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EPA Docket Center (Air Docket)  
U.S. EPA West (6102T)  
Room B-108  
1200 Pennsylvania Ave., NW  
Washington, DC 20460

Attention Docket ID No. OAR-2002-0056

To Whom It May Concern:

Please find enclosed comments on the U.S. Environmental Protection Agency's, "Proposed National Emissions for Hazardous Air Pollutants; and, in the Alternative, Proposed Standards of Performance for New and Existing Stationary Sources: Electric Utility Steam Generating Units".

This proposed rule was published in the Federal Register at 69 Fed. Reg. 4652.

These comments were coordinated by the Hubbard Brook Research Foundation and are signed by thirty-five mercury scientists from across North America.

Please feel free to contact us if we can provide additional information.

Sincerely,

David Sleeper  
Executive Director



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On December 15, 2003, U.S. Environmental Protection Agency Administrator Michael Leavitt, signed a regulatory package entitled “Proposed National Emission Standards for Hazardous Air Pollutants; and, in the alternative, Proposed Standards of Performance for New and Existing Stationary Sources: Electric Utility Steam Generating Units.” The package was published in the Federal Register at 69 Fed. Reg. 4652 (January 30, 2004).

We, the undersigned members of the scientific community, are pleased that the U.S. Environmental Protection Agency (EPA) is considering controls on mercury emissions from electric utilities. Methyl mercury is a highly toxic substance and electric utilities are the largest source of mercury emissions to the atmosphere in the United States. We urge the EPA to take into consideration the extensive mercury research that has been recently developed when finalizing the proposed rule for mercury emissions from electric utilities. We present some of the relevant findings here.

While this review is not intended to be exhaustive, it provides key examples from the literature and ongoing studies regarding the cycling, effects, and management of mercury in the environment. These comments suggest that special attention should be paid to the timing and levels of mercury emissions reductions, as well as any cap and trade proposal for mercury pollution.

### **Summary**

A. Atmospheric deposition of mercury is an important cause of elevated mercury concentrations in fish and piscivorous birds (such as loons and mergansers). Therefore, we expect that mercury emissions reductions from electric utilities in the U.S. will result in decreased mercury concentrations in aquatic biota.

B. Newly deposited mercury is more bioavailable than existing mercury in many ecosystems. As a result, mercury concentrations in aquatic biota in these ecosystems respond rapidly to decreases in emissions. Therefore, human and wildlife health will benefit significantly from controlling mercury emissions sooner rather than later.

C. Not much is known about the risks associated with a mercury cap and trade program. However, the potential exists to contribute to greater heterogeneity in mercury deposition and to exacerbate biological “hotspots”. Therefore additional research should be conducted to determine the ecosystem response to this potential management option.

D. Our current infrastructure to monitor mercury in the environment needs to be strengthened and expanded in order to track the environmental response to controls on mercury emissions.

## 1. Mercury pollution in the environment is widespread and severe.

Mercury (Hg) levels in the environment are elevated across the United States, even in remote areas (Fitzgerald et al. 1998 and others). Figure 1 shows wet deposition (rain, snow, fog, etc.) of mercury ranging from 4 to 18  $\mu\text{g}/\text{m}^2$ -yr, with higher deposition in the eastern U.S.

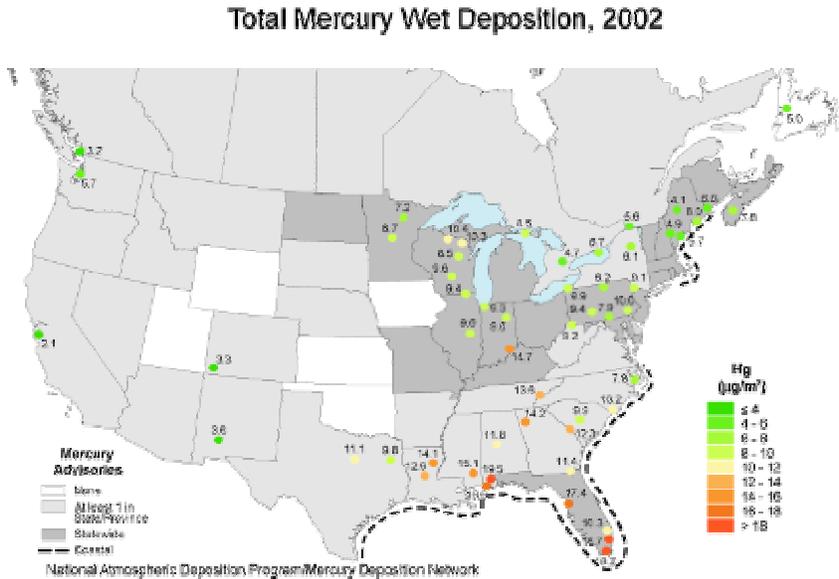


Figure 1: Map of wet deposition of total mercury at Mercury Deposition Network sites. Deposition values are expressed as  $\mu\text{g}/\text{m}^2$ -yr.

As a result of elevated mercury in the environment, fish consumption advisories exist in 45 states encompassing 12 million lake acres and 473,000 river miles -- illustrating that the mercury problem is widespread (see [www.epa.gov/ost/fish](http://www.epa.gov/ost/fish)). Statewide fish consumption advisories for mercury (advisories for 100 percent of surface waters) exist in nineteen states for fresh waters, and in eleven states for coastal waters (EPA 2003). Areas of the northeastern U.S. are particularly hard-hit. Ninety-six percent of the lakes in the Adirondack region of New York (Driscoll et al. 1994) and 40 percent of the lakes in New Hampshire and Vermont (Kamman et al. 2004) exceed the recommended EPA action level for methyl mercury in fish ( $0.3 \mu\text{g}/\text{g}$ ). Of the 767 yellow perch collected in the Adirondack study 66 percent exceed the  $0.3 \mu\text{g}/\text{g}$  action level.

Patterns of mercury contamination in fish and wildlife often mirror atmospheric mercury deposition. For example, a study of mercury in loon eggs shows that mean concentrations are lowest in Alaska and highest in Maine (3.6 times higher than levels in Alaska), indicating a general pattern of increasing egg-mercury concentrations from western to eastern North America (Evers et al. 2003). The same pattern exists in blood-mercury concentrations of adult and juvenile loons (Evers et al. 1998).

## 2. Mercury emissions and corresponding atmospheric deposition are the largest source of mercury in most surface waters in the U.S.

The predominant source of mercury in most watersheds is atmospheric deposition (Lindqvist et al. 1991, Mason et al. 1994, Hudson et al. 1995, Peterson et al. 1995, Driscoll et al. 1998, Fitzgerald et al. 1998). Fitzgerald et al. (1998) systematically rule out alternate hypotheses, such as the role of natural weathering processes as possible mercury sources.

Deposition of mercury has generally increased two to three-fold over the past two centuries following the increase of mercury emissions associated with industrialization, with some locations exhibiting greater than a twenty-fold increase (Meili 2003, Nriagu and Becker 2003, Pirrone et al. 1998). This increased deposition has been recorded in lake sediment cores across North America (see Figure 2).

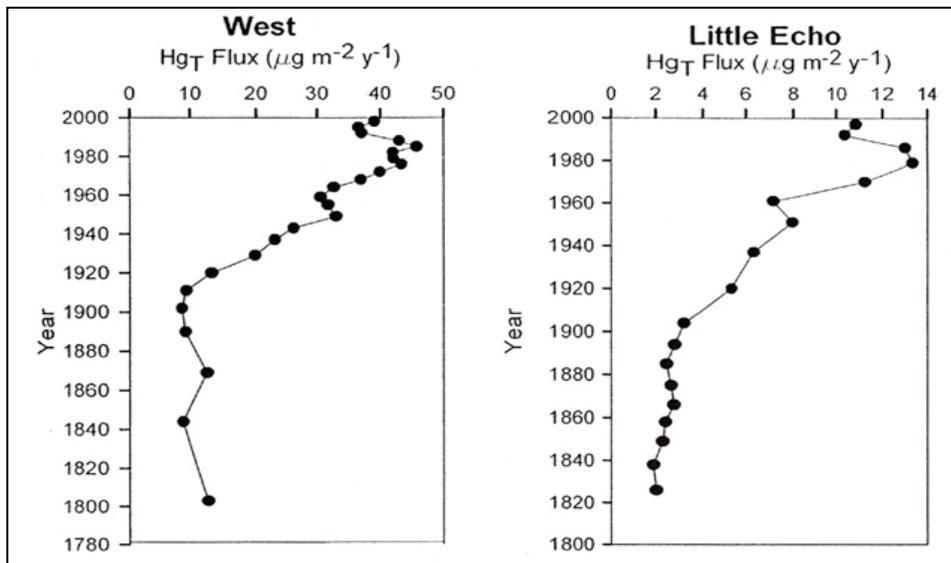


Figure 2: Changes in historical deposition of mercury to sediments in West Pond and Little Echo Pond in the Adirondack region of New York from 1820 to the present (after Lorey and Driscoll 1999). The sediment patterns reflect changes in mercury emissions and deposition over time.

Mercury is emitted to the atmosphere in three basic forms: elemental mercury (Hg(0)), reactive gaseous mercury (RGM) and particulate mercury (Hg(p)). Elemental mercury is capable of transport over long distances in the atmosphere (tens of thousands of kilometers). Reactive gaseous mercury is generally removed within tens to a few hundreds of kilometers from emission sources, and particulate mercury is likely to be deposited at intermediate, distances depending on aerosol diameter/mass (Schroeder and Munthe 1998).

Given its many species, mercury can be a local, regional and global pollutant with short (1-2 days) or long (1-2 years) residence times in the atmosphere. Consequently, it can deposit locally or travel long distances depending on its form (Dastoor and Larocque 2004). The potential for

mercury to act as a global pollutant does not obviate the need for local, regional and continental control efforts in addressing the problem of mercury deposition (Engstrom and Swain 1997).

In the atmosphere, elemental mercury is subjected to many physiochemical processes before deposition to the Earth. Elemental mercury is typically assumed to circulate globally before depositing. Therefore it is often considered less important when evaluating the impact of mercury emissions from domestic electric utilities. However, recent research shows that elemental mercury can be rapidly converted to RGM and deposited locally or regionally. This process occurs after polar sunrises in the Arctic and Antarctic atmosphere (Ebinghaus et al. 2002, Lindberg et al. 2002) and in the marine boundary layer in the presence of marine aerosols.

Recent field studies confirm the presence of elevated RGM in the marine boundary layer (Laurier et al. 2003). Modeling studies corroborate this finding (Hedgecock and Pirrone 2004). Furthermore, such boundary layer oxidation could also be important in the coastal zone, although there is contradictory evidence as to the importance of this process (Mason and Sheu 2002; Laurier and Mason, unpublished data; Malcolm et al. 2003). Observations near Chesapeake Bay, Maryland show strong evidence for RGM formation in the coastal atmosphere (Figure 3).

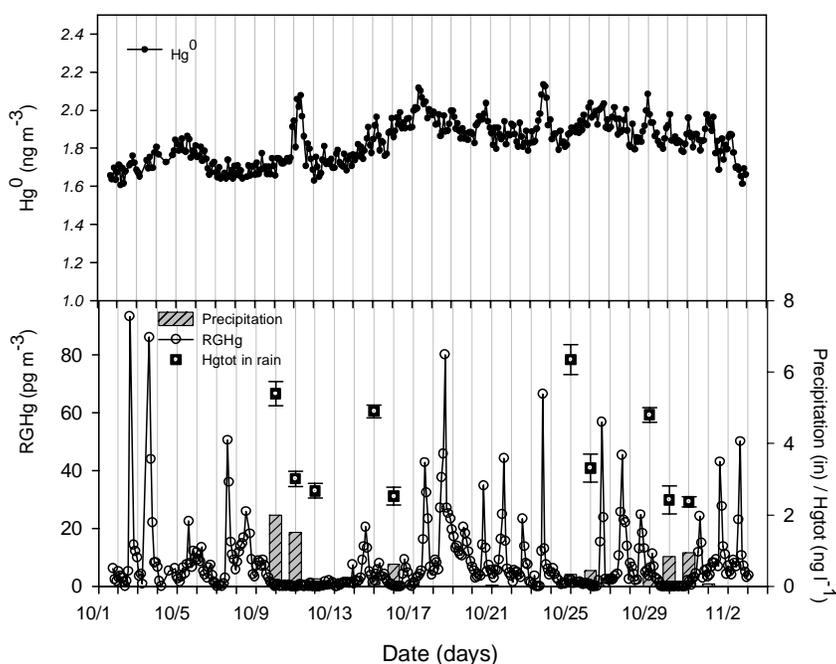


Figure 3: Concentrations of reactive gaseous mercury (RGM) measured at the Chesapeake Bay showing the diurnal cycle of RGM and its subsequent removal from the atmosphere by precipitation. There are no coincident increases in elemental mercury that match the diurnal RGM cycle. From Laurier and Mason (unpublished data).

It is also possible for elemental mercury to be taken up directly by forests through leaf stomata and enter the ecosystem through leaf litter (Lindberg and Johnson 1995; Rea et al. 2002). This provides another mechanism for the transfer of elemental mercury from the atmosphere to

watersheds at a local or regional scale. This process can have important consequences for heavily forested areas such as the northeastern U.S.

Taken together, recent research on the dynamics of elemental mercury, including elemental mercury emitted by domestic electric utilities, suggests that its atmospheric residence time can be significantly decreased in certain environments, and it can therefore contribute to local and regional mercury pollution.

With respect to the oxidized forms of mercury, it is generally assumed that they are not reduced to elemental mercury in the atmosphere. Current information suggests, therefore, that oxidized mercury is not likely to enter the global pool, but rather is more likely to be deposited regionally or locally.

On a global scale, mercury emissions are estimated at 4,840 metric tons per year with 2,860 tons due to current anthropogenic emissions (NESCAUM 2003). The remaining 1,980 tons are released from natural sources or re-released from past anthropogenic emissions (Lamborg et al. 2002). Of the 2,860 tons of anthropogenic mercury emitted to the global pool each year, approximately 1,450 tons (50 percent) are emitted from stationary combustion sources (Pacyna et al. 2003).

The U.S. receives approximately 87 tons of mercury deposition each year (EPA 1997). Identifying the specific sources of this mercury is difficult. Nevertheless, most research indicates that mercury sources in the U.S. are the largest contributor to mercury deposition in the U.S. A recent project funded by the New York State Energy Research and Development Authority (NYSERDA 2002) assessed the contributions of local, regional and global mercury sources to mercury deposition in New York State.

For three study areas in New York (the Adirondacks, Catskills and Finger Lakes), the report shows that mercury emissions within the U.S. are the largest single source of mercury deposition (NYSERDA 2002). For the baseline scenario representing current conditions, the authors report that 37 percent of the mercury reaching the Adirondacks is emitted in the U.S. This relative contribution increases to 52 percent in the Finger Lakes and 60 percent in the Catskills of New York. In all regions, the next largest anthropogenic source area is Asia at 13-19 percent. These results are consistent with the 1997 Mercury Report to Congress which estimated that approximately 60 percent of the mercury deposited in the U.S. originates in the U.S., with the remaining 40 percent coming from the global reservoir (EPA 1997).

In marine systems, mercury budgets demonstrate that mercury emissions and atmospheric wet and dry deposition (primarily as RGM) are the major source of mercury in coastal waters (Rolfhus and Fitzgerald 1995, Fitzgerald et al. 2003) and the open ocean (Mason and Sheu 2002). This pattern is particularly important in light of the fact that consumption of marine fish and seafood is one important way in which humans are exposed to toxic methyl mercury (Fitzgerald et al. 2003).

### 3. Mercury emissions from domestic coal-fired electric utilities are the largest quantified source of atmospheric mercury in the U.S.

Mercury emission trends in the U.S. have changed dramatically over the past decade. In 1990, medical waste incinerators emitted 50 tons per year (25 percent of total U.S. emissions) (EPA 2003). In 1999, they generated only 2.8 tons (2 percent of total U.S. emissions) due to implementation of federal regulations (EPA 2003). Conversely, electric utility emissions have remained largely unchanged and their contribution to total U.S. emissions has increased from 25 percent to 40 percent over the past decade (See Table 1). The next largest category is combined emissions from industrial, commercial, institutional boilers and process heaters at 12 tons per year (EPA 2003).

Table 1: Mercury emissions in the United States by source category.

Source Category	1990 Emissions (tpy) Baseline NTI	1996 Emissions (tpy) 1996 NTI	1999 Emissions (tpy) Version 3.0 1999 NEI for HAPs
Utility coal boilers	51	51	48
Medical waste incinerators	50	40	3
Municipal waste combustors	42	32	5
Industrial/commercial/institutional boilers and process heaters	12	12	12
Chlorine production	10	8	7
Hazardous waste incineration	7	4	7
Portland cement manufacturing	4	6	2
Pulp & paper production	2	4	2
Total (all categories)	195	185	120

Source: EPA National Emission Inventory. July 9, 2003.

While the species of mercury emitted from electric utilities varies depending on the type of coal burned and the pollution control measures in place (NESCAUM 2003, Rossler 2002), Pacyna et al. (2003) estimated the average emission profile from coal combustion in the U.S. (Table 2). These figures are similar to estimates reported by the Electric Power Research Institute (EPRI 2000).

Table 2: Relative distribution of mercury species emitted from electric utilities.

<u>Mercury Species</u>	Pacyna et al. (2003)	EPRI (2000)
Hg(0) (elemental gaseous)	50%	58%
RGM (reactive gaseous mercury)	40%	40%
Hg(p) (particulate mercury)	10%	2%

Using these basic estimates of mercury emissions and speciation, it follows that: of the 87 tons of mercury received in the U.S. each year (1997), 35 tons are likely to originate from global sources (i.e., Hg(0)) and 52 tons from U.S. sources (i.e., RGM and Hg (p) (Rossler 2002, EPA 1997). Of the 52 tons deposited in the U.S. from sources, approximately 24 tons are likely to originate from domestic utility coal boilers (48 tons of emitted mercury x 0.5 fraction likely to be deposited within the U.S.). Based on this calculation, 28 percent of the total mercury deposited in the U.S. is likely to result from coal-fired electric utility emissions in the U.S. This is consistent with earlier EPA estimates of 33 percent (EPA 1997). Stated another way, 46 percent of the mercury deposited within the U.S. that originates within the U.S. (and is therefore subject to U.S. law) is likely to come from electric utility emissions. We consider these numbers to be relatively conservative since some of the elemental mercury emitted from power plants will be converted to oxidized mercury and re-deposited in the U.S. and is not included in this calculation. Moreover, while these calculations represent an “average” speciation ratio for power plants in the U.S., utilities may actually emit 10-90 percent of their mercury as reactive gaseous mercury (Rossler 2002). At the high end of this range, electric utilities emissions would contribute more mercury deposition than the 24 tons estimated here.

While additional research will help improve the precision of these estimates, these calculations provide important insights and identify domestic coal-fired electric utility facilities as a significant source of reactive mercury deposited in the U.S.

#### 4. Mercury has adverse environmental effects.

Once deposited on the Earth, mercury can be converted to methyl mercury through biological processes that occur in all watersheds. Methyl mercury is a potent neurotoxin that bioaccumulates and bioconcentrates through the food chain. Because of its high bioconcentration factor, a small amount of methyl mercury can result in substantial exposure to humans and wildlife. As environmental scientists, our comments focus on the environmental consequences of mercury emissions and deposition. However, the human health effects of mercury are also well-researched and will likely be discussed by others with specific expertise in that area.

Large databases of mercury concentrations in surface waters show that mercury is ubiquitous in the environment. There are data on the concentrations of mercury in lakes across North America under a wide range of conditions. Measured values for mercury in lakes in range from 0.2 to 20 ng/L in remote areas, 80 ng/L in urban areas, and over 100 ng/L near abandoned mines (Driscoll et al. 1994, Chen et al. 2000). Datasets also show that considerable variability exists in Hg concentrations both at the regional and local scale. A survey of lakes in Vermont and New Hampshire by Kamman et al. (2004) documents total mercury concentrations in lake-water ranging from <1 to 34 ng/L across the state of New Hampshire. The statewide variability reflects differences in watershed features as well as deposition patterns.

In aquatic ecosystems, methyl mercury causes neurological, behavioral and reproductive effects, and at extremely high levels may cause direct death in fish and wildlife (Chan et al 2003; Schehammer 1998; Spalding et al. 2000; Bouton et al. 1999; Heinz 1974, 1979; Barr 1986; Burgess et al. 1998; Meyer et al. 1998; Nocera and Taylor 1998). Existing scientific literature on the effects of mercury in some species of fish, birds, and mammals is summarized in Table 3.

Table 3: Summary of wildlife effects of mercury pollution

<b>Organism &amp; Exposure</b>	<b>Symptoms</b>	<b>References</b>
<b>FISH</b>	<i>Reproductive and toxicity effects:</i>	
0.07-0.10 mg/Kg by maternal transfer	Embryo mortality in lake trout eggs	Wiener and Spry 1996
	Adverse effects on growth, development and hormonal status of early life stages	Friedman et al. 1996, Wiener and Spry 1996
0.88 mg/Kg to 8.46 mg/Kg in diet	Spawning success decreased in low. medium and high doses by 50% to 64%	Hammerschmidt et al. 1999, 2003
10-30 mg/Kg	Acute toxicity to fish	Wiener and Spry 1996

<b>BIRDS</b>		
5 mg/Kg in diet	<i>Neurotoxicity effects:</i>	
	Brain lesions	Scheuhammer 1988, Spalding et al. 2000
	Spinal cord degeneration	Scheuhammer 1988
	Central nervous system dysfunction	Scheuhammer 1988
	Tremors	Scheuhammer 1988
	Difficulty flying, walking and standing	Spalding et al. 2000
	Inability to coordinate muscle movement	Spalding et al. 2000
	Reduced feeding, weight loss	Scheuhammer 1988
	Progressive weakness in wings and legs	
0.5 mg/Kg in diet	<i>Behavioral effects:</i>	
	Less likely to hunt, seek shade	Bouton et al. 1999
	Less time flying, walking or pecking	Bouton et al. 1999
	Increased time preening	Bouton et al. 1999
	Exaggerated response to fright stimulus	Heinz 1974, 1979
0.1 mg/Kg in diet	<i>Reproductive effects:</i>	
	Fewer eggs produced	Heinz 1974, 1979
	Lower reproductive success	Heinz 1974, 1979
	Offspring less responsive to maternal calls	Heinz 1974, 1979
	Lower reproductive success in wild loons	Barr 1986, Burgess et al. 1998, Meyer et al. 1998, Nocera and Taylor 1998
0.05 to 5.5 mg/Kg in eggs	Reduced hatchability	Burger and Gochfeld 1997, Fimreite 1971, Gilbertson 1974, Heinz 1979
	Reduced chick survival	Burger and Gochfeld 1997
	Decreased egg volume	Evers et al. 2003, Fimreite 1971
	compromised embryonic development	Fimreite 1971, Gilbertson 1974, Heinz 1979
0.5 to 5.0 mg/Kg	<i>Immunological effects:</i>	
	Lower packed cell volume	Spalding et al. 2000
	Increased lymphocytic cuffing	Spalding et al. 2000
	Greater bone marrow cellularity	Spalding et al. 2000
	Decreased bursal wall thickness	Spalding et al. 2000
	Decreased thymic lobule size	Spalding et al. 2000
	Fewer lymphoid aggregates	Spalding et al. 2000
	Increased perivascular edema in lung	Spalding et al. 2000
<b>MINK &amp; OTTER</b>		
	Anorexia, weight loss	Wren et al. 1987
	Neural necrosis leading to impairment of sensory and motor skills	Wren et al. 1987
	Acute toxicity leading to death	Aulerich et al. 1987, Wren et al. 1987, Dansereau et al. 1999

Adapted from Chan et al. (2003).

The pattern of ecological effects associated with mercury pollution is again consistent with the geographic pattern of mercury deposition in the U.S. For example, using the extensive database on mercury in loon eggs, Evers et al (2003) report that 20 percent of loon eggs in Maine were at high to extremely high risk of failure due to mercury contamination; 25-40 percent of eggs in Michigan, Minnesota and Wisconsin were in the moderate to high category; and 100 percent of loon eggs in Alaska were in the low risk category (Evers et al. 2003). At a more localized scale, Figure 4 shows areas of greater biological impact in New Hampshire that are spatially linked with historically large sources of mercury emissions from southern New Hampshire.

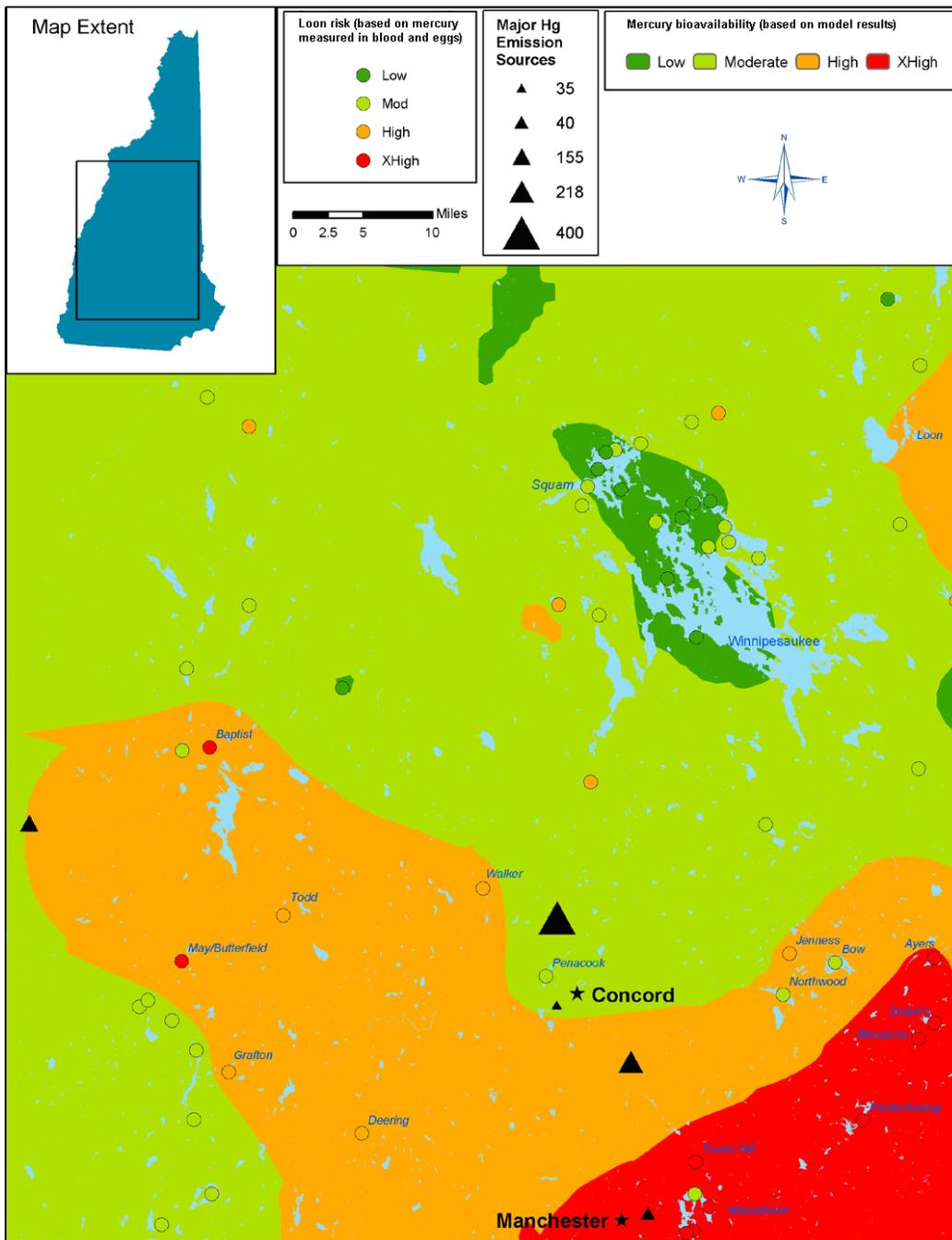


Figure 4: Map showing areas of high mercury bioavailability and the proximity of major mercury sources in southeastern New Hampshire (Evers 2001).

Moreover, it is important to note that some ecosystems are more mercury-sensitive than others because of substantial differences in mercury transport (cf Hurley et al. 1995), net methylation (Benoit et al. 2003), and bioaccumulation rates among ecosystems. Wiener et al. (in press) define mercury-sensitive environments as those that include wetlands, low-alkalinity or low pH lakes, surface waters with nearby wetlands, and dark-water lakes and streams. They note that these ecosystems can experience “significant methyl mercury contamination of fish and wildlife in upper trophic levels” with relatively small inputs of total mercury (<1 to 10 g Hg/ha) (Wiener et al. in press).

The biological impacts of mercury pollution occur throughout entire ecosystems. In addition to the action level for fish, EPA has established wildlife criteria for fish-eating mammals (see Table 4). Under current emissions and deposition regimes, it is expected that it will not be uncommon for freshwater systems to exceed the Mammalian Wildlife Criteria for methyl mercury developed by EPA (Chan et al. 2003). In addition to its aquatic impacts, mercury is also known to accumulate in forest soils and vegetation, but the long-term effects are not yet well understood (Erickson et al. 2003).

Table 4: Wildlife criteria for methyl mercury in water (EPA 1997).

Organism	Wildlife Criterion (ng/L)
Kingfisher	0.033
Loon	0.082
Osprey	0.082
Bald Eagle	0.100
Mink	0.057
Otter	0.042

## **5. Reducing mercury emissions from electric utilities will yield important environmental benefits.**

Results from field studies and model predictions demonstrate that reductions in mercury emissions from sources within the U.S. will lead to decreased mercury deposition and diminished negative effects on the environment.

Lake sediments and peat bogs provide a historical record of mercury deposition over time (Fitzgerald et al. 1998, Lorey and Driscoll 1999). Lake sediments in several regions of the U.S. document a period of mercury enrichment from 1850 through the 1960s and 1970s (Fitzgerald et al. 1998, Engstrom and Swain 1997, Lorey and Driscoll 1999, Kamman and Engstrom 2002). These studies all suggest that over the past 20-30 years mercury deposition has decreased in association with reductions in total U.S. mercury emissions. These patterns demonstrate a strong relationship between emissions of mercury from sources within the U.S. and mercury levels in U.S. lakes. This relationship was confirmed by Engstrom and Swain (1997) by using lake sediments from a range of sites to track the environmental response to changes in emissions. With this method, they showed that decreased mercury deposition in the Upper Midwest is attributable to reductions in emissions from U.S. sources (Engstrom and Swain 1997).

One of the difficulties in establishing strong relationships between changes in mercury emissions, subsequent changes in atmospheric deposition, and the associated response in surface water and fish is the lack of long-term monitoring data for mercury. However, there are two long-term studies that suggest a direct relationship between atmospheric emissions and atmospheric mercury and ecological changes. First, a study from the Florida Everglades shows that reducing emissions of mercury has led to decreased mercury concentrations in fish (Atkeson et al. 2002). Second, two studies in northern Wisconsin also link changes in mercury deposition and ecosystem mercury. Watras et al. (2000) observed that decreases in the mercury concentrations in a seepage lake were related to decreases in atmospheric mercury deposition. In 2002, they reported a rapid decline in mercury in fish as mercury inputs to the system declined (Hrabik and Watras 2002). Based on this and other information, Hrabik and Watras (2002) concluded that even though there is a large reservoir of mercury in lake sediments and in decomposing organisms, newly deposited mercury dominates bioaccumulation processes in their study lakes (Hrabik and Watras 2002).

An experimental study (METAALICUS) is being conducted at the Experimental Lakes Area in Ontario, Canada in which stable mercury isotopes are being added to a whole ecosystem (lake and watershed) to examine the response in fish mercury concentrations. The first two years of results show that mercury added directly to a lake surface is readily converted to methyl mercury and can be observed in fish in the same season the additions start. After 2 years, approximately one quarter of the mercury in young perch was due to mercury added to the lake surface (Blanchfield, unpublished data). Furthermore, the mercury added to the lake surface each year appears to be more available for conversion to methyl mercury (and then move through system to fish) than mercury that has been in the ecosystem for longer periods (Gilmour et al., 2003).

By contrast, the amount of mercury exported from a terrestrial system in runoff does not appear to respond as quickly to changes in atmospheric deposition (Hintelmann et al. 2002). Very little of the mercury applied to the terrestrial ecosystem (upland and wetland) was transported to adjacent surface waters after the first two years of METAALICUS.

These results together suggest that lakes receiving the bulk of their mercury directly from deposition to the lake surface (e.g. some seepage lakes) would see fish mercury concentrations respond more rapidly to changes in atmospheric deposition than lakes receiving most of their mercury from terrestrial runoff. The timing and eventual magnitude of the response remain to be determined, and the project is ongoing.

The results from this important study show that aquatic ecosystems do respond to changes in mercury loading. Further, they confirm field observations from the Florida Everglades, that changes in fish mercury concentrations can be observed in some ecosystems within a relatively short time frame (a couple to a few years) after a change in the mercury deposition rate. For ecosystems that receive a substantial fraction of their mercury load from direct deposition to water and wetland surfaces, rapid reductions in mercury emissions should lead to rapid benefits to human and wildlife health (Atkeson et al. 2003, Blanchfield unpublished data, Gilmour et al. 2001, Gilmour et al. 2003). However, METAALICUS results also show that recovery from mercury pollution could be an extended process in ecosystems with large watershed to waterbody ratios. One examples of such an ecosystem is the Sunday Lake watershed in New York that has been the subject of recent modeling efforts using the Mercury Cycling Model (MCM).

The mercury cycling model (MCM) used to simulate the dynamics of mercury in lakes and watersheds. A version of this model (MCM:HD) has been applied to the Sunday Lake watershed in the Adirondack region of New York. Results from this model suggest that if total mercury loading were reduced by 50 percent methyl mercury concentrations in both the water and yellow perch of Sunday Lake would decrease in response to this reduction (Munson et al., in preparation). The results from this model show that loading reductions would result in improvements in water and fish mercury over the time scale of years to decades for ecosystems that have large watershed to lake ratios such as Sunday Lake (see Figure 5).

An important uncertainty in the MCM:HD model is the response of mercury concentrations in runoff from terrestrial ecosystems in response to changes in atmospheric mercury deposition. Predictions of fish response times for ecosystems receiving most of their mercury from terrestrial runoff strongly depend on assumptions made regarding the watershed response and the bioavailability for methylation of this mercury source. If newly added mercury is more available for methylation, the response times would be expected to shorten compared to current simulations.

While this model should be applied and tested at other sites, initial results suggest that deep cuts in emissions of mercury from electric utilities would have a substantial biological benefit to some aquatic ecosystems, that there is a connection between mercury loading and fish response, and that recovery in some ecosystems will be a long-term process.

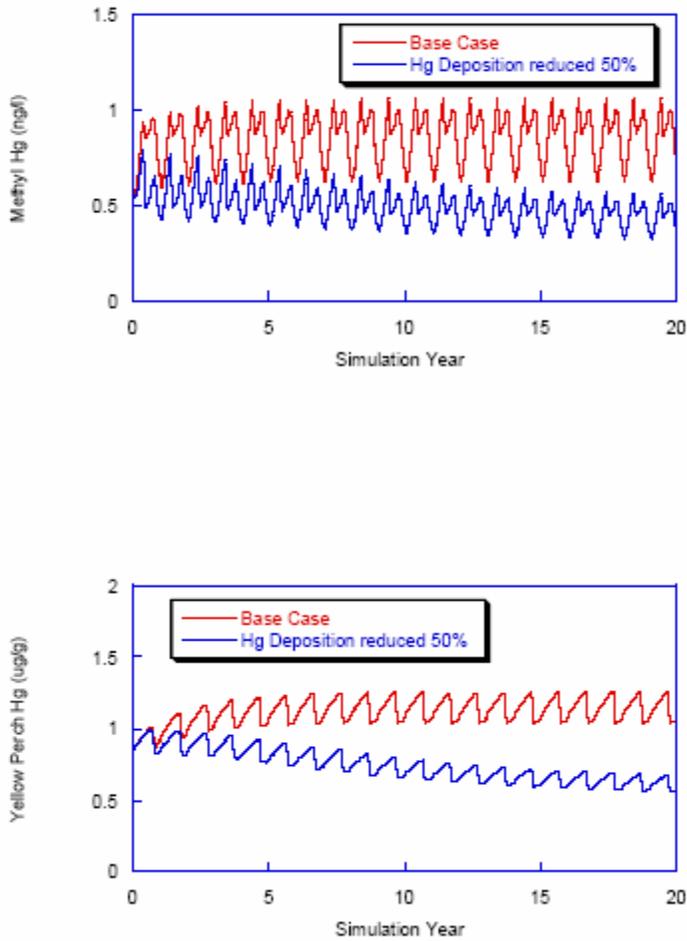


Figure 5: Results from a simulation using MCM:HD showing concentrations of methyl mercury in lake water and yellow perch. Shown is the base case simulation under constant atmospheric mercury deposition and changes resulting from a step decrease of 50 percent in atmospheric mercury deposition (Munson et al., in preparation).

**6. There are ecosystem benefits associated with reducing emissions from electric utilities sooner rather than later.**

We suggest that the EPA revisit the 3 to 15-year implementation timeline in the proposed rule. EPA's proposal cuts utility emissions of mercury from 48 tons/year to 34 tons/year by 2010 and, under one scenario, to 15 tons/year by 2018.

As discussed above, results from mercury-addition studies suggest that recently deposited mercury contributes more to methyl mercury production and bioaccumulation than does existing mercury. The experiments conducted in two very different watersheds in Ontario and the Florida Everglades suggest that mercury that is newly added through pollution is more likely to accumulate in the food chain than previously deposited mercury (Atkeson et al. 2003, Blanchfield unpublished data, Gilmour et al. 2001, Gilmour et al. 2003). Therefore, reducing mercury emissions sooner rather than later would reduce the amount of "new," more bioavailable mercury added to ecosystems.

This finding was confirmed by a study in Wisconsin, which indicated that inputs of "new" mercury strongly control the level of mercury in surface waters and biota. In other words, bioaccumulation of mercury in ecosystems such as the Wisconsin lakes is more dependent on new inputs of mercury from the atmosphere than the recycling of previously deposited mercury. Therefore, it is the newly added mercury from emissions that is of primary importance in aquatic ecosystems and the sooner these inputs are reduced, the greater impact these reductions will have on ecosystem recovery (Hrabik and Watras 2002).

The mercury cycling model results from Sunday Lake in the Adirondacks of New York show that ecosystems with large watersheds may take longer to respond to mercury reductions than "seepage lakes". Therefore, it would be beneficial to initiate emissions reductions as soon as possible in order to facilitate recovery of these ecosystems.

One of the important controls on the amount of mercury that is transported to surface waters and converted to toxic methyl mercury is watershed retention. Data from Lorey and Driscoll (1999) and Kamman and Engstrom (2002) indicate that retention of mercury in lake-watersheds in Vermont, New Hampshire and the Adirondacks of New York (see Figure 6) is decreasing. While the specific mechanisms responsible for this trend have not yet been determined, the data suggest that these watersheds have become more sensitive to mercury deposition. In other words, each year a greater percentage of the mercury is transported to surface waters where it may contribute to adverse environmental effects. This relationship has important consequences for public policy and suggests that greater reductions would be needed in five years (for example) to achieve the same impact these cuts would have today. Therefore, the sooner mercury emissions are reduced, the more effective these reductions will be.

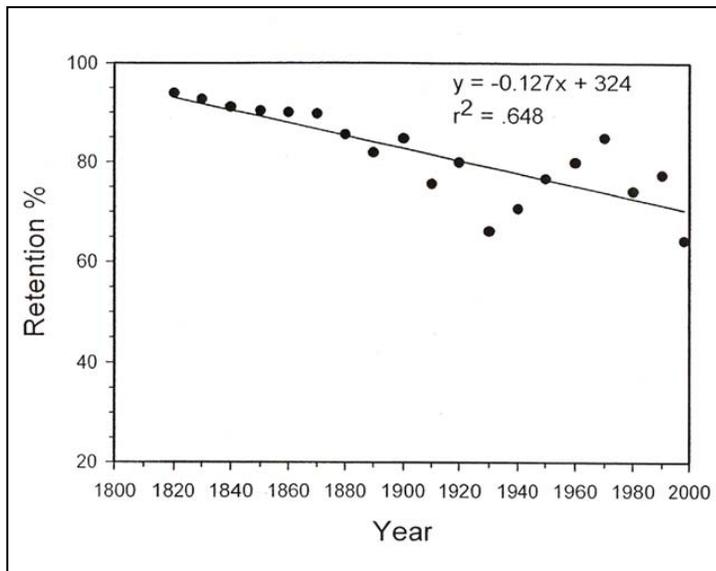


Figure 6: Observations from lake sediment cores in the Adirondack region of New York showing lake-watershed retention of mercury has decreased over the last 200 years (after Lorey and Driscoll 1999).

### 7. Decreases in acid rain should enhance the impact of mercury reductions.

Several research studies suggest a linkage between acidic deposition and mercury levels in fish. This relationship is related to the fact that atmospheric deposition of sulfate associated with sulfur dioxide emissions provides the necessary substrate for methylating bacteria (Gilmour et al. 1992). Many studies across eastern North America have reported increases in fish mercury concentrations with decreases in surface water pH (Grieb et al. 1990, Suns and Hitchin 1990, Driscoll et al. 1994, Kamman et al. 2004).

The role of sulfate in the production of methyl mercury is under investigation by a team of researchers in Minnesota. Swain et al. experimentally added sulfate to a wetland and observed an increase in methylation and export of methyl mercury. They inferred that reducing sulfur dioxide emissions and sulfate deposition would result in decreased methyl mercury in the fish of receiving waters (Swain et al. 2003). Similar experiments have been conducted with similar results in Sweden and Canada (Branfireun et al. 1999, 2001).

Hrabik and Watras (2002) used reference data and data from a lake experimentally acidified with sulfuric acid to examine the relative contribution of atmospheric mercury deposition and acidic deposition in regulating changes in fish mercury concentrations. They observed that decreases in fish mercury in an experimentally de-acidified basin exceeded those in the reference basin. Specifically, they found that one-half of the change in fish mercury over a six-year period could be attributed to de-acidification (Hrabik and Watras 2002). This study suggests that acidification of lakes by acid deposition has enhanced fish mercury concentrations and that concentrations of mercury in fish are likely to decrease with decreasing acid deposition associated with controls on emissions from electric utilities. Nevertheless, acid rain controls are not likely to be sufficient to facilitate full recovery from mercury pollution.

## **8. The trading of mercury has the potential to create or exacerbate mercury “hotspots”.**

Among the options proposed by EPA is a regulatory scheme under which mercury emission allowances could be traded between electric utilities, without geographic limitations, similar to the trading of sulfur dioxide allowances in the acid rain program. If, as many analysts suggest, this regulatory scheme does not include a cap that is low enough to require all electric utilities to reduce their emissions below current levels, and does not address the geographic distribution of allowance trading, then the potential exists to create or exacerbate areas of high mercury deposition. While overall atmospheric mercury deposition will likely decrease with mercury controls, trading could result in variability in the deposition pattern, with some areas showing decreases in deposition and other areas experiencing little or no decrease. As a result, “deposition hotspots” may emerge. Deposition hotspots are areas that receive very high levels of airborne mercury compared to other areas in the U.S.

It is also possible that unconstrained trading with an inadequate cap could result in limited or unchanged deposition in areas with high mercury sensitivity. This could lead to “biological hotspots” that show elevated concentrations of mercury in biota compared to other areas in the U.S. These biological hotspots can be created by elevated deposition, high watershed sensitivity to mercury deposition, or both. Watershed sensitivity is a function of natural and human-induced conditions such as mercury retention capacity, ecosystem methylation rates, wetland occurrences, in-lake trophic structure and lake acidity.

An example of a biological hotspot exists in southern New Hampshire. Data from this region suggest that areas near large sources of mercury emissions are subject to high deposition and elevated methyl mercury concentrations in water, fish and piscivorous birds (for example see Figure 4). While some of this variability can be attributed to differences in watershed characteristics, local sources likely play an important role.

Given the toxicity of mercury, and the paucity of information on the potential effects of mercury trading, it should be approached with great caution. Considerably more research is needed to understand the effects of a mercury trading policy.

## **9. The U.S. mercury monitoring system is not adequate to detect the emergence of hotspots and impact of the proposed rule.**

Despite advances in mercury monitoring and research, the current national monitoring network for airborne mercury in the U.S. is insufficient to measure the full impact of the proposed new regulations. In particular, the network is not designed to detect environmental response to changes in mercury emissions or the emergence of hotspots that might accompany a pollution trading program. As a result, the system is inadequate to ensure government accountability and policy effectiveness.

Currently there are only 63 federal monitoring sites in the U.S. run by the National Atmospheric Deposition Program and only a fraction of these are in or near urban areas (see Figure 7). A proposed monitoring strategy developed by thirty-two mercury scientists from academia, industry, government and non-profit organizations details the need for a comprehensive

investigations at a number of locations (intensive sites), to improve our understanding of the links between atmospheric mercury deposition and methyl mercury in fish and wildlife. In addition, samples should be collected for a subset of indicators at a large number of sites that are widely distributed at a continental scale, and across ecosystems (cluster sites) (Mason et al. in review). The proposed mercury monitoring strategy would create a framework to answer the question of whether change is occurring in atmospheric mercury inputs and how this change is reflected within a variety of aquatic ecosystems and in their organisms. In addition, this strategy would provide the information necessary to demonstrate the impact of mercury emission reductions from anthropogenic sources in the U.S.

We urge the EPA to provide funding to implement key aspects of this strategy before finalizing this rule so that there is a system in place to provide the necessary accountability for this new rule. It is critical that at least the cluster sites discussed in this strategy be identified and established as soon as possible in order gather the necessary background information from which to measure future improvement or degradation.

### National Atmospheric Deposition Program Mercury Deposition Network

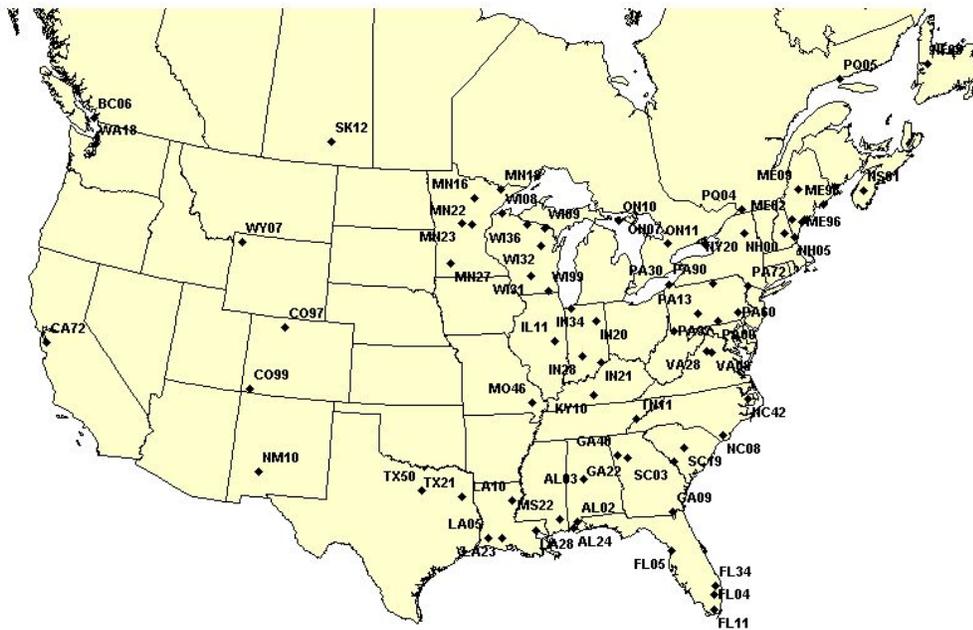


Figure 7: Location of sites that are part of the National Atmospheric Deposition Program Mercury Deposition Network (from <http://nadp.sws.uiuc.edu/mdn/sites.asp>).

Signature Page for comments to the U.S. Environmental Protection Agency regarding the “Proposed National Emission Standards for Hazardous Air Pollutants” and the “Alternative, Proposed Standards of Performance for New and Existing Stationary Sources: Electric Utility Steam Generating Units.”

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