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Ecological risk of methylmercury in Everglades National Park, Florida, USA

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Abstract Dramatic declines in mercury levels have been reported in Everglades biota in recent years. Yet, methylmercury (MeHg) hot spots remain. This paper summarizes a risk assessment of MeHg exposure to three piscivorous wildlife species (bald eagle, *Haliaeetus leucocephalus*; wood stork, *Mycteria americana*; and great egret, *Ardea albus*) foraging at a MeHg hot spot in northern Everglades National Park (ENP). Available data consisted of literature-derived life history parameters and tissue concentrations measured in 60 largemouth bass (*Micropterus salmoides*), 60 sunfish (*Lepomis* spp.), and three composite samples of mosquitofish (*Gambusia holbrooki*) collected from 2003 to 2005. To assess risk, daily MeHg intake was estimated using Monte Carlo methods and compared to literature-derived effects thresholds. The results indicated the likelihood was very high, ranging from 98–100% probability, that these birds would experience exposures above the acceptable dose when foraging in northern ENP. Moreover, the likelihood that these birds would experience exposures above the lowest-observed-adverse-effect level (LOAEL) ranged from a 14% probability for the wood stork to 56% probability for the eagle. Data from this study, along with the results from several other surveys suggest that biota in ENP currently contain the highest MeHg levels in South Florida and that these levels are similar to or greater than other known MeHg hot spots in the United States. Given these findings, this paper also outlines a strategic plan to obtain additional measured and modeled information to support risk-based management decisions in ENP.

Keywords Methylmercury · Risk · Bald eagle · Wood stork · Great egret

Introduction

Elevated mercury (Hg) levels were first reported in biota from Everglades National Park (ENP) in 1974 (Ogden et al. 1974). High levels were later found to be widespread in fish from the entire Greater Everglades region, i.e., ENP and the Water Conservation Areas (WCAs) (Ware et al. 1990). Based on reported levels, fish consumption advisories were issued for certain species for the protection of human health (Florida Department of Health and Rehabilitative Services and Florida Game and Fresh Water Fish Commission, March 6, 1989, press release). The high levels also prompted several surveys of piscivorous wildlife that revealed elevated Hg levels in wading birds (Spalding and Forrester 1991), raccoons, alligators, and Florida panthers (Roelke et al. 1991). A key early finding was that methylmercury (MeHg) biomagnification was spatially highly variable in the Everglades (Spalding and Forrester 1991; Roelke et al. 1991; USEPA 1998; Lange et al. 1999). For example, based on a comparison of their results with that of Ogden et al. (1974), Spalding and Forrester (1991)

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concluded that great egrets (*Ardea albus*) inhabiting Shark River Slough in ENP had some of the highest Hg levels in South Florida. Similarly, Roelke et al. (1991) concluded Florida panthers (*Felis concolor coryi*) from Shark River Slough had higher Hg levels than cats outside of central Everglades habitats. Based on extensive surveys in 1995 and 1996, the USEPA (1998) identified several hot spots where Hg levels in mosquitofish (*Gambusia holbrooki*) were almost twofold higher than the Everglades basin-wide average. One of the MeHg hot spots was located in northern ENP; another hot spot, which later garnered much more attention from investigators, was located outside of ENP in the center of WCA-3. Concerns regarding possible toxic effects to wildlife from Hg exposures in the Everglades prompted an ecological risk assessment by Jurczyk (1993) and the formation of the Interagency South Florida Mercury Science Program (http://www.floridadep.org/labs/mercury/index.htm; http://sofia.usgs.gov/).

More recent data has shown dramatic declines in Hg levels in Everglades fish (Lange et al. 2000, 2005; Rumbold et al. 2005b), birds (Rumbold et al. 2001a; Frederick et al. 2002), and alligators (*Alligator mississippiensis*) (Rumbold et al. 2002). Lange et al. (2000, 2005), for example, reported that Hg concentrations in age-standardized three-year-old largemouth bass (*Micropterus salmoides*) in WCAs-1, 2, and 3 have declined by approximately 40–80% over the past 10–15 years. Frederick et al. (2002) reported similar declines in feather Hg concentrations in great egrets from several colonies, mostly in WCA-3, and concluded that "the Everglades had undergone a biologically significant decline in Hg availability."

However, because MeHg biomagnification is so highly variable across the fairly large expanse of the Greater Everglades, caution must be exercised when making system-wide generalizations. A number of areas either show no Hg decline or a reversal in the downward trend in recent years (Axelrad et al. 2005). Although there is some indication of Hg decline in biota from northern ENP, the magnitude is uncertain due to differences in analytical techniques employed over time (Rumbold 2005a). In any case, levels presently remain high compared to other areas in south Florida. Sunfish (*Lepomis* spp.) collected from northern ENP in 2004 had an average concentration of 0.44 mg Hg/kg (in whole fish), with concentrations as high as 1.5 mg Hg/kg (Rumbold et al. 2006). Largemouth bass collected at the same time contained an estimated 1.2 mg Hg/kg in age-standardized three-year-old fish (in fillet), with concentrations as high as 2.8 mg Hg/kg measured in older fish (Rumbold et al. 2006). Bass collected over a longer period of record from North Prong Creek in the lower Shark River Slough exhibit an unmistakable reversal in the downward trend. Mercury levels in bass from this site have increased nearly 20% each year between 1998 and 2005, with a standardized Hg concentration now at 1.6 mg Hg/kg and maximal measured concentrations ranging as high as 4.8 mg Hg/kg (Lange et al. 2005). Finally, there is no evidence of any significant Hg decline in fish collected from Florida Bay over the past decade; depending on the species, levels in fillets typically average over 1 mg Hg/kg (Evans et al. 2003, cf. Strom and Graves 2001).

Anomalies to the general downward trend are not limited to ENP. Additional MeHg hot spots have been identified in the northern Greater Everglades region. In particular, an unprecedented water-column MeHg concentration was recently reported at a constructed wetland, Stormwater Treatment Area-2 (STA-2), located north of the WCAs (Rumbold and Fink 2006). Not unexpectedly, that study also reported that Hg levels in biota tracked concentrations in water, eventually reaching anomalous levels in mosquitofish. Nearby in the Holey Land Wildlife Management Area, Hg levels have steadily increased in bass over the past seven years, with an average concentration now at 0.65 mg Hg/kg (Rumbold et al. 2006).

The objective of this paper is to assess the ecological risk of MeHg at the hot spot in northern ENP. Concerns regarding potential risks at this location were first raised in 1999 when an average size bluegill (*L. macrochirus*; 137 mm) captured at this site was found to contain 3.3 mg Hg/kg, almost five times greater than the next highest concentration previously reported for this species in Florida (Rumbold et al. 2001b). In 2004, this site was used as a reference in assessing the significance of risks at STA-2 (Rumbold 2005b). This paper updates that previous assessment based on more recent data and includes additional receptors.

Methods

Site description

Everglades National Park encompasses approximately 6,000 km^2 of freshwater sloughs, sawgrass prairies, mangrove forests, and estuaries extending from U.S. Highway 41 south into Florida Bay (Fig. 1). It was authorized as a national park by the U.S. Congress in 1934 and formally established in 1947. The park's ecological importance was recognized by the international community when it was designated as an International Biosphere Reserve under the Programme on Man and the Biosphere (MAB) of the United Nations Educational, Scientific and Cultural Organization (UNESCO) in 1976, a World Heritage Site by UNESCO in 1979, and a Wetland of International Importance in the Ramsar Convention in 1987 (Maltby and Dugan 1994).

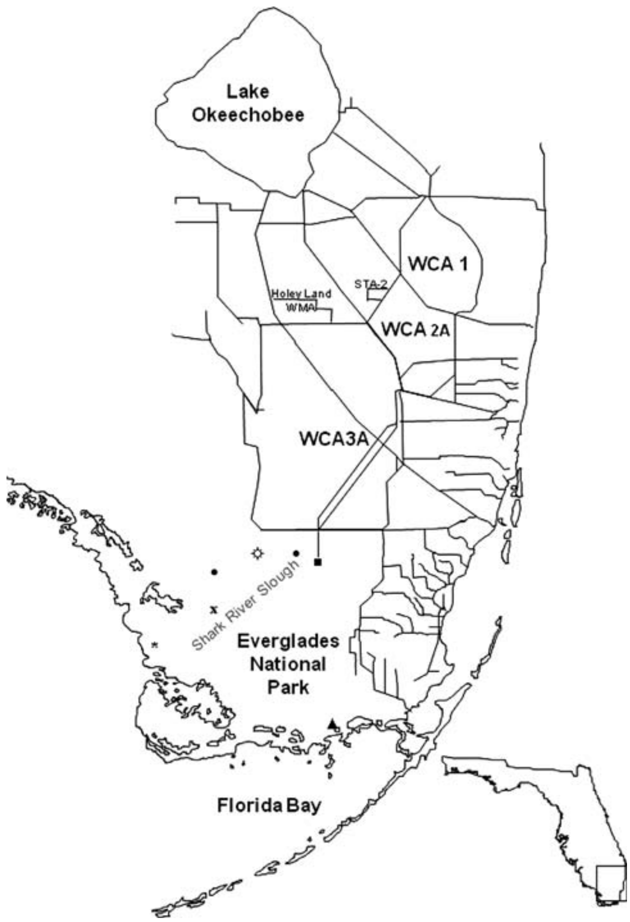

Fig. 1 Map of South Florida showing location of the sampling site (■) within the Greater Everglades region. Also shown are approximate sampling locations of Rumbold et al. (2003 ▲), Lange et al. (2005 *), Ugarte et al. (2005 **x**), Rumbold et al. (2006b ●, i.e., P33 and P34)

Exposure assessment

Three avian receptors known to forage and nest within ENP—the bald eagle (*Haliaeetus leucocephalus*), the wood stork (*Mycteria americana)* and, the great egret (*Ardea albus*)—were used in this assessment. Two of these species have special designation under the Endangered Species Act, as amended (16 U.S.C. 1531, et seq.). The wood stork is designated as endangered and the bald eagle is currently designated as threatened in the lower 48 states (Federal Register Vol. 64, No. 128. 6 July, 1999; for more information, see http://www.fws.gov/endangered).

Life history parameter values and distributions used to estimate Hg exposure to the receptors are summarized in Tables 1 and 2 (for additional life history details, see Rumbold 2005b, 2006a). Potential daily intake of MeHg was estimated using Monte Carlo methods described by Rumbold (2005b, 2006a). Hg levels in prey were derived from the South Florida Water Management District's permit-mandated biomonitoring program. This program required annual sampling of mosquitofish, sunfish, and largemouth bass at various sites in the Greater Everglades. At each location, a grab sample of more than 100 mosquitofish was collected using a dipnet and composited. Also at each location, efforts were made to collect twenty each of sunfish and largemouth bass primarily via electroshocking methods. Whole sunfish were homogenized. Largemouth bass were filleted and a section of muscle removed for analysis. For this assessment, fillet concentration (Cf) was converted to whole fish concentration (Cwf) using the equation $Cwf = 0.695 \times Cf$, derived from Lange et al. (1999). Tissues were frozen pending shipment to FDEPs Chemistry Laboratory (Tallahassee, Florida) for THg analysis following either USEPA Method 245.6 or, if at low level, a modification of Method 1631 (for additional details on sampling design, analytical methods, quality assurance, and geographical and temporal trends in tissue concentrations, see Rumbold et al. 2006). The data used in this assessment represent collections made at the southern terminus of the L-67 extension canal (designated as site L67F1), which is near the headwaters of Shark River Slough (Fig. 1). Although these birds are known to be far-ranging within South Florida, a 100% contact time was used in this assessment. Probability density functions (Table 2) were developed based on empirical distribution functions of Hg levels measured in 60 bass, 60 sunfish, and three composite samples of mosquitofish collected from 2003 to 2005. Empirical relationships employed as extrapolations where a given prey species (e.g., Florida gar, *Lepisosteus platyrhincus*; yellow bullheads, *Ameiurus natalis*; lake chubsucker, *Erimyzon sucetta*; pike killifish, *Belonesox belizanus*) was not collected as part of the biomonitoring program are described in Rumbold (2005b, 2006a).

The trophic transfer model described in Table 1 was previously validated for great egrets by comparing predicted tissue concentrations to measured concentrations in eggs and feathers collected at two egret colonies in WCA-3, which are routinely monitored by the District (see Rumbold 2005a). A lack of locally measured tissue concentrations prevented model validation for the eagle and the stork.

Effects assessment

Biological effects of Hg on birds and other biota were recently summarized by Wiener et al. (2003) and will not be reviewed here. The effects analysis used in this assessment is based on the work of Heinz (1979) in which three generations of mallards (*Anas platyrhynchos*) were dosed with 0.5 mg Hg/kg food and statistically significant effects in both the pre-nesting female and associated

Table 1 Life history parameter values and distributions used to estimate Hg exposure to three receptors

Life history parameter	Value (distribution)	Reference
Bald eagle		
Body weight (BW; kg)	4.5; CV of 4% (normal[a])	Wiemeyer 1991 as cited in USEPA 1993; USEPA 1997; CV based on Bortolotti 1984
Ingestion rate (IR; g/day)	12% of BW	USEPA 1993
Diet (% biomass)	TL 4 fish: bass and warmouth: 0, 18, 100 (triangular)	USEPA 1997; upper range for birds from USEPA 1993; McEwan 1977 as cited in USFWS 1999
	TL 3 fish: other Lepomids: 0, 74, 100 (triangular)	
	Piscivorous birds: 0, 8, 17 (triangular)	
Max. prey size (mm)	No size limit	
Wood stork		
Body weight (BW; kg)	2.4 ± 0.35 (normal[a])	Kahl 1964; Hancock et al. 1992
Ingestion rate (IR; kg/day)	$(10^{0.966\log(BW-0.64)})/1000$	Kushlan 1978; Kahl 1964
Diet (% biomass)	Florida gar: 0, 9.2, 42.9[b] (triangular)	Kahl 1964; Ogden et al. 1976; Kushlan 1979; Depkin et al. 1992; Surdick 1998; Gariboldi et al. 1998
	Warmouth: 21–38.8, (uniform)	
	Other *Lepomis* spp.: 14, 18, 39[b] (triangular)	
	Largemouth bass: 0–3.2, (uniform)	
	Y. bullhead: 8–16.4, (uniform)	
	Small TL 2 and 3 fish: 11–47, (uniform)	
Max. prey size (mm)	≤250	Kahl 1964; Ogden et al. 1976; Surdick 1998; Gariboldi et al. 1998
Great egret		
Body weight (BW; kg)	1.0 ± 0.134 (normal[a])	Palmer 1962; Smith 1995; Henning et al. 1999
Ingestion rate (kg/day)	$(10^{0.966\log(BW-0.64)})/1000$	Kushlan 1978; also see USEPA 1993
Diet (% biomass)	Bass: 0, 3, 13 (triangular)	Frederick et al. 1997; Frederick et al. 1999
	Warmouth: 0, 25, 79 (triangular)	
	Other Lepomids: 0, 19, 35 (triangular)	
	Pike killifish: 0, 5, 15 (triangular)	
	Chubsucker: 0, 3, 9 (triangular)	
	Small TL 1 and 2 fish: 0, 4, 8 (triangular)	
Max. prey size (mm)	≤170	Frederick et al. 1997; Surdick 1998

[a] Normal distributions were truncated at ±2SD to avoid the simulation of implausibly low or high values

[b] Extremes based on diet in Georgia Colonies

hatchlings were observed. Results from Heinz (1979) have been used previously in deriving a lowest-observed-adverse-effect level (LOAEL) of 0.078 mg MeHg/kg bw/day (USEPA 1997; Rumbold 2005b). This LOAEL was then divided by an uncertainty factor (UF) of 3 to derive a chronic no-observed-adverse-effect level (NOAEL) of 0.026 mg MeHg/kg bw/day for use as the acceptable daily MeHg dose (USEPA 1997). Based on guidance in USEPA (1997), an uncertainty factor of 1 was used for this inter-species comparison; however, as discussed by Rumbold (2005b), more recent data indicates that, contrary to earlier assumptions, mallards may not be ultra-sensitive to Hg as compared to wading birds (G. Heinz, USGS, personal communication), and thus this UF may provide little margin of safety.

Results

Risk characterization

Figure 2 shows the results of the Monte Carlo simulations of daily MeHg intake by the three avian receptors foraging in northern ENP. The likelihood that these modeled exposures would be harmful to the receptors was assessed by superimposing the LOAEL and its derived NOAEL

Table 2 Hg levels, distribution, and sample size of prey species used to estimate exposure to avian receptors

Receptor	Prey	Hg (mg Hg/kg)
Bald eagle	TL4 fish: bass and warmouth	0.8 ± 0.4 (normal); $n = 69$
	TL3 fish: other Lepomids	0.32 ± 0.24 (lognormal); $n = 51$
	Piscivorous birds	TL3 × BAF
		BAF = 10.9–13.8 (uniform)
Wood stork	Florida gar	Standard bass x 0.43, 0.74, 0.96 (triangular)[a]
	Warmouth	0.55 ± 0.34 (lognormal); $n = 9$
	Other Lepomids	0.31 ± 0.24 (lognormal); $n = 51$
	Bass	Alpha = 5.6; Beta = 13.1; scale = 1.82(Beta); $n = 16$
	Yellow bullhead	Standard bass × 0.5–0.78 (uniform)[b]
	Small TL 1 & 2 fish[a]	0.023–0.05 (uniform); $n = 3$
Great egret	Bass	0.38 (point est.)[c]; $n = 31$
	Warmouth	0.41 ± 0.17 (lognormal); $n = 6$
	Other Lepomids	0.32 ± 0.26 (lognormal); $n = 44$
	Pike killifish	0.95 × Warmouth[d]
	Lake chubsucker	0.37 × Bluegill[e]
	Small TL 1 & 2 fish	0.023–0.05 (uniform); $n = 3$

Lognormal distributions defined by mean ± 1SD; range 0.0 to + infinity, i.e., no upper truncation

[a] Based on relationship derived from levels reported by Loftus et al. 1998 (ten Florida gar and 24 largemouth bass from ENP); T. Lange (personal communication, a total of 16 Florida gar and 34 largemouth bass from two WCAs sites in 1997-1998)

[b] Based on relationship derived from levels reported by Loftus et al. 1998 (40 yellow bullhead and 24 largemouth bass from ENP); T. Lange (personal communication, five yellow bullhead and eight largemouth bass from a WCA site in 1997)

[c] Based on linear regression using fish <300 mm in length

[d] Based on an empirically derived relationship with warmouth collected at Shark Slough from 1995-1998 ($n = 31$ pike killifish and 20 warmouth; Loftus et al. 1998)

[e] Based data on bluegill ($n = 95$) and lake chubsucker ($n = 97$) collected from six sites from 1995 through 1999 (T. Lange, personal communication; Loftus et al. 1998)

(i.e., the acceptable dose; Fig. 2). The results indicate that an eagle would have a 98% probability of experiencing exposures above the acceptable dose and a 56% probability of experiencing exposures above the LOAEL (Fig. 2a). Similar simulations indicate that a wood stork would have a 100% probability of exceeding the acceptable dose and a 14% probability of exceeding the LOAEL (Fig. 2b). A great egret would have a 99% probability of experiencing exposures above the acceptable dose and a 19% probability of exceeding the LOAEL (Fig. 2c).

Discussion

Based on the assumptions used in this assessment, the results place bald eagles, wood storks, and great egrets foraging in the northern ENP near the L-67 extension canal in a high risk category for adverse effects. These results are consistent with previous assessments that showed risks to be much higher in ENP than in the WCAs (Rumbold 2005b; 2006a). Because there is no consensus of approach among risk assessors, comparisons of modeled risk across studies is difficult. However, based simply on Hg levels observed in fish, the estimated risk from MeHg in ENP appears similar to or greater than other known hot spots across the United States (cf. Brumbaugh et al. 2001; USEPA 2002, 2004; Davis et al. 2003; Kamman et al. 2005), even in areas where impacts to birds are believed to be occurring (Evers et al. 2005; Schwarzbach et al. 2006). Clearly, this risk has added significance given the world-wide recognition of ENP as a unique ecological resource. Ironically, under the Clean Air Act (43 U.S.C. §§ 7470–7492), a national park of this size is designated as a Class 1 area and is expected to be afforded the highest level of air quality protection (http://www2.nature.nps.gov/air/regs/psd.cfm). Yet, atmospheric deposition is the dominant source of bioavailable Hg to the system for subsequent methylation and possible biomagnification.

As argued previously, caution must be exercised concerning system-wide generalizations. In this case, the assessment was based on data from only one site (L67F1) within the 6,000 km^2 park. Nonetheless, results from several other surveys suggest that ENP currently contains the highest Hg levels in South Florida and that elevated levels are not limited to this single location. First, as previously mentioned, fish from the lower Shark River Slough have

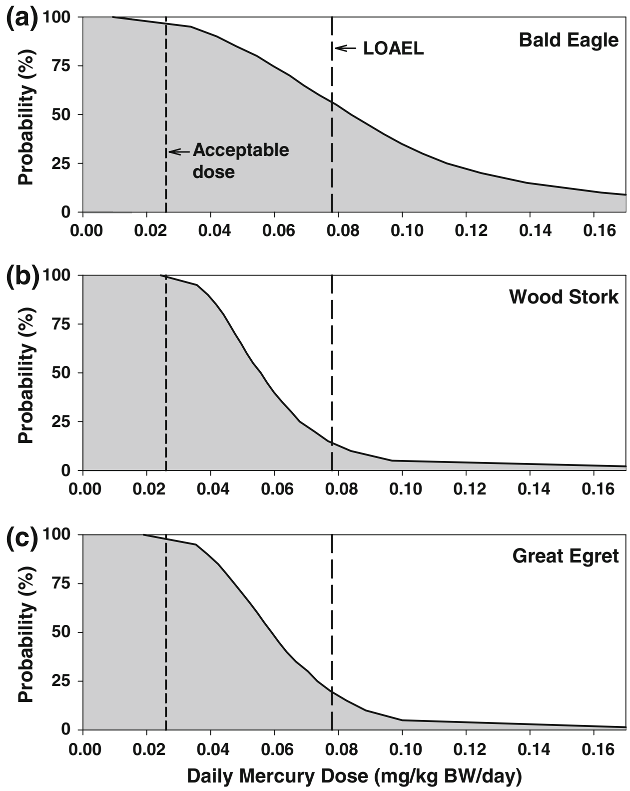

Fig. 2 Comparison of reverse cumulative probability distributions of daily MeHg intake for the bald eagle (**a**), the wood stork (**b**), and the great egret (**c**) to the acceptable dose and LOAEL. The ordinate value (*y*) represents the probability of exceeding the daily Hg intake on the abscissa (*x*)

Hg levels (Lange et al. 2005) similar to those observed at L67F1. Likewise, in 1998 and 1999, fish collected nearby at site P33 contained Hg levels similar to or higher than fish collected contemporaneously from site L67F1 (Rumbold et al. 2001b). Over the past few years, site P33 and another nearby site (P34) have occasionally been found to have elevated water-column MeHg concentrations relative to average concentrations observed in the northern Everglades and STAs (Rumbold 2006b). Furthermore, the geographical patterns first observed by Roelke et al. (1991) in both alligators and raccoons still appear valid with elevated Hg levels in animals collected from a number of sites in northern ENP (Rumbold et al. 2002; Porcella et al. 2004). In addition to water, fish, alligators, and raccoons, pig frogs (*Rana grylio*) collected from central ENP have recently been reported to have elevated Hg levels relative to other Everglades areas, with a maximum concentration of 2.05 mg Hg/kg wet mass in leg tissue (Ugarte et al. 2005). Further to the south within the mangrove ecotone, sediment, water, and fish have also been found to contain elevated MeHg levels (Rumbold et al. 2003). Lastly, it must not be overlooked that ENP, in addition to most of South Florida, remains under fish consumption advisories for the protection of human health (http://www.doh.state.fl.us/environment/community/fishconsumptionadvisories/index.html); of course, this provides little protection for wildlife.

At this time there is no biological effects data to suggest that any of these species have been adversely affected; however, the probabilities of exceeding the LOAEL suggest a high likelihood that dietary MeHg exposure poses a serious threat. In this instance, lack of evidence of biological effects is no cause for optimism (i.e., absence of evidence is not evidence of absence). Previous studies have focused on bioindicators of exposure, not effects. Because this risk presumption for northern ENP cannot be dismissed by available effects data, it is crucial that it be investigated further.

Strategic plan for the collection of additional data

A strategic plan is proposed to collect additional measured and modeled information to support risk-based management decisions in ENP. While the present assessment focuses on piscivorous birds, possible risk is not limited to avian receptors. Other biota may be experiencing adverse biological effects from dietary MeHg exposure in ENP. Given their relevance to resource management goals, potential receptors for further evaluation would include fish (see Wiener et al. 2003), river otter (*Lutra canadensis*), Everglades mink (*Mustela vison*), bobcat (*Lynx rufus*), and Florida panther. It should be noted that a recent assessment of Hg risk to panthers concluded that current risks were low (Barron et al. 2004). However, that study assumed a 70–90% decline in Hg exposures based on the generalization made by Frederick et al. (2002), which has been shown here not to be applicable to ENP. As such, potential risks to the panther should be reassessed based on measured concentrations.

Although much emphasis to date has been on piscivorous wildlife, another key receptor that should not be overlooked is the Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*), an endangered species with a population dependent on critical habitat within ENP. Non-piscivores, including other passerines, have been shown to bioaccumulate high levels of MeHg (for review, see Evers et al. 2005). The Cape Sable seaside sparrow is a dietary generalist that commonly feeds on a variety of soft-bodied insects (USFWS 1999). Although these small, lower trophic prey do not bioaccumulate the same MeHg body burden as fish, Loftus et al. (1998) found these invertebrate taxa, in this same general vicinity, to have relatively high tissue Hg concentrations. Furthermore, because daily exposure and toxicity is based on the ratio of the Hg amount consumed per day to body weight, the sparrow's small size increases its susceptibility. Such body weight influence on dose rate is exemplified in the mercury study report (USEPA 1997), in which estimated daily MeHg intake for kingfishers (*Ceryle alcyon*), and thus their risk, was found to be greater than for eagles. Paradoxically, as the discussion below will explain, water management to protect the critical habitat of the Cape Sable seaside sparrow, a ground-nesting species, within ENP (i.e., to ensure marl prairies are dry during the nesting season; USFWS 2002) may actually boost MeHg production and thus risk.

A first step in assessing risk is to determine exposure. In this case, this might involve measuring Hg levels in specific prey consumed by these receptors. Clearly, this will require standardized analytical techniques or demonstrated equivalency (i.e., USEPA Method 7473—direct mercury analysis versus USEPA Method 245.6 versus USEPA Method 1631). Alternatively, models, such as the Everglades Mercury Cycling Model (E-MCM) (Tetra Tech 2003), could be used to simulate Hg cycling, methylation, and food-web transfers. Otherwise, these steps could be circumvented simply by measuring tissue concentrations within the receptors themselves, using non-invasive techniques in the case of listed species. The results would then be compared to literature-derived critical tissue concentrations. Although this would be an improvement over the risk estimate presented here, what is most urgently needed is information that can unequivocally refute or confirm biological effects. Therefore, bioindicators of effect should be aggressively pursued.

Ideally, to assess possible biological effects, the relationship between measured Hg burdens and reproductive success and demographics should be investigated for each receptor. However, it must be recognized that eco-epidemiological studies have limitations in describing the receptor-specific exposure histories and in establishing causality due to the confounding influence of covarying factors. These limitations are quite problematic, particularly in the case of Hg because it has such a long latency period between exposure and effect. Therefore, multiple lines of evidence from both controlled and observational field studies may be required. Ongoing controlled exposure studies on American kestrels (*Falco sparverius*; see Haebler et al. 2004) and white ibises (*Eudocimus albus*; see Axelrad et al. 2005) may provide valuable toxicity thresholds data in the future. Ultimately this information could then be input into demographic models to predict risk at the population level, which is sorely needed (see Nacci et al. 2005). However, the requisite data collections and model development will require considerable resources and time. In the interim, several suborganismal bioindicators of effect have recently been shown to have promise and should be considered. These include a micronucleus test (Porto et al. 2005), a porphyrin profile assay (Casini et al. 2003), and a neurochemical receptor-binding assay (Basu et al. 2005). Along with the more traditional methods of assessing effects, this approach will provide additional supporting data to guide future management decisions.

Management decision making will also likely require an answer as to why this area remains a hot spot. To begin to answer this question requires that we first examine the factors thought to be responsible for the general Hg decline in Everglades biota. The consensus is that atmospheric loading is the dominant, proximate source of inorganic Hg to the Everglades (for review, see Atkeson and Axelrad 2004). Given this, Atkeson and Axelrad (2004) hypothesized that a decrease in local emissions was the primary driver for the decline in Hg levels in Everglades biota over the past 10–15 years, including the decline observed at the notorious MeHg hot spot within WCA-3 (i.e., site WCA3A-15). Other investigators have challenged this hypothesis, at least in regard to the decline at WCA3A-15, advancing the alternative hypothesis that a concomitant decline in sulfate concentrations observed over time at this site has reduced the Hg methylation rate by sulfate-reducing bacteria (D. Krabbenhoft and W. Orem, U.S. geological survey meeting. Reston, VA, August 17–18, personal communication, 2004; for review, see Axelrad et al. 2005). This argument was supported by model results that suggested a reduction in atmospheric deposition could account for some but not all the reductions observed in biota (Pollman et al. 2004). However, hydroperiod may be a third confounding factor. Based on an observed negative correlation between fish Hg concentrations and maximum depth and drying scores, Snodgrass et al. (2000) hypothesized that a key process in periodically flooded systems involves release of bound Hg from sediments during dry periods and uptake by biota when sediments are reflooded. Krabbenhoft and Fink (2001) also reported increased MeHg biomagnification in Everglades marshes following drydown and reflooding. However, their results suggested that it was not Hg release from sediments but oxidation of sulfide pools in the sediments (e.g., organic sulfide, disulfides, and acid volatile sulfides) during the drydown and subsequent release of sulfate upon rewetting that stimulated sulfate-reducing bacteria and Hg methylation. Based on that work, Rumbold et al. (2002) attributed the decline in Hg levels in alligators in some areas to above-average wet years during the late 1990s and reduction in drydowns that would have reduced the frequency or amplitude of MeHg pulses to the system. In summary, there is no clear consensus on why Hg levels declined in biota in some areas of the Everglades and not in others. Most likely the answer is a combination of factors. Clearly, requisite data collection described above for bioindicators of exposure and effects should be done in combination with continued atmospheric monitoring and biogeochemical studies that coincide with drying and rewetting events.

Conclusions

Risk characterization indicated a high likelihood that MeHg exposures to bald eagles, wood storks, and great egrets foraging in northern ENP would exceed effects thresholds at the time of this assessment. This places these birds in a high risk category, given the assumptions on which the assessments were based. Ample evidence was found to suggest that risk might be elevated in other areas of ENP. Contrary to conclusions of others, the results of this assessment also suggest that elevated Hg levels in the ENP may pose unacceptable risk to other ecological receptors, including the endangered panther. Because this risk presumption cannot be dismissed by available effects data, further investigation is crucial. Interestingly, a similar conclusion and recommendation was made by Ogden et al. (1974). Ultimately, the primary goal is to provide public resource managers with an understanding of mercury risks and the management tools necessary to reduce these risks to acceptable levels (South Florida Mercury Science Program; http://sofia.usgs.gov/).

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