OSPREY: WORLDWIDE SENTINEL SPECIES FOR ASSESSING AND MONITORING ENVIRONMENTAL CONTAMINATION IN RIVERS, LAKES, RESERVOIRS, AND ESTUARIES

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In the United States, many fish and wildlife species have been used nationwide to monitor environmental contaminant exposure and effects, including carcasses of the bald eagle (Haliaeetus leucocephalus), the only top avian predator regularly used in the past. Unfortunately, bald eagles are sensitive to investigator intrusion at the nest. Thus, the osprey (Pandion haliaetus) is evaluated as a potential sentinel species for aquatic ecosystems. Several characteristics support the choice of the osprey as a sentinel species, including: (1) fish-eating diet atop the aquatic food web, (2) long-lived with strong nest fidelity, (3) adapts to human landscapes (potentially the most contaminated), (4) tolerates short-term nest disturbance, (5) nests spatially distributed at regular intervals, (6) highly visible nests easily located for study, (7) ability to accumulate most, if not all, lipophilic contaminants, (8) known sensitivity to many contaminants, and (9) nearly a worldwide distribution. These osprey traits have been instrumental in successfully using the species to understand population distribution, abundance, and changes over time; the effects of various contaminants on reproductive success; how contaminants in prey (fish on biomass basis) contribute to egg concentrations (i.e., biomagnification factors); and spatial residue patterns. Data summarized include nesting population surveys, detailed nesting studies, and chemical analyses of osprey egg, organ, blood, and feather samples for contaminants that bioaccumulate and/or biomagnify in aquatic food webs; and biochemical evaluations of blood and various organs. Studies in the United States, Canada, Mexico, Europe, and elsewhere have shown the osprey to be a useful sentinel species for monitoring selected environmental contaminants, including some emerging contaminants in lakes, reservoirs, rivers, and estuaries.

Thousands of chemical compounds have been introduced into the environment since the early 1900s, polluting the air, soil, water, and biota. The physiological effects environmental contaminants have on living organisms depend on their bioavailability, toxicity, concentration, duration of exposure, and species sensitivity. Many of the contaminants found to persist in the environment today are classified as polyhalogenated aromatic hydrocarbons. These toxic compounds (e.g., organochlorine pesticides [OC], polychlorinated biphenyls [PCB], dioxins [PCDD], furans [PCDF], brominated flame retardants [PBDE], perfluorooctane sulfonate compounds [PFOS]) resist environmental or metabolic breakdown, are lipophilic, and may bioaccumulate and/or biomagnify up the food web. Humans and vertebrate species that share the same local environs share common biochemical, molecular, and cellular responses resulting from exposure to toxic agents (Fox 2001). Because of the recognized commonalities, animals have been used for centuries as early warning to exposure of environmental hazards. A classic example was the in situ use of canaries to detect the toxic buildup of carbon monoxide in coal mines, as they are more sensitive to the odorless gas than humans. Certain wildlife species by virtue of their position in the food web and biological traits (e.g., ability to live in human-altered environments) are among the first to be exposed and respond to newly introduced environmental stressors. The concept of “sentinel species,” based on position in the food web and species sensitivity, is important because these species are used to empirically assess bioavailability, tissue concentrations present, and effects related to contaminant exposure (National Research Council, 1991; Golden & Rattner, 2003). A sentinel species can be defined as an organism used to evaluate environmental contamination and its implications on environmental health based on its chemical sensitivity, position in the biotic community, exposure potential, and geographic distribution or abundance (Lower & Kendall, 1990; O’Brien et al., 1993).
For an animal to be considered a key sentinel species (Basu et al., 2007), it must meet certain requirements, which include: (1) widespread distribution, (2) nonmigratory status, (3) position atop of the food web, (4) ability to bioaccumulate contaminants, (5) a restricted home range, (6) a well-known biology and natural history, (7) sensitivity to contaminants, (8) available in sufficient numbers, and (9) maintained and studied in captivity.

Though animal studies under controlled laboratory conditions are important for determining causation, dose-response relationships, and molecular mechanisms of action from single chemical exposures, these studies have limited use for understanding complex biological responses resulting from wildlife exposed to real-world concentrations of contaminant mixtures. Fish and wildlife species studied outside the laboratory provide important information on contaminant exposure routes and effects under natural settings. Historically, the starling (*Sturnus vulgaris*) and mallard (*Anas platyrhynchos*) were used in the United States for a period of years to monitor national contaminant residue trends, but not effects (White, 1976, 1979a, 1979b; White & Heath, 1976). For many years, carcasses of bald eagles found dead across the United States (a biased sample, though useful in many respects) were necropsied to determine cause of death, which included analysis of tissues for contaminants (Cromartie et al., 1975; Kaiser et al., 1980). In addition, many bird species (e.g., bald eagle, American kestrel [*Falco sparverius*], black-crowned night-heron [*Nycticorax nycticorax*], brown pelican [*Pelecanus occidentalis*], Forster’s tern [*Sterna forsteri*], Franklin’s gull [*Larus pipixcan*], great blue heron [*Ardea herodias*], tree swallow [*Tachycineta bicolor*], white-faced ibis [*Pegadis chihi*], double-crested cormorant [*Phalacrocorax auritus*]) were studied by scientists in localized areas to quantify both contaminant exposure and effects, especially reproductive success (Lincer, 1975; Blus, 1982; Henny et al., 1985, 2002; Henny & Herron, 1989; Tillitt et al., 1992; Anthony et al., 1993; Custer et al., 1997; Blus et al., 1998; Bowerman et al., 1998; Custer et al., 1999; Thomas & Anthony 1999; Buck et al., 2005). Like the cosmopolitan black-crowned night-heron, the osprey (*Pandion haliaetus*), a large (approximately 1.6 kg) piscivorous bird of prey, has a nearly worldwide breeding distribution. Ospreys utilize both salt- and freshwater habitats, living, nesting, and hunting along coasts in saltwater marshes, lagoons, estuaries, coral reefs, and occasionally deeper waters off-shore, and inland along rivers, lakes, reservoirs, ponds, and marshes. Because of its position at the top of the aquatic food web as an obligate piscivore, its ability to accumulate lipophilic contaminants, and its sensitivity to many contaminants, the U.S. Geological Survey, U.S. Fish and Wildlife Service, Canadian Wildlife Service, and scientists from other organizations have often selected the osprey for contaminant studies. Several osprey life history traits make it a species of choice for contaminant monitoring and research, justifying its use as a sentinel species. These characteristics include:

1. An aquatic diet consisting almost exclusively (99+%) of fish.
2. Localized feeding habits, within relatively short distance of the nest. Fish species captured can be identified based on prey remains at nest sites, direct observations, and/or photographs.
3. A long-lived species, living up to 25 yr, with strong nest site fidelity, returning year after year to the same or a nearby nest. Ospreys have readily observable nests, constructed of large sticks at exposed locations, that are easily detected during both aerial and ground/boat surveys. Ospreys commonly nest on artificial structures, generally facilitating access for egg and blood/tissue collection.
4. Adapts to human landscapes including industrial and municipal sites where contamination may be most severe, and readily habituates to human activity.
5. Tolerates short-term nest disturbance for egg/blood collection, resulting in little or no effect on nest success.
6. Removal of a “sample egg” from the usual 3-egg clutch for contaminant analyses has limited effect on productivity at the nest, i.e., loss of 0.28 young fledged for each egg collected on the Columbia River in 1997–1998 (Henny et al., 2004).
7. Nests spatially distributed at regular intervals along waterways, as opposed to clumped at a limited number of regional colonies. This distribution permits random egg and tissue collections along river segments or strategic collections related to potential contaminant sources.
The osprey is also sensitive to many bioaccumulative contaminants, including \( p,p' \)-dichlorodiphenyl-nylethylene (DDE) and other OC, PCB, PCDD, PCDF, and mercury (Hg). Information is available regarding reproductive effect concentrations in osprey eggs (for DDE, Johnson et al., 1975; Spitzer et al., 1978; Wiemeyer et al., 1988; Henny et al., 2004; for 2,3,7,8-tetrachlorodibenzo-p-dioxin [TCDD] Woodford et al., 1998).

Studies involving ospreys in the Pacific Northwest and elsewhere in North America and Europe over the last several decades focused on the following issues: (1) investigations of diet (biomass basis), (2) surveys of population numbers and changes over time, (3) evaluations of spatial contaminant residue patterns (via egg, plasma, feather collections) relative to known point sources, or along great distances of major rivers and estuaries, and (4) investigations of effects of various contaminants on reproductive success, including percentage of population adversely affected. After decades of DDE-associated nesting failures (eggshell thinning and reduced egg viability) and population declines, most osprey populations throughout their worldwide breeding range have now recovered. By habituating to human activities and structures, the species has recently established nesting populations along some of the most industrialized and polluted waters in the Pacific Northwest (e.g., Portland Harbor, Seattle Harbor) and elsewhere (Chesapeake Bay, Delaware Bay). This avian piscivore is also being studied as a wildlife receptor of concern for assessment of exposure and effects to PCB and other contaminants at a U.S. Environmental Protection Agency (EPA) Superfund site in Washington State as part of a National Resource Damage Assessment by the Department of the Interior. Therefore, it is postulated that the osprey is a logical candidate for use as an avian sentinel species because of its (1) global distribution, (2) position at the top of the aquatic food web, (3) ability to bioaccumulate lipophilic contaminants, (4) chemical sensitivity to many contaminants, (5) well-known biology and natural history, and (6) tolerance to research activities. Despite the many advantages for using the osprey as a sentinel species, there are a few perceived disadvantages which are described next:

1. Migratory behavior as a factor confounding contaminant exposure—The osprey is migratory over much of its range, spending about half of its life away from the nesting area. Concern exists regarding the relative contribution of contaminants accumulated on the breeding grounds and/or during migration with that accumulated on the breeding grounds prior to egg laying. Ospreys arrive on the nesting grounds in early spring, spending approximately 1 mo (while feeding on local fish) building or rebuilding their nests prior to egg laying. The University of Minnesota Raptor Center in cooperation with the U.S. Geological Survey, plus the Canadian Wildlife Service, captured adult female ospreys at their nests in Oregon, Washington, and British Columbia, equipping them with satellite radio transmitters to determine migration routes and wintering areas (Martel et al., 2001; Elliott et al., 2007). An egg was also collected from each of a series of nests incubated by radio-tagged adult females. The radio-tagged females were tracked to their wintering areas in southern Mexico and Central America, where prey fish of appropriate size were collected for chemical analyses. Ospreys were found to consistently return to the same wintering areas year after year, with a rapid migration (mean duration 13 d) from the Pacific Northwest to their wintering grounds. Though DDE concentrations in osprey eggs ranged from 0.02 to 10.14 \( \mu \)g/g (wet weight, ww), fish collected from wintering locations in Latin America did not significantly contribute to the total \( p,p' \)-dichlorodiphenyltrichloroethane (DDT) or DDE egg concentrations (Elliott et al., 2007). Rather, DDE concentrations in osprey eggs were strongly correlated to concentrations found in fish from their nesting areas, with the highest DDE egg concentrations found in nests along the lower Columbia River. An earlier study of bioaccumulative PCDD in osprey eggs from British Columbia documented significantly higher concentrations in nests downstream of pulp and paper mills than in upstream nests, another indication that lipophilic contaminants were accumulated on the nesting grounds (Elliott et al., 1998). Both studies indicate that contaminant accumulation in eggs occur primarily on the nesting grounds for ospreys from the Pacific Northwest. Local contaminant uptake, as reflected in contaminant levels of fish on the breeding grounds, also contributed significantly to eggs of ospreys nesting in Delaware Bay (Steidl et al., 1991). Osprey nestlings (~1500 g at 6 wk of age) can also be used to evaluate localized contaminant accumulation and possible effects (Martin et al., 2003),
as they have grown 25- to 30-fold from a 50- to 60-g egg and have greatly diluted the maternal contaminant contribution (also acquired locally). In the case of Hg, both feather and blood samples from nestlings can be collected nonlethally to determine local accumulation and possible effects.

2. Suitability for use in laboratory studies—The osprey is not amenable to conducting laboratory studies, primarily due to difficulties related to captive rearing (Poole, 1989); there are no records of ospreys successfully breeding in captivity. However, osprey eggs were collected from nests in the field and successfully incubated under laboratory conditions to determine developmental effects and biochemical changes resulting from maternal contaminant exposure (Elliott et al., 2001). Yolk-sac samples collected from freshly hatched osprey chicks were analyzed for contaminants and compared with cytochrome P-450 (P450) 1A protein levels and ethoxyresorufin O-deethylase (EROD) activity in the liver, vitamin A and retinol concentrations in the blood, liver, and kidneys, and porphyrin levels in the liver, and with histological observations. Laboratory studies showed that developing bird embryos are consistently more sensitive than other life stages to contaminants like PCB, TCDD and Hg (Birge & Roberts, 1976; Hoffman et al., 1996; Heinz et al., 2008). Thus, the artificial incubation technique of field-collected eggs permits the evaluation of deformities, edema, morphology, and a variety of biochemical and physiological endpoints associated with contaminant exposure. Effects of the DDT on osprey reproduction (separation of intrinsic influence [egg] from extrinsic effects [adult behavior, food availability]) were evaluated in the late 1960s by the exchange of eggs and young from relatively clean areas in Maryland to heavily contaminated areas in New Jersey (Spitzer, 1978). Tolerance of short-term nest disturbance and the species nesting on accessible structures permitted this approach to work successfully.

OSPREY CONTAMINANT RESEARCH HISTORY

Changes in North American osprey populations received little attention until the mid-1960s when Ames and Mersereau (1964) documented the population crash of the Connecticut River nesting colony in the northeastern United States, and implicated pesticides. Within a few years, exceptionally poor productivity and declining population numbers were reported at many locations (Henny, 1977). At that time with seemingly all populations declining, nobody knew how many young should be produced per nest in a normal situation. A population model with survival rate estimates based on North American banding data (primarily from eastern United States) was developed, including the estimated production rates necessary to maintain a stable population (Henny & Wight, 1969; Henny et al., 1970). The initial 0.95 to 1.3 young per nesting attempt believed necessary to maintain a stable osprey population was further refined to 0.8 young per nesting attempt by Spitzer (1980), who compared observed rates of population change with expected rates of population change. The osprey population was initially classified “status undetermined” by the U.S. Fish and Wildlife Service (1973). This classification probably resulted in studies of most osprey populations over the last four decades throughout North America, with studies also conducted in Europe and elsewhere. In fact, nationwide osprey population estimates were made for the United States in 1981 (Henny, 1983) and again in 1994 (Houghton & Ryman, 1997). Much of the research has been directed toward understanding the status of and factors influencing local populations, including pesticides and industrial contaminants. Thus, the osprey’s biology, ecology, and position in the aquatic food chain became better understood (Poole, 1989).

In addition to long-term monitoring of regional osprey population numbers, local populations were studied to evaluate selected contaminants: (1) DDT and other OC pesticides, PCDD, PCDF, PCB, Hg and other metals, including possible effects on reproduction (Tables 1–3), (2) lead (Pb) exposure and accumulation from mining activities and possible effects on blood parameters, survival, and reproduction, (3) various contaminants, to estimate empirical biomagnification factors from fish to osprey eggs, and (4) emerging contaminants, including PBDE, PFOS, herbicides, and fungicides, to determine exposure and possible effects.
TABLE 1. Osprey Studies that Included Tissues Analyzed for Organochlorine Pesticides

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*Number of samples analyzed.

Regional Long-Term Population Studies of Nesting Ospreys in the United States, Canada, and Mexico

Early U.S. Fish and Wildlife Service aerial surveys (using partial double-sampling by ground/boat) of nesting ospreys in Chesapeake Bay, the Carolinas, New Jersey, Delaware, coastal Virginia,
California, Oregon, and northwestern Mexico (Baja California, Sonora, Sinaloa) (Henny & Noltemeier, 1975; Henny & Anderson, 1979; Henny et al., 1974, 1977, 1978a, 1978b) provided the base for many recent population comparisons. These 1970s estimates likely represented population lows for most of the areas surveyed. A summary of estimated osprey pairs nesting in each state was prepared

### Table 2. Osprey Studies that Included Tissues Analyzed for PCBs, PCDDs, PCDFs

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*Number of samples analyzed.
for 1981 (Henny, 1983) and again for 1994 (Houghton & Ryman, 1997). More recent surveys showed dramatic increases in nesting numbers of ospreys for both North America and Europe (Henny et al., 1991, 2004, 2008c; Henny & Kaiser, 1996; Meyberg et al., 1996; Ewins, 1997; Saurola, 1997; Henny & Anderson, 2004; Watts et al., 2004; Nygård et al., 2006; Saurola, 2005; Zachos & Schmölecke, 2006). Increases in nesting numbers now appear to be limited only by nest site availability, food resources, and perhaps exposure to contaminants in heavily polluted areas.
As recently as 1997/1998, DDE still impacted reproductive success of a small portion of nests along the lower Columbia River, and although the nesting population was increasing, the rate of increase was slower compared to the nearby Willamette River where DDE levels in fish and osprey eggs were lower (Henny et al., 2008b).

**Effects of OCs on Mortality and Reproduction**

For over 50 yr, scientists studied the exposure, accumulation and effects of DDT, its breakdown product DDE, heptachlor, dieldrin, chlordane, endrin, and other OC in birds, especially as they relate to reproductive impairment. All are neurotoxic agents that either affect ion permeability or interact with nerve receptors. Exposure to these pesticides has yielded adult mortality, reduced egg production, eggshell thinning, decreased fertility and hatchability, and decreased survival of young (Lundholm, 1997; Henny & Elliott, 2007). However, concentrations of the highly lipophilic pesticides in eggs and bird tissues (e.g., plasma, liver, brain) are highly correlated with each other, as reported in several bird species (Enderson & Berger, 1970; Vermeer & Reynolds, 1970; Bogan & Newton, 1977; Henny & Meeker, 1981).

As with the bald eagle, some osprey carcasses were collected in the eastern United States and Ontario, Canada, from 1964 to 1982 (Table 1). Carcasses of 68 ospreys were collected for necropsy to determine the cause of death and general body condition, with tissue samples of selected birds analyzed for a series of OC (Gilbertson & Reynolds, 1974; Sundlof et al., 1986; Wiemeyer et al., 1975, 1980, 1987). DDE concentrations in the brains of ospreys ranged from <0.1 to 30 μg/g (ww) and from <0.1 to 48 μg/g (ww) in carcasses. Dieldrin in the brain ranged from <0.01 to 7.5 μg/g (ww), with carcass residues from <0.1 to 9.5 μg/g (ww). The authors were cautious in interpreting OC body burdens because some birds had little body fat in comparison to others. Fat depletion results in contaminant mobilization and redistribution, which increases concentrations in other tissues such as the brain (Bogan & Newton, 1977; Wiemeyer et al., 1980). Wiemeyer et al. (1980) showed a significant negative correlation in ospreys between percent lipid in the carcass and the brain–carcass ratio of DDE, with redistribution of DDE from the carcass to the brain occurring as lipid reserves were depleted. The redistribution of other OC and chlorinated hydrocarbons also occurred, and was also shown in other bird species (Bogan & Newton, 1977; Van Velzen et al., 1972). Blus (1996) reviewed the DDE literature and concluded that a lower lethal limit in the brain of 300 μg/g (ww) DDE seems to provide a reasonable criterion for birds as a whole, though evidence exists that lower concentrations can prove lethal. In a study by Porter and Wiemeyer (1972), two experimental kestrels (Falco sparverius) died after a long dosage period on a dietary dosage of 2.8 μg/g DDE, with 213 and 301 μg/g DDE (ww) in their brains. Three kestrels in another laboratory study died several days after receiving diets containing 160 to 250 μg/g of DDE; their brains had 230 to 280 μg/g DDE (Henny & Meeker, 1981). None of the brain DDE concentrations of ospreys found dead exceeded 50 μg/g; thus, none were suspected of dying from DDE poisoning. However, two extremely emaciated ospreys were suspected of dying from dieldrin poisoning with brain concentrations of 7.5 and 3.8 μg/g (ww) (Wiemeyer et al., 1975, 1980). Their brain concentrations of dieldrin approached and exceeded the suggested 4 μg/g by Stickel et al. (1969) and supported as a reasonable diagnosis by Peakall (1996).

Of primary interest are the well-documented effects of DDE on avian reproduction and eggshell thinning. Along with a severe annual decrease of 30% in the nesting population of ospreys observed by Ames and Mersereau (1964) on the Connecticut River for the years 1960–1963 was the exceptionally poor reproduction of 0.23 young fledged per active nest. Similar declines in reproduction were observed elsewhere along the East Coast of the United States (Ames & Mersereau, 1964; Ames, 1966; Wiemeyer et al., 1975; Puleston, 1977; Wiemeyer, 1977). A large number of eggs failed to hatch, with some eggs disappearing during incubation for reasons unknown (Wiemeyer et al., 1975). Though nestling survival appeared normal, the number of young fledged per active nest was well below the 0.8 young needed to maintain a stable osprey breeding population (Spitzer, 1980). Ratcliffe (1967) noted the incidence of broken eggs in nests of peregrine falcons (Falco peregrinus), sparrowhawks (Accipiter nisus), and golden eagles (Aquila chrysaetos) in Great Britain had increased considerably since the early 1950s, with eggshell mass of the respective species 19,
24, and 8% less than mean eggshell mass of the same species collected before 1947. Ratcliffe (1968, 1970) suggested that the decrease in eggshell mass (thinning) may be related to the accumulation of persistent OC like DDT, which had been shown experimentally to disturb calcium physiology in birds. The widespread agricultural use of OC in the late 1940s in Great Britain coincided closely with the onset of eggshell change observed in several raptor species. The first field studies to statistically associate decreasing eggshell thickness with increasing DDE concentrations in the eggs of several bird species were those of Hickey and Anderson (1968), Anderson et al. (1969), and Fyfe et al. (1969). A controlled laboratory study also confirmed that DDE induced significant eggshell thinning, cracking, and embryo mortality using penned mallards (Heath et al., 1969). Heath et al. (1969) also showed that DDT and DDD impaired reproduction, but less severely than DDE. These studies supported Ratcliffe’s hypothesis, associating catastrophic declines of bird species (including the osprey) with decreasing eggshell thickness and implicating DDE as the causative agent. The major cause hypothesized for low osprey productivity in the 1960s was failure of eggs to hatch, rather than adult fertility and ability to breed. Eggs exchanged in 1968 and 1969 between nests from Connecticut and nests in Maryland, where hatching rates were much higher, showed that hatching failure was intrinsic with the egg and did not include abnormal parental behavior or human disturbance (Wiemeyer et al., 1975). The eggshells from Connecticut were also 15 to 20% thinner than the mean eggshell thickness of osprey eggs collected from the eastern United States before 1947 (Anderson & Hickey, 1972). Several studies have since looked at the relationship between osprey eggshell thinning and DDE (Table 1). Lundholm (1997) reviewed the effects and mechanism of action of DDE on eggshell formation in birds. The inhibition of prostaglandin synthesis in the eggshell gland mucosa is considered the mechanism for DDE-induced eggshell thinning. Ewins et al. (1999) reviewed OC accumulation in eggs of known-age osprey females from the 1980s and found no significant difference in OC concentrations (ww) associated with female age or the year eggs were collected. The age of the female also did not affect eggshell thickness. Ewins et al. (1999) found that eggs of females sampled over multiple years showed considerable variation in OC concentrations, reflecting possible recent exposure of the female and/or the extent the female used endogenous fat reserves to form the eggs.

Wiemeyer et al. (1988) reported that 15 and 20% eggshell thinning of osprey eggs was associated with 4.2 and 8.7 μg/g (ww) DDE, respectively, Wiemeyer et al. (1988) also found no association between concentrations of PCB and eggshell thinning, nor any synergistic/additive effects with DDE. Though no direct data were available on osprey productivity related to DDE, Lincer (1975) noted that not one North American raptor population exhibiting ≥18% eggshell thinning was able to maintain a stable self-perpetuating population. Therefore, mean DDE concentrations in osprey eggs between 4.2 and 8.7 μg/g would be expected to result in a declining population. Data from our osprey nesting studies in the Pacific Northwest from 1993 to 1998 verified the reproductive relationships (Henny et al., 2003, 2004), with eggs at nests containing <4.2, 4.2–8, and ≥8 μg/g DDE from the Columbia River System producing 1.61, 1.25, and 1 young/active nest, respectively. Eggshell thickness followed a classic semilogarithmic DDE response with the three DDE categories –3.4%, –12.7%, and –17%. By 2004, DDE concentrations in osprey eggs from the Lower Columbia River were all below the 4.2-μg/g threshold and no longer influenced osprey reproduction (Henny et al., 2008a). The decline of OC was shown in other studies throughout the United States, Canada, Europe, and elsewhere. The nesting population of ospreys will continue to grow, at least until nest site availability, food, or other factors become limiting.

Some evidence suggests that DDE is embryotoxic in osprey eggs at concentrations approaching 10 μg/g (ww). Elliott et al. (2001b) found a higher percentage of osprey eggs failing to hatch at concentrations above 6 μg/g DDE, with all eggs above 10 μg/g failing to hatch in an artificial incubation study. Mean concentrations of other OC reported during the incubation study were lower in eggs that failed to hatch than those associated with embryotoxicity in other laboratory and field studies (Noble & Elliott, 1990). They speculated that the critical concentration of DDE affecting hatching success of ospreys was toward the higher end of the 5 to 10 μg/g range. In northern Idaho (1972 and 1973), 3 of 8 osprey nests with a randomly collected egg failed to produce young, and the sample eggs contained 12, 14, and 15 μg/g DDE (Johnson et al., 1975). The authors, though, could not separate hatching failure from egg breakage or mortality of chicks. However, an egg collected from a
nest in northern Idaho on the Coeur d’Alene River (1986) with 48.1 μg/g DDE (ww) managed to fledge one young (Henny & Anthony, 1989). Thus, a given threshold point above which all nests fail is probably not a useful concept; a graded reproductive response seems to be the norm, as for the white-faced ibis (Henny & Herron, 1989).

**PCB, PCDD, and PCDF**

PCB, PCDD, and PCDF are known for their toxicity to biological organisms, eliciting a number of responses that include, but are not limited to, mortality, thymic atrophy, immunotoxic effects, reproductive impairments, porphyria, and related liver damage, with birds exhibiting varying degrees of sensitivity when exposed to these compounds (Hoffman et al., 1996). Several laboratory studies suggest an approximate brain concentration of 300 μg/g (ww) is indicative of probable PCB toxicosis in several species of birds, though cormorants appear much more sensitive (Prestt et al., 1970; Koeman et al. 1973; Stickel et al., 1984). PCB residues in the carcass and brain of the 68 osprey carcasses mentioned earlier were all below 50 μg/g (ww), except in 1 male collected in South Carolina. It was emaciated and had a carcass and brain concentration of 140 and 220 μg/g (ww) PCBs, respectively. Osprey carcasses were not analyzed for PCDD or PCDF, and no mortality reports were found from other studies of PCDD or PCDF toxicity.

Bird embryos are much more sensitive to the adverse effects of PCB than adults, as determined by dietary and egg injection studies using Aroclor and congener PCB mixtures (Barron et al., 1995; Hoffman et al., 1996). Adverse effects of embryos resulting from PCB exposure include pericardial edema, cardiovascular malformations, liver lesions, external deformities, thymic hypoplasia, lack of lymphoid development, and subcutaneous edema (Gilbertson et al., 1991). Sensitivity to PCB toxicity in eggs does vary considerably among bird species (Hoffman et al., 1996). However, ospreys appear to tolerate rather high concentrations of PCB in eggs, with little effect on overall production rates. Several studies failed to identify any effects associated with PCB in eggs on osprey reproduction, even at a mean concentration of 25 μg/g (ww) (Wiemeyer, 1988; Poole, 1989; Elliott et al., 2001b; Martin et al., 2003). This is similar to what Elliott and Harris (2001/2002) suggested for the bald eagle as the reproductive effect threshold (20 μg/g) after reviewing existing data, while Helander et al. (2002) determined a lowest observable effect level of 25 μg/g on productivity for the white-tailed sea eagle (Haliaeetus albicilla). Spitzer et al. (1978) found osprey populations recovering rapidly in Connecticut and Long Island, New York, after DDT was banned, despite continued high concentrations of PCB during the same time period. Except for sites of heavy contamination, several osprey studies showed PCB gradually decreasing in the environment, with 50% decrease in tissues every 5 to 10 yr (Steidl et al., 1991; Audet et al., 1992; Clark et al., 2001; Henny et al., 2004, 2008).

Martin et al. (2003) determined plasma concentrations of PCB in osprey nestlings to further understand local contaminant uptake and compared findings with egg concentrations. Relative contribution of PCB congener groups (tri-, tetra-, penta-, hexa-, hepta-, octa-, nona-) in nestling plasma samples generally reflected that found in osprey eggs collected from the same region. In fact, eggs and plasma from one region reflected the unique PCB congener profile consistent with unweathered Aroclor 1254. This, the authors suggested, further supports the use of eggs and nestling plasma to evaluate localized PCB sources.

The industrial by-products of PCDD and PCDF were first evaluated in osprey eggs in the early 1990s after being reported in other fish-eating species of birds associated with bleach-kraft pulp mills, where elemental chlorine was used as a bleaching agent (Elliott et al., 1998; Woodford et al., 1998). Several studies showed significant uptake of PCDD and PCDF in osprey eggs from nests downstream of kraft pulp mills compared to nests upstream (Whitehead et al., 1993, 1995; Elliott et al., 1998; Woodford et al., 1998). Egg concentrations of TCDD ranged from less than the detection limit to 162 pg/g (ww), with most studies reporting concentrations <100 pg/g TCDD (Whitehead et al., 1993, 1995; Augspurger et al., 1996; Elliott et al., 1998, 2001a; Woodford et al., 1998; Martin et al., 2003; Henny et al., 2003, 2004, 2008; Jiménez et al., 2007). PCDD and PCDF concentrations reported did not appear to affect either osprey hatching success or fledging of young, although Woodford et al. (1998) reported lowered growth rates of nesting ospreys from a contaminated site when compared to a reference site in Wisconsin. Toxic equivalent
concentrations (TEQs), derived from toxic equivalency factors of select non- and mono-ortho PCB, PCDD, and PCDF based on their structural similarity and toxicity relative to TCDD, were also used to evaluate reproductive and biochemical endpoints for osprey eggs and young. Woodford et al. (1998) indicated that the no-observable-adverse-effect concentration for osprey embryo survival was equal to or greater than 136 pg/g (ww) in eggs for the combined TEQ (PCB, PCDD, PCDF). Findings from Elliott et al. (2001a) supported the 136 pg/g TEQ value in whole eggs suggested by Woodford et al. (1998) for hatching in ospreys as a conservative estimate given the available data. Elliott et al. (2001a) also found significant correlations between hepatic CYP1A protein concentrations of day-old osprey chicks and whole-egg TEQ concentrations and various PCB congeners. They concluded that stereoisomers of TCDD (primarily PCB 126) induced production of hepatic CYP1A protein and EROD activity at a lowest observed effect/response concentration of 130 pg/g TEQ for day-old chicks.

Significant decreases in concentrations of lower chlorinated PCDD and PCDF (based on osprey egg concentrations) occurred since the curtailment of chlorine bleaching at kraft pulp mills in the early 1990s (Elliott et al., 1998; Henny et al., 2008a, 2008b). For example, by 1992 all kraft pulp mills in southern British Columbia, Canada, had changed their bleaching process from chlorine to minimize PCDD/PCDF formation. Between 1988 and 1992, concentrations of TCDD in mountain whitefish (Prosopium williamsoni) collected from the Thompson River below a mill site decreased from 61 to 1.7 pg/g (ww), while 2,3,7,8-terachlorodibenzofuran (TCDF) declined from 390 to 8.2 pg/g (Elliott et al., 1998). Concentrations of PCDD/PCDF in osprey eggs collected near mill sites also showed substantial declines. However, more highly chlorinated PCDD and PCDF congeners in osprey eggs and nestling plasma samples collected from sites throughout the Northwest were still present, with no apparent association to pulp mills (Elliott et al., 1998; Henny et al., 2003). Fifteen eggs from sites in British Columbia, which had the highest mean concentrations of total heptachlorodibenzo-p-dioxin and octachlorodibenzo-p-dioxin, were analyzed for chlorophenolic compounds (Elliott et al., 1998). Only pentachlorophenol was detected in the osprey eggs, and was quantified in 8 of 15 samples. Elliott et al. (1998) concluded that pentachlorophenol (locally used as a wood preservative) was the likely source for the higher chlorinated PCDD and PCDF.

**Mercury**

Mercury (Hg) frequently is a contaminant of concern in aquatic ecosystems. Human health advisories have been issued in more than 40 states relating to consumption of Hg-contaminated fish. In addition to the human threat, a potentially serious threat exists for piscivorous wildlife like the osprey whose diet is comprised almost exclusively of fish. The Madison Declaration on Mercury Pollution (MDMP, 2007, p. 63) warned that “long-lived piscivores and other top predators atop aquatic food webs are at greatest risk for elevated methylmercury (MeHg) exposure, accumulation and toxicity.” Atmospheric deposition is the primary source of Hg for most ecosystems, although localized geologic and anthropogenic point-sources (e.g., historic mining and smelting, coal-fired power plants) are also important. Methylation of inorganic mercury (IoHg) to the potent neurotoxin MeHg is an important process that occurs primarily in aquatic ecosystems. MeHg readily bioaccumulates in living organisms, which over time may reach toxic levels. Birds eliminate a substantial amount of MeHg through feather growth and molting, with females eliminating additional MeHg via transfer to eggs (Bäckström, 1969; Furness et al., 1986; Thompson & Furness, 1989; Burger & Gochfeld, 1992; Becker et al., 1994). Hg in blood, feathers, and eggs is almost always present as MeHg and therefore can be analyzed less expensively as total mercury (THg).

Though the use of wild birds in controlled laboratory breeding studies is the best way to evaluate harmful Hg concentrations in eggs, the great expense and time required to establish captive breeding colonies of wild birds for Hg dosing studies make it unlikely that these types of studies will be conducted (Heinz et al., 2008). As an alternative, eggs collected from the nests of wild birds can be brought into the laboratory and injected with graded concentrations of Hg. Heinz et al. (2008) conducted such studies, injecting varying doses of MeHg into the air cells of eggs of 26 species of birds, including the osprey, and examined the species specific dose-response curves of embryo survival. Of the bird species tested, the osprey was categorized as exhibiting high sensitivity (LC₅₀).
was less than 0.25 μg/g MeHg). When comparing data from their injection studies to that of published studies where MeHg was fed to breeding adult females and then deposited into the egg by, Heinz et al. (2008) found the injected MeHg to be more toxic than the same concentration deposited naturally. Concentrations of Hg in osprey eggs reported in the literature rarely exceed 0.50 μg/g (ww).

During growth of new feathers, MeHg in the blood is bound to keratin at high concentrations. As feather growth proceeds, MeHg levels decrease in the body (an important detoxification process) as the contaminant is sequestered into the newly formed feathers, which are eventually lost during molt. Several studies evaluated Hg accumulation in osprey tissues and feathers throughout North America and Europe (Table 3). Juvenile and adult feathers were found to account for 85 and 93% of the “MeHg” body burden in ospreys (Hughes et al., 1997). Though MeHg in nestling feathers is a reliable indicator of local contamination, MeHg in molted feathers of adults during the nesting season may not result from local MeHg contamination; feathers may have been grown elsewhere under different exposure patterns if the species is migratory, especially for larger birds with irregular or incomplete molts. However, contrary to an earlier study from Sweden (Jensen et al., 1972), a recent osprey study from the Carolinas (USA) suggests that the molt sequence of adult primary feathers does not influence MeHg concentrations (Hopkins et al., 2007). Finally, evidence from studies of other fish-eating birds (egrets, herons) shows that they can demethylate MeHg to a less toxic form in a concentration-dependent manner. The demethylated Hg is sequestered with selenium in the liver and kidneys, providing additional protection from some of the more toxic MeHg; i.e., less would appear in blood, feathers and eggs (Henny et al., 2002).

### Lead

Few studies have been conducted on Pb exposure and accumulation in ospreys (Table 3). In one study, mining and smelting activities at Kellogg, ID, resulted in Pb contamination throughout the Coeur d’Alene River Basin where ospreys nested (Henny et al., 1991). Both adults and nestlings from the Coeur d’Alene River demonstrated elevated blood Pb concentrations compared to those found in the upper portion of Coeur d’Alene Lake and reference areas at Lake Pend Oreille, Idaho, and Flathead Lake, Montana. Pb concentrations in fish collected from the same areas paralleled those found in osprey blood. Inhibition of δ-aminolevulinic acid dehydratase (ALAD) activity and elevated protoporphyrin in blood of adult and nestling ospreys was evidence of Pb exposure. Adult ALAD activity (an important enzyme in the heme synthesis pathway) was negatively correlated with blood Pb concentrations, whereas protoporphyrin (a porphyrin precursor) was positively correlated. Neither hemoglobin nor hematocrit was adversely affected by the relatively modest Pb levels found in adult and nestling osprey blood, especially when compared to ducks, geese, and swans that were ingesting Pb contaminated sediment along with vegetation. No observed behavioral abnormalities, reduced productivity, or mortality associated with Pb was observed in ospreys. In fact, ospreys produced young at nearly identical rates for the Coeur d’Alene River, intermediate and reference areas, which were among the highest reported reproductive rates for the western United States at the time. Ospreys often do not ingest fish bones (where Pb preferentially accumulates), and if ingested, most are not digested (regurgitated as pellets), which limits Pb exposure. Furthermore, Pb has limited transferability to eggs (Pattee, 1984; Leonzio & Massi, 1989), thus reducing potential embryonic toxicity.

### Determination of Biomagnification Factors (BMF) Between Fish Prey Species and Osprey Eggs

In order to evaluate toxicological effects resulting from environmental contaminant exposure, it is necessary to understand the relationship between contaminant concentrations found in fish eaten by ospreys and the resultant accumulation in osprey eggs, i.e., the BMF. To document this relationship, a series of osprey eggs and fish species important in the ospreys diet were collected and analyzed from the Willamette River in 1993 (Henny et al., 2003). BMF were estimated for each contaminant by using concentrations in the egg divided by concentrations in each fish species found in the ospreys’ diet (weighted by the fraction of each prey species in the diet on a biomass basis). The
BMF can be expressed on a wet mass basis, or preferably as the ratio normalized to lipid content because of differences in lipid content between eggs and fish. To validate these findings, a second series of osprey eggs and fish were collected in 2001 at similar locations on the Willamette River (Henny et al., 2008b). The second collection was used to determine concentration changes in the interim for both osprey eggs and fish, and to evaluate the consistency of the BMF estimates.

Empirical BMF estimates calculated varied for each contaminant depending upon its chemical properties; e.g., PCB congeners reflected 10- to 20-fold increases, with a 6.7-, 25-, 87-, and 174-fold increase for dieldrin, heptachlor epoxide, DDE- and OCDD, respectively. These findings demonstrated that contaminants with the same concentrations in prey fish do “not” yield equivalent concentrations in osprey eggs. Osprey eggs and fish collected in 2001 indicated that most contaminant concentrations decreased during the 8-yr interim in both osprey eggs and fish (Henny et al., 2008b); however, BMF values remained consistent during both time periods, e.g., DDE 87 versus 79. This confirmatory evidence lends further support to the utility of the BMF concept in understanding contaminant-specific uptake dynamics and validity of using the osprey as a sentinel species. Empirical BMF values can also be used in risk assessments to estimate both osprey egg concentrations if fish residue data are available and percent biomass of each fish species ingested is available, or, conversely, to “back calculate” average fish tissue concentrations from osprey eggs. However, it needs to be noted that to determine fishing locations for ospreys (either on the water body of concern, or elsewhere on adjacent ponds and lakes), as well as dietary information during the pre-egg-laying stage, can take considerable effort. Furthermore, the diet can sometimes change during the nesting season as ospreys take advantage of local fish runs or hatchery releases.

Emerging Contaminants

Emerging contaminants (e.g., brominated flame retardants, perfluorooctane sulfonate compounds, etc.) have received increasing interest, especially those that potentially biomagnify up food webs and accumulate in top predators. While the chemical characteristics of some emerging contaminants may not result in bioaccumulation in ospreys or transfer to their eggs, several contaminants do have the potential to accumulate. Understanding the risks posed by such contaminants will require research, including the development of analytical approaches to document their ecotoxicological kinetics in avian species. As an initial study, a series of 15 osprey eggs was collected in 2003 from Duwamish Waterway, Lake Washington, and Elliott Bay, near Seattle plus Everett, WA, to evaluate the uptake of emerging contaminants. Based on predicted chemical structure–activity relationships (SAR), 12 currently used chlorophenoxy herbicides along with the fungicide chlorothalonil were hypothesized to accumulate in osprey eggs. Of those compounds, chlorothalonil, the herbicide dacthal, and its isomer dimethyl tetrachlorophthalate (diMe-TCP) were consistently found, though at low picograms per gram (ww) concentrations (Chu et al., 2007). To our knowledge, this is the first time dacthal and diMe-TCP were reported in bird eggs. Their presence in osprey eggs indicates the ability to bioaccumulate in the aquatic food web, and to persist to the top trophic level. In ovo transfer of these contaminants from the osprey female to eggs is of concern, because of potential effects on the health of the developing chick embryo and overall health of ospreys.

Polybrominated biphenyl ethers (PBDE) are a class of flame-retardant additives, extensively used in a wide variety of products such as circuit boards and plastics of electronic equipment, polyurethane foam used in furniture, textiles, and other materials since the 1960s (de Wit, 2002; Siddiqi et al., 2003). PBDE are structurally similar to PCB, and are mixed with but not covalently bound into polymer matrices. Over time, these flame retardants diffuse or leach out of their respective matrices, dispersing widely via air and water. Main point sources are landfills and sewage effluent (Siddiqi et al., 2003, Gevao et al., 2008). As with PCB, PBDE are persistent, remaining in the environment for years with minimal degradation. Lower brominated congeners are highly lipophilic and bioaccumulate/biomagnify into fats of animal tissue. PBDE were found in environmental samples since the late 1970s, and are increasing in concentration throughout the food web at an alarming rate. PBDE were found in fish tissues worldwide, with concentration-doubling varying between 1.6 and 4 yr (Rayne et al., 2003; Zhu & Hites, 2004; Johnson-Restrepo et al., 2005; Batterman et al.,
2007). Not surprisingly, PBDE were also reported in the tissue and eggs of several species of fish-eating birds, including ospreys (Jansson et al., 1993; Elliott et al., 2005; Herzke et al., 2005; Rattner et al., 2004; Toschik et al., 2005; Voorrips et al., 2006; Gauthier et al., 2007; Verreault et al., 2007). Osprey eggs collected from the eastern United States, British Columbia, Canada, and Norway contained PBDE residues ranging from 8 to 928 μg/kg (ww). Though toxicity of PBDE is not well understood, they are believed to act on biological systems like PCB. Continued monitoring of PBDE in aquatic systems is warranted due to their marked increases in tissues of ospreys and other fish and wildlife species over the past two decades (Norstrom et al., 2002; She et al., 2002; Law et al., 2003; Rayne et al., 2003; Zhu & Hites, 2004; Elliott et al., 2005).

Perfluorinated acids and sulfonate compounds were evaluated in osprey eggs collected in Chesapeake Bay, the Delaware River, and Delaware Bay of the eastern United States (Rattner et al., 2004; Toschik et al., 2005). This family of compounds was used for decades to make commercial products resistant to heat, oil, grease, stains, and water, such as stain-resistant carpets, fabrics, firefighting foam, and nonstick cookware. Concentrations of perfluorooctanesulfonate in osprey eggs from Chesapeake and Delaware Bays were comparable to concentrations found in egg yolks and blood serum and plasma of other waterbirds in the United States, Japan, and Norway (Kannan et al., 2001; Taniyasu et al., 2003; Verreault et al., 2005). Little is known of perfluorinated compound toxicity in birds, though laboratory studies with rodents suggest these compounds are hepatotoxic, induce CYP4A isoenzymes, and reduce embryo viability (Harris & Birnbaum, 1989; Chinje et al., 1994).

**FUTURE RESEARCH DIRECTIONS**

The findings of dacthal isomers, chlorothalonil, and brominated flame retardants in Pacific Northwest ospreys illustrates that they are constantly exposed to an increasingly complex profile of bioaccumulative contaminants. Contaminants like PBDE, herbicides, fungicides, pharmaceuticals, and personal care products are presently found in a variety of aquatic environments, raising concerns about their potential for accumulation, fate, and effects (Richardson et al., 2005; Siddiqi et al., 2003; Batterman et al., 2007). Some of these contaminant concentrations (e.g., PBDE) are increasing at alarming rates. New extraction and analytical methods are being developed to identify and quantify these emerging contaminants, as illustrated with the determination of alkylphenol and alkylphenol-ethoxylates in osprey eggs from the Chesapeake Bay (Schmitz-Afonso et al., 2003). Continued research evaluating potential sources, quantitative SAR, and food web dynamics is needed to understand the ecological implications of these emerging contaminants. Well-designed field studies must certainly play an important role in understanding exposure patterns and effects on wildlife.

**CONCLUSION**

In order to effectively measure contaminant distribution and effects with avian species, it is crucial to select a species that will optimize the usefulness of information collected. A suitable sentinel species needs to respond to chemical insults manifested by a broad spectrum of pathologic conditions, including reproductive and behavioral dysfunctions, immunologic and biochemical perturbations, and anatomic changes (birth defects). It is recognized that no single avian species can be used to effectively evaluate all types of contaminants, which necessitates an understanding of the physiology of animals used and the chemistry and toxic mechanisms of contaminants in question. Much is known about the life history and biology of the osprey, and it has been used successfully as a sentinel species in aquatic ecosystems to document exposure and effects of many chemical contaminants. Spatial nesting at regular intervals has allowed evaluation of contaminants at random along large river segments or at strategic locations above and below known or suspected contaminant point sources. Identifying regional diets of osprey based on biomass allows for determination of BMF between fish prey species and osprey eggs. This approach is now being used in risk assessments associated with Superfund sites. The osprey has also proved useful in evaluating potential exposure and potential effects of several emerging contaminants. To date, however, only a limited
number of studies have compared contaminant concentrations in osprey tissues to currently used biochemical biomarkers (Henny et al., 1991; Elliott et al., 2001b). Additional research is needed to better understand uptake kinetics, effect levels, and responses to emerging contaminants, which will require the development of new methodologies. These considerations aside, the osprey has many characteristics delineated in this article that make it an excellent candidate for these new investigations.

The studies proposed earlier in this article (especially collecting an egg in early incubation) are not feasible using the other top aquatic raptor of North America, the bald eagle. Studies using the bald eagle, due to the eagle’s sensitivity to nest disturbance, would result in considerable nest abandonment (Grier, 1982; Anthony et al., 2007). Fortunately, fish-eating ospreys are present at many locations within the United States and elsewhere in the world. Osprey populations have recovered from the DDT era at almost all locations, and have now pioneered back into some of the most industrialized and polluted regions where they can now play a critical role addressing continued legacy and emerging contaminant issues.

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