Røst, Norway succeeded the depletion of the west-Norway herring stock and subsequent increase in sandeel fishing (Anker-Nilssen & Barrett, 1991). However in most cases it is difficult to distinguish between effects of fisheries and natural phenomena in causing short-term reductions in fish stock biomass (Duffy & Schneider, 1994; Furness, 1996b; Wright, 1996).

Seabird prey such as small pelagic fish, zooplankton and cephalopods generally fluctuate in abundance from year to year, as a result of short lifespans and highly variable recruitment, and fluctuations caused by fisheries are likely simply to exaggerate these natural variations (Furness, 1996b). Seabirds exhibit a number of physiological, behavioural and life-history mechanisms that reduce the impact of these variations in food supply on long-term population size (Hamer et al., 1993; Montevecchi, 1993). Thus only major fishery-induced reductions in food supply are likely to affect seabird population sizes, although less extreme reductions may have short-term effects on diets, time/activity budgets and other aspects of breeding ecology (Hamer et al., 1991; Uttley et al., 1994). These effects are felt most strongly in species with specialized surface-feeding habits, limited foraging ranges, limited ability to increase time spent foraging and energetically expensive foraging techniques (Furness & Ainley, 1984). These include penguins, cormorants, gulls, skuas and especially terns.

### *Direct mortality*

Many different species of seabird may be accidentally caught and drowned in fishing gear, particularly monofilament nylon drift nets, gill nets and baited hooks. Bag nets and pound nets set close to shore to catch salmon (*Salmo salar*) can also catch large numbers of pursuit-diving seabirds, particularly auks (Strann et al., 1991; Murray et al., 1994). Seabird life-histories are characterized by delayed reproduction, low annual productivity and long lifespans. Thus, increased adult mortality has a much stronger potential effect on population size than does decreased reproductive success or lowered juvenile survival, and so direct mortality in fishing gear is a greater cause for immediate concern than depletion of prey stocks.

Many authors have documented auks, particularly guillemots, drowning in large numbers in gill nets in the north Atlantic and north Pacific Oceans (King, 1984; Piatt et al., 1984; Bertram, 1995). For instance, Strann et al. (1991) estimated conservatively that 200,000 common guillemots (*Uria aalge*) drowned in cod fishing nets in north Norway in April 1985. These were mainly immature birds from a wide geographical range within the species' distribution, and so the impact of this mortality on population sizes was not likely to have been great. Murray et al. (1994) similarly concluded that drowning of common guillemots in salmon nets in north-east Scotland had only a minor impact on breeding populations. However, inshore fisheries using drift-nets and gill nets have in some cases been situated in close proximity to large breeding colonies of seabirds during the summer months, with the result that many adult seabirds have been caught and drowned. Prior to their prohibition in 1989 as a fisheries protection measure, drift nets set for salmon in north Norway caught up to 50,000 adult common guillemots during June and July each year. Breeding populations of guillemots in the area decreased by as much as 95% between 1965 and 1989, probably as a direct result of adult mortality in fishing nets (Strann et al., 1991).

The fishing method currently under most scrutiny for its impact on seabird populations is demersal and pelagic longlining using baited hooks (Bergin, 1997). This technique is favoured by fisheries managers because it catches mainly target species, causes no damage to the seabed and allows capture of high-quality fish at low fuel consumption. Longlining is one of the major fishing techniques in both hemispheres (Løkkeborg, 1998) but is causing particular concern in the southern hemisphere, because of its impact on albatrosses and petrels (Robertson & Gales, 1997; Weimerskirch et al., 1999). These are killed in large numbers by drowning after striking at baited hooks set to catch a range of species, particularly tuna (*Thunnus* spp.) and Patagonian toothfish (*Dissostichus eleginoides*). The rate of seabird bycatch in longlining is generally very low but because of the large number of hooks set, total mortality can be sufficient to reduce breeding population sizes of some species. For instance the Japanese pelagic longline fishery for southern bluefin tuna (*T. maccoyi*) has deployed up to 100 million hooks or more annually during the last decade. In Australian waters, the average rate of seabird bycatch from this fishery was 0.15 birds per 1,000 hooks set, but this translated into an annual mortality of up to 3,500 birds per year (Gales et al., 1998). Longline fishing operations have been implicated in serious population declines of a number of albatrosses including wandering albatross (*Diomedea exulans*), grey-headed albatross (*Thalassarche chrysostoma*), black-browed albatross (*T. melanophrys*), Atlantic yellow-nosed albatross (*T. chlororhynchos*) and light-mantled sooty albatross (*Phoebetria palpebrata*) (Croxall et al., 1990; Weimerskirch et al., 1997; Gales et al., 1998). Some populations have declined by as much as 90% and are facing extinction if current catch rates are not reduced. Hence much current research effort is focusing on methods to reduce bycatch, including the use of tori poles, streamer lines and other devices to scare birds away from hooks, deploying hooks beneath the water surface via a setting funnel, setting hooks at night and luring birds away from hooks with offal or other sources of alternative food (Cherel et al., 1996; Løkkeborg, 1998).

## Climate change

Evidence of a discernible human influence on global climates is accumulating (Houghton et al., 1996) and current global circulation models predict that this influence will increase further over the next 100 years (Hulme & Jenkins, 1998), particularly at high latitudes. For instance mean surface air temperature is predicted to increase by 2–6 °C during summer and by 8–12 °C during winter in the Barents Sea region of northern Europe (Mitchell et al., 1990). These changes could have profound impacts on seabirds, particularly those breeding at high latitudes. The marginal ice-zones and lead systems in the northern Barents Sea and eastern Svalbard are an important foraging area for several species of seabirds, especially Brünnich's guillemot (*Uria lomvia*) (Mehlum & Bakken, 1994) and a decrease in the area of sea-ice resulting from warmer sea surface temperatures could greatly increase the foraging distances of these birds from their breeding colonies, with potentially profound impacts on breeding productivity and population size. In the southern hemisphere, a 94% decline in the population of rockhopper penguins at Campbell Island, New Zealand over the past 50 years has been attributed to warmer sea surface temperatures causing their euphausiid prey to move further offshore, beyond the foraging ranges of adults (Cunnigham & Moors, 1994).

Seabirds breeding at lower latitudes are predicted to experience less warming than those at high latitudes, but could nonetheless be adversely affected by climate change, for instance through an increase in the frequency and severity of El Niño Southern Oscillations (Timmermann et al., 1999), which are associated with high adult mortality in addition to breeding failure (Schreiber & Schreiber, 1984). Some species could however benefit from climate change. For example, increases in populations of northern gannets (*Morus bassana*) in Newfoundland over the past 50 years are correlated with warming surface water conditions and a corresponding increase in the availability of mackerel close to breeding colonies (Montevecchi & Myers, 1997).

Increased summer air temperatures at breeding colonies themselves could cause problems for highlatitude species such as skuas, which have high mass-specific basal metabolic rates, high body temperatures, heavy insulative plumage and high levels of activity during the breeding season (Ellis, 1984). The breeding distributions of skuas may currently be confined to regions with midsummer temperatures below about 14 °C due to problems of heat dissipation by adults during chick-rearing (Furness, 1988), in which case warmer summer temperatures would be likely to result in a contraction of their breeding distributions to higher latitudes.

Impacts of climate change on seabirds may interact synergistically with other factors. For example, over-exploitation of the Pacific anchoveta (*Engraulis ringens*) stock resulted in the failure of Peruvian guano seabird populations to recover after El Niño events (Duffy, 1983). In other cases, impacts of climate-induced changes in marine ecosystems may be ameliorated by other factors. For example, many birds breeding at high latitudes need to trade-off foraging against brooding of chicks to maintain nestling body temperature, and increased time foraging in warmer climates could be mitigated by a reduction in the chicks' requirements for brooding. The question of how climate change might be affecting seabird populations requires much more investigation, particularly near the limits of species' geographical distributions and the margins of oceanographic regions, where initial changes are most likely to be detected.

## Seabirds as monitors of marine ecosystem health

### *Long-term trends in marine mercury contamination*

Unlike synthetic chemicals, heavy metals occur naturally as part of biogeochemical cycles within marine systems. Consequently, it is often extremely difficult, if not impossible, to distinguish between 'natural' and 'anthropogenic' metal burdens within many marine biota. In seabirds, relatively high heavy metal burdens may represent accumulation of background concentrations, exposure to pollution sources or a combination of these two processes. However, seabirds have proved an excellent monitor of long-term patterns of mercury burdens within marine systems. Furthermore, it has been possible to delineate between relatively high mercury concentrations that are likely the result of natural accumulation, and elevated mercury concentrations that are the result of exposure to anthropogenic sources of mercury.

There are several key factors which have enabled the use of seabirds to illuminate historical trends of mercury contamination in marine ecosystems. Seabirds deposit ingested mercury into growing feathers following moult. Mercury excreted into feathers is bound strongly to disulphide linkages within the keratin molecule (Crewther et al., 1965), and is extremely resistant to further physical and chemical alteration (Applequist et al., 1984). Mercury concentrations in feathers have been shown to reflect those in the diet in a dose-dependent manner (Lewis & Furness, 1991), and to correlate positively with those in internal tissues (Thompson et al., 1991). Hence, a sample of feathers represents a robust, non-destructive monitor tissue with which to track mercury burdens in seabirds. Perhaps most importantly, many seabirds taken for collections, some dating back to the middle of the nineteenth century, have been preserved as study skins in museums, providing a source of well-documented (location and year of capture) historical feather material. By measuring mercury concentrations in these historical samples and in samples collected recently, long-term trends in mercury burdens can be elucidated.

Thompson et al. (1992b) applied this approach to a number of seabirds from the UK. In all but one species, the northern fulmar, significant positive trends in mercury concentrations were recorded over the period pre-1930 to post-1980. Increases in mercury concentrations were most pronounced in Manx shearwaters (+175% and +152%, depending on region. Figure 1A and 1B) and Atlantic puffins (+120%. Figure 2), although modest, but statistically significant, increases were also measured in great skuas and north Atlantic gannets. The authors concluded that these results were indicative of an increase in mercury concentrations within the marine ecosystem in the north and north-east Atlantic, most likely as a consequence of increased anthropogenic emissions and widespread global transport (Thompson et al., 1992b). Interestingly, northern fulmars exhibited a significant decrease in mercury burdens over the same time period, a result attributed to a shift in diet to prey of lower trophic status, later confirmed using stable isotope analysis (Thompson et al., 1995).

A comparable study was undertaken with common terns (*Sterna hirundo*) and herring gulls (*Larus argentatus*) from the German North Sea coast (Thompson et al., 1993a). Similarly, common terns exhibited an increase in mercury concentrations of +377% in feather samples collected prior to 1940

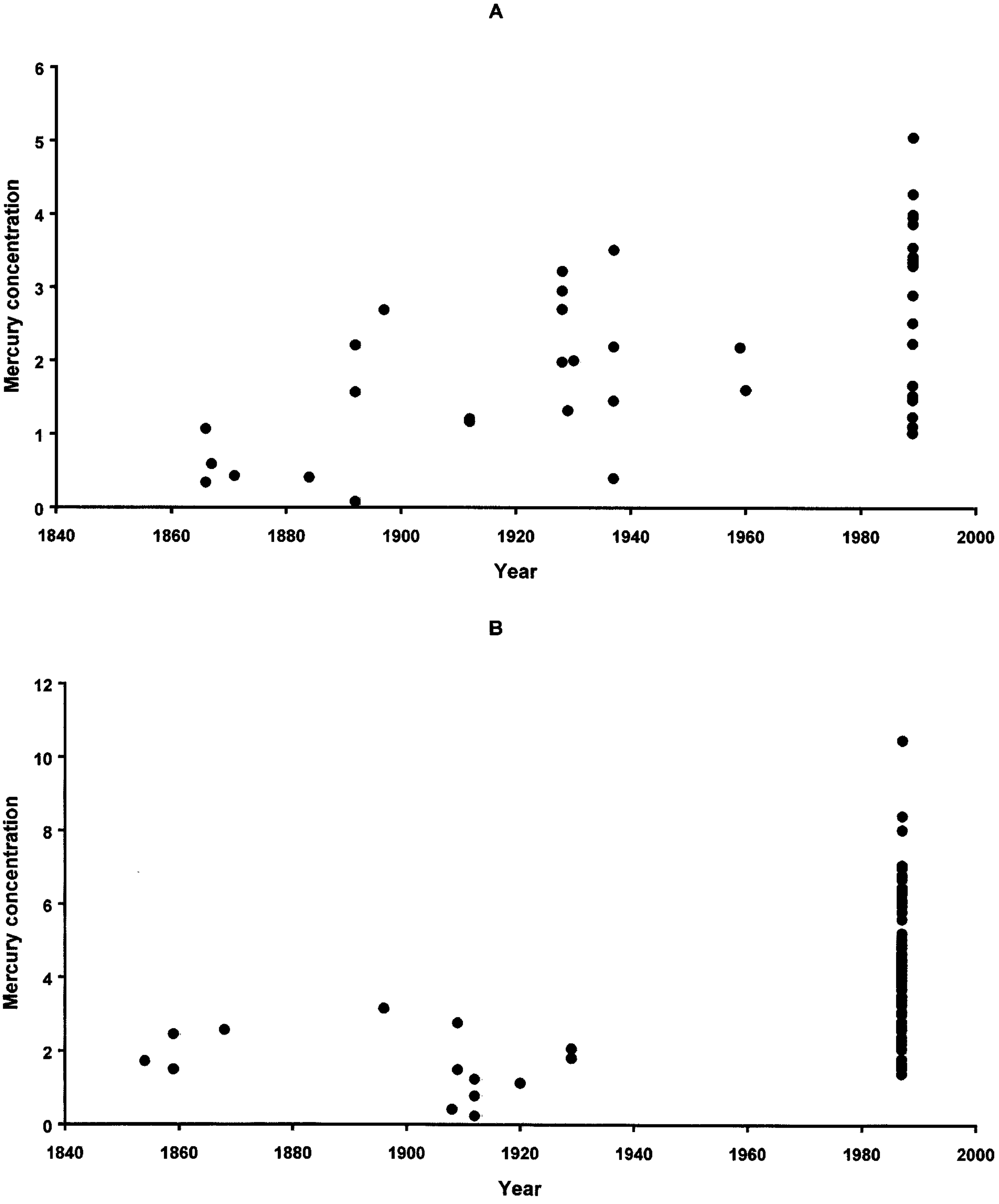

*Figure 1.* Temporal variation in mercury concentrations ($\mu$g g$^{-1}$) of body feathers of Manx shearwaters (*Puffinus puffinus*) sampled from (A) the south-west of Britain and Ireland, and from (B) the north-west of Britain and Ireland. After Thompson et al. (1992b).

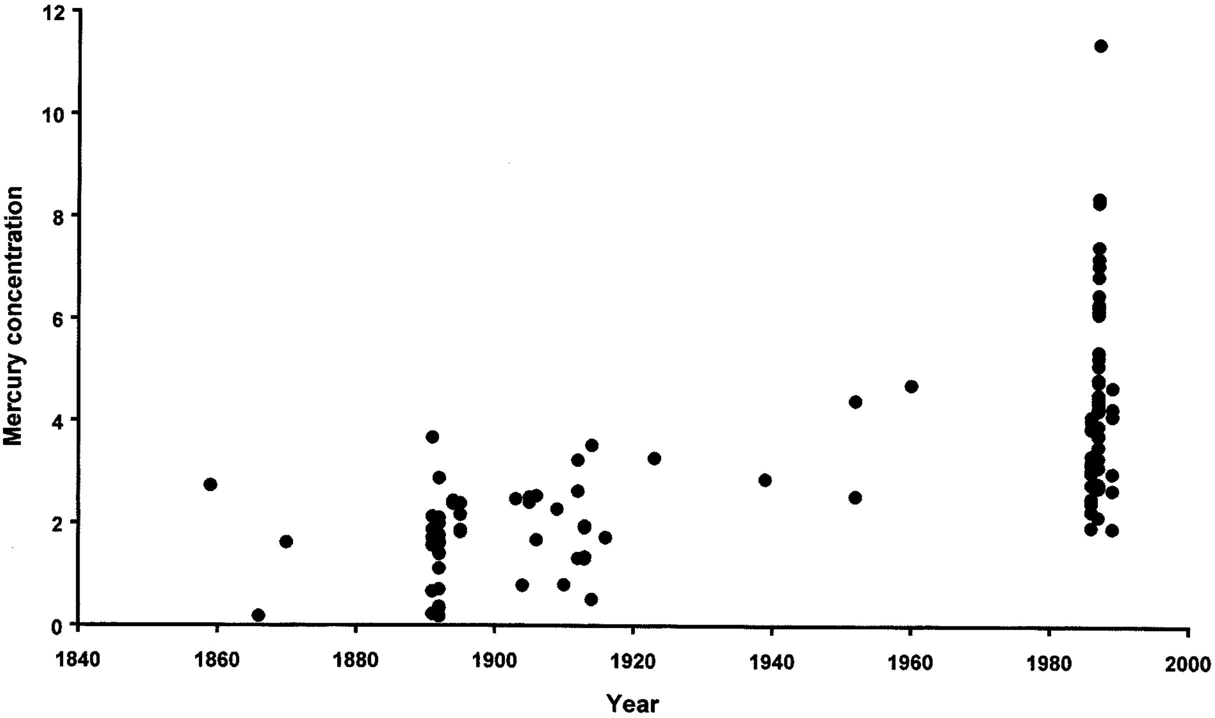

*Figure 2.* Temporal variation in mercury concentrations ($\mu$g g$^{-1}$) of body feathers of Atlantic puffins (*Fratercula arctica*) sampled from the south-west and west of Britain and Ireland. After Thompson et al. (1992b).

(earliest samples from 1866) and after this date (mainly from 1989 and 1990), whilst in herring gulls the increase was +75% over the same period (Thompson et al., 1993a). The relatively large number of herring gull samples allowed an analysis of variation in mercury concentrations by decade, revealing that mercury burdens peaked during the 1940s (mean ca. 12 $\mu$g g$^{-1}$) presumably due to unrestricted industrial output during World War II, fell during the 1950s only to rise to a second, but slightly lower peak during the 1970s (mean ca. 10 $\mu$g g$^{-1}$), at which point restrictions on mercury emissions resulted in a decline to lower concentrations during the 1980s (Thompson et al., 1993a). In contrast to global transport of mercury, the authors argued that riverine transport of mercury to a relatively well-defined coastal marine ecosystem, was the most likely source of mercury over the study period.

Unlike the results of the above studies, seabirds from the southern hemisphere exhibited very weak or no increases in mercury burdens over similar time scales, indicating that virtually all mercury in contemporary southern ocean seabirds results from natural accumulation, and that anthropogenic mercury emissions have been far less in the southern hemisphere compared to mercury emissions in the northern hemisphere (Thompson et al., 1993b).

Recently, this approach has been extended to investigate long-term trends in mercury burdens in seabirds based on their specific dietary specialisations (Monteiro & Furness, 1997; Thompson et al., 1998). Interestingly, Monteiro and Furness (1997) recorded the most pronounced historical mercury increases in seabirds which fed predominantly on meso-pelagic prey (increase in mercury concentration of 3.5–4.8% year$^{-1}$), with relatively modest increases (1.1–1.9% year$^{-1}$) in mercury burdens in seabirds feeding on epi-pelagic prey. Monteiro and Furness (1997) suggested these results were entirely in keeping with methylation and bioassimilation of mercury below the thermocline.

Hence, through the analysis of time-series of feather samples from seabirds, it has proved possible to track long-term trends in mercury contamination of a range of marine ecosystems, from pelagic to coastal. Additionally, seabirds can track increases in mercury loading within specific marine compartments (meso-pelagic versus epi-pelagic). Whilst there is little evidence that contemporary mercury concentrations in seabirds result in deleterious effects (see Heavy metals section, this paper), the increases in mercury concentrations in northern hemisphere seabirds revealed through these analyses warrant further monitoring and could prove harmful in the future. The use of feathers in this way is restricted to mercury contamination; analyses of other metals and organic pollutants in feathers suffer from either extremely low or undetectable concentrations (cadmium and organics) or exogenous surface contamination (lead).

## Summary

Seabirds experience a diverse array of stresses. Clearly, specific stresses do not impact upon different species or groups of seabird equally. For example, albatrosses and other large petrels are especially prone to being impaled on bated longline hooks; auks, which spend relatively large amounts of time on the sea's surface while diving for food, are particularly prone to oiling.

Whilst seabirds can be used to track and monitor concentrations of toxic heavy metals and synthetic organic pollutants within marine systems, and to assess levels of marine plastic pollution, these stresses appear to have relatively minor impacts upon individuals and populations at present. However, this situation is unlikely to remain constant, levels of some of these pollutants could increase in the future and their effects become more serious. Acute oil pollution incidents can impact relatively large numbers of seabirds, and given unfavourable circumstances (location, weather, season, and so on) could pose extremely serious threats to some seabird species, particularly those with relatively restricted distributions or small populations. Most seabird mortality due to oil occurs as a result of chronic oil pollution, but even this probably accounts for only a small part of the total annual mortality in seabird populations.

Croxall (1997) identified incidental catch of albatrosses on longline fishing gear as potentially the most serious threat faced by this group. Virtually all albatross species are affected and most meet the International Union for the Conservation of Nature's criteria for globally threatened status (Croxall, 1997; Croxall & Gales, 1997). In October 1998, however, the Food and Agricultur Organisation (FAO) of the United Nations agreed on an International Plan of Action for reducing incidental catch of albatrosses and other seabirds in longline fisheries. The agreement provides that each nation should identify problems of seabird bycatch in their longline fisheries, and develop a National Plan of Action by 2001. It remains to be seen whether the impact of longlining on seabirds can be reduced or eliminated.

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