Estimating contribution of precipitation scavenging of atmospheric particulate mercury to mercury wet deposition in Japan

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Abstract

Mercury wet deposition is dependent on both the scavenging of divalent reactive gaseous mercury (RGM) and atmospheric particulate mercury (Hg(p)) by precipitation. Estimating the contribution of precipitation scavenging of RGM and Hg(p) is important for better understanding the causes of the regional and seasonal variations in mercury wet deposition. In this study, the contribution of Hg(p) scavenging was estimated on the basis of the scavenging ratios of other trace elements (i.e., Cd, Cu, Mn, Ni, Pb and V) existing entirely in particulate form. Their wet deposition fluxes and concentrations in air, which were measured concurrently from April 2004 to March 2005 at 10 sites in Japan, were used in this estimation. The monthly wet deposition flux of mercury at each site correlated with the amount of monthly precipitation, whereas the Hg(p) concentrations in air tended to decrease during summer. There was a significant correlation ($P<0.001$) among the calculated monthly average scavenging ratios of trace elements, and the values in each month at each site were similar. Therefore, it is assumed the monthly scavenging ratio of Hg(p) is equivalent to the mean value of other trace elements. Using this scavenging ratio ($W$), the wet deposition flux ($F$) due to Hg(p) scavenging in each month was calculated by $F=WK_P$, where $K$ and $P$ are the Hg(p) concentration and amount of precipitation, respectively. Relatively large fluxes due to Hg(p) scavenging were observed at a highly industrial site and at sites on the Japan Sea coast, which are strongly affected by the local sources and the long-range transport from the Asian continent, respectively. However, on average, at the 10 sites, the contribution of Hg(p) scavenging to the annual mercury deposition flux was 26%, suggesting that mercury wet deposition in Japan is dominated by RGM scavenging. This RGM should originate mainly from the in situ oxidation of Hg$^0$ in the atmosphere.

Keywords: Scavenging ratio; Reactive gaseous mercury; Amount of precipitation; Trace elements; Asian continent

1. Introduction

In recent years, there have been increased human health and environmental concerns associated with the toxicity of methylmercury bioaccumulated in fish through the food chain. Atmospheric deposition has been suggested to be a principal pathway of mercury in aquatic environments, even in regions (e.g., pristine inland lakes in North America and Scandinavia) far from direct sources of emissions.
discharges of mercury (Sorensen et al., 1990; Lindqvist et al., 1991). Mass balance studies of mercury in Lake Superior (Rolfhus et al., 2003) and Lake Michigan (Landis et al., 2002) verified that atmospheric deposition is predominant in the supply of mercury to the lakes. A significant amount of methylmercury is produced by the in situ methylation of inorganic mercury deposited into aquatic environments (Pirrone, 2001; Rolfhus et al., 2003).

Mercury in the atmosphere primarily exists in the gaseous elemental form, Hg$^0$ (generally >95%) (Lindqvist and Rodhe, 1985; Ames et al., 1998), and <5% of the total amount is typically present as divalent reactive gaseous mercury, RGM and particulate mercury, Hg(p). Hg$^0$ is believed to have an atmospheric lifetime of about 1 yr (Lindqvist and Rodhe, 1985; Slemr et al., 1985), while RGM and Hg(p) have much shorter atmospheric lifetimes. Recently, the lifetime of Hg$^0$ in the marine boundary layer was reported to be significantly shorter (around 10 days under typical summer conditions) on the basis of model calculations (Hedgecock and Pirrone, 2004). The RGM and Hg(p) concentrations in air are dependent on direct emissions from local and regional sources and on in situ formation in the atmosphere. For the latter cases, RGM is formed by the oxidation of Hg$^0$ by O$_3$ and OH radicals in the gas and/or aqueous phases (Pai et al., 1997; Schroeder and Munthe, 1998; Shia et al., 1999). Also, Sakata and Marumoto (2002) reported that the formation of Hg(p) in the Tokyo metropolitan area may be governed by the physical adsorption equilibrium of Hg$^0$ between gas and particle phases.

Mercury in precipitation results from the scavenging of Hg(p) and RGM (Guentzel et al., 2001). Hg$^0$ has a very low solubility in water and must first become oxidized in clouds before it is efficiently scavenged by precipitation events. In North America, the wet deposition flux of mercury is currently being monitored at 89 active Mercury Deposition Network (MDN) sites, comprising 80 sites in the United States, 7 sites in Canada and 2 sites in Mexico (National Atmospheric Deposition Program, 2006). In Japan, the wet and dry deposition fluxes of toxic trace elements, including mercury, were monitored at 10 sites by the Central Research Institute of Electric Power Industry (CRIEPI) from December 2002 to March 2006 (Sakata and Marumoto, 2005; Sakata et al., 2006). Estimating the contribution of precipitation scavenging of RGM and Hg(p) is important for better understanding the causes of the regional and seasonal variations in mercury wet deposition. In this study, the contribution of Hg(p) scavenging to mercury wet deposition in Japan was estimated on the basis of the scavenging ratios of other trace elements (i.e., Cd, Cu, Mn, Ni, Pb and V) existing entirely in particulate form. Their wet deposition fluxes and concentrations in air, which were measured concurrently from April 2004 to March 2005 at 10 sites, were used in this estimation.

2. Experimental

2.1. Sampling sites

Ten sampling sites throughout Japan were selected. The locations are shown in Fig. 1. The sampling sites are roughly classified into three sites (sites E, G and H) from urban and industrial areas, six sites (sites B, C, D, F, I and J) from remote areas, and one site (site A) from background areas. Among these 10 sites, three (sites C, F and J) are on the Japan Sea coast. Sites A and B in northern Japan have much lower amounts of annual precipitation than other sites. The amounts of annual precipitation measured in this study were 734 mm at site A and 583 mm at site B, and 1638 mm on average for the 10 sites (see Fig. 2). Sampling was conducted within substations and other facilities of electric power companies and at CRIEPI for site E.

2.2. Sampling procedures

Precipitation samples were collected using an automatic wet-only sampler, which is equipped with two sampling trains that consist of a polycarbonate funnel (177 cm$^2$ collection area), a Teflon tubing (10 mm inner diameter) and a 5 L Teflon sample bottle. The sample bottle for mercury measurement initially contains approximately 50 mL of 5 mol L$^{-1}$ HCl solution for stabilizing mercury without adsorption or volatilization after collection. Prior to each sampling, the funnels, bottles and tubing were carefully cleaned with HCl solution and Milli-Q water. All cleaning and sample-handling processes were conducted using gloves.

In principle, samples were collected simultaneously at all sites every half-month from April 2004 to March 2005. All samples collected at each site were sent to the CRIEPI laboratory (site E) in Komae City, Tokyo. The samples were weighed immediately. Then, BrCl (0.002 mol L$^{-1}$) was added...
to the samples for mercury measurement to oxidize all Hg compounds to Hg$^{2+}$. For the samples for the measurements of other trace elements, HNO$_3$ ($\sim$0.3 mol L$^{-1}$) was added to dissolve the trace elements associated with atmospheric particles and to prevent their adsorption on the walls of the bottles. All samples were stored in a refrigerator until analysis.

To determine concentrations of Hg(p) and other trace elements in air, airborne particles were collected on a quartz fiber filter (8 in. × 10 in.) using a high-volume air sampler (Shibata HV-700F). Sampling was performed at a flow rate of 300 L min$^{-1}$ in the second half ($\sim$2 weeks) of each month from April 2004 to March 2005. Prior to the sampling of particles, the filters were heated overnight at 500°C to remove any Hg present on the filters. After the collection of the samples, the filters were immediately placed in glass containers and sent to the CRIEPI laboratory. They were weighed and stored in a refrigerator until analysis.

There are potential sources of error in sampling by filtration, which may lead to imprecise and inaccurate determination of Hg(p). Landis et al. (2002) found a significant RGM artifact on quartz fiber filter samples collected for Hg(p) determination when KCl-coated annular denuders were not utilized. However, they observed no significant artifact when the RGM concentration was <15 pg m$^{-3}$. The RGM levels observed at site E
Fig. 2. Monthly changes in mercury wet deposition flux and atmospheric particulate mercury concentration and amount of precipitation at 10 sites (April 2004–March 2005). Values given in each figure indicate, in order, annual wet deposition flux of mercury, amount of annual precipitation and annual average concentration of atmospheric particulate mercury.
during May (12 ± 19 pg m⁻³), July (11 ± 14 pg m⁻³), September (6 ± 7 pg m⁻³) and November (3 ± 5 pg m⁻³) 2002 were, in general, less than this value (Sakata and Marumoto, 2004). It is anticipated that there will be no marked difference in the atmospheric RGM concentration throughout Japan, as will be described in the discussion. Our method of measuring Hg(p) should enable the exclusion of a significant positive artifact due to RGM at the sites investigated. Lynam and Keeler (2005) reported the influence of elevated O₃ concentrations in causing an artifact during the sampling of Hg(p). In our study, no significant correlation (at \( P = 0.05 \)) was observed between the average Hg(p) levels and the number of hours at which photochemical oxidant (primarily O₃) levels exceeded the 0.06 ppmv of the environmental quality standards in Japan. Oxidant data are based on hourly concentrations from 5:00 to 20:00 at the pollution-monitoring station nearest to each site in fiscal year 2004 (Ministry of the Environment, Japan, 2006). This suggests a small effect of elevated O₃ concentrations in causing an artifact in Hg(p) measurement.

2.3. Analytical procedures

The precipitation samples were filtered through 0.4 \( \mu \)m filters. Mercury concentration in the filtrates was measured by cold-vapor atomic fluorescence spectrometry (CVAFS; Tekran, model 2600) following Hg⁰ generation with SnCl₂ as the reducing agent (Fitzgerald and Gill, 1979). Other trace elements (Cd, Cu, Mn, Ni, Pb and V) in the precipitation samples were measured directly by inductively coupled plasma mass spectrometry (ICP-MS; Micromass, platform ICP and Shimadzu ICPM-8500). The method detection limits (MDLs) for water samples were approximately as follows: (CVAFS) Hg, 0.1 ng L⁻¹; (ICP-MS) Cd, 0.01 \( \mu \)g L⁻¹; Cu 0.02 \( \mu \)g L⁻¹; Mn, 0.02 \( \mu \)g L⁻¹; Ni, 0.05 \( \mu \)g L⁻¹; Pb, 0.01 \( \mu \)g L⁻¹; V, 0.01 \( \mu \)g L⁻¹. The MDL was defined as three times the standard deviation of replicate measurements of a blank solution. The levels of trace elements in the precipitation samples were generally >5 times higher than the MDLs. The precision and accuracy of the analyses of Cd, Cu, Mn, Ni and Pb were verified using a standard reference material (River Water, JAC0032) from the Japan Society for Analytical Chemistry. The relative standard deviation (RSD) for replicate measurements using a standard solution was generally less than 5\%. The monthly wet deposition fluxes (\( \mu \)g m⁻²) of each element were calculated on the basis of the sum of the amounts (element concentration \times sample weight) in the two samples collected for each month.

For the determination of Hg(p) in air, mercury in two \( 3 \times 3 \) cm² portions of the sample filter was measured by atomic absorption spectrometry (AAS) following thermal desorption and gold trap amalgamation (Nippon Instruments, SP-3D). For the determination of other trace elements (Cd, Cu, Mn, Ni, Pb and V), two \( 3 \times 3 \) cm² portions of the sample filter were digested using HF–HNO₃–HClO₄. After evaporation to dryness, the residue was dissolved in 50 mL of 0.3 mol L⁻¹ HNO₃. The elements in this solution were measured by inductively coupled plasma atomic emission spectrometry (ICP-AES). The concentrations of Cu, Mn and Ni were corrected for the filter blanks. The precision and accuracy of the analyses of elements were verified using a NIST standard reference material (urban particulate matter, SRM 1648). The relative standard deviations of replicate measurements including filter digestion were <10\%.

3. Results

3.1. Monthly changes in mercury wet deposition flux and particulate mercury concentration in air

The monthly changes in mercury wet deposition flux and Hg(p) concentration in air from April 2004 to March 2005 are indicated in Fig. 2. The amounts of monthly precipitation, obtained from the amounts of precipitation collected during wet deposition sampling, are also provided. The monthly wet deposition flux of mercury at each site correlated with the amount of monthly precipitation (sites D, E, G, H, I and J, \( P < 0.001 \); site B, \( P < 0.01 \); sites A, C and F, \( P < 0.05 \)). At sites other than sites C and F on the Japan Sea coast, the mercury wet deposition flux decreased during winter (December–February) when the amount of precipitation decreased. Conversely, relatively high deposition flux was observed during winter at sites C and F. However, the increase in the mercury wet deposition flux during winter was much smaller than that for other trace elements (Cd, Cu, Mn, Ni, Pb and V). As an example, the monthly changes in wet deposition fluxes of those elements at site F are presented in Fig. 3. The increase in their wet deposition fluxes during winter is due to a greater
The contribution of the long-range transport of those elements from the Asian continent (Sakata et al., 2006). Various substances, including sulfates, heavy metals and organic compounds, are transported from the Asian continent to the Japan Sea coast during winter and spring, because the airflows from the Asian continent are dominant (Mukai et al., 1994; Ichikawa et al., 1998; Tanaka and Sakata, 2002).

The annual wet deposition flux of mercury at each site was close to the annual average during the past two years (December 2002–November 2004) (Sakata and Marumoto, 2005; Sakata et al., 2006). Similar to the data for those years, there was a significant correlation between the annual wet deposition flux of mercury and the amount of annual precipitation ($r^2 = 0.81$, \(P<0.001\)), and the regression line passed near the origin, as given

![Graphs showing monthly changes in wet deposition fluxes of Cd, Cu, Mn, Ni, Pb, and V at site F (April 2004–March 2005).](image-url)
The Hg(p) concentrations in air at all sites tended to decrease during summer (Fig. 2). This change was considerably large at site G where the Hg(p) level was the highest. In general, this can be explained by a dilution effect of clean air masses entering from the Pacific Ocean during summer (Tanimoto et al., 2005). However, the ratio of Hg(p) concentration during winter (December–February) to that during summer (June–August) was much higher than that of other elements (Cd, Cu, Mn, Ni, Pb and V). For example, at site G, the order of the ratios observed for trace elements was Hg(p), 6.4 > Cd, 1.5 > Pb, 1.2 > Cu, 1.1 > Ni, 1.1 > Mn, 0.99 > V, 0.50. This implies that the seasonal change in Hg(p) concentration is affected by adsorption or condensation of RGM and Hg⁰ onto atmospheric particles, which is dependent on air temperature (Sakata and Marumoto, 2002). Similar seasonal changes in Hg(p) concentration have been observed in other areas (Iverfeldt, 1991; Keeler et al., 1995).

3.2. Scavenging ratios of trace elements

The concept of the scavenging ratio has been applied to precipitation scavenging of aerosol trace constituents and oxides of sulfur and nitrogen (e.g., Lindberg, 1982; Barrie, 1985). Under the simple premise that the concentrations of constituents in the precipitation (C) depend on their concentrations in the air (K) within which the precipitation is formed, the scavenging ratio (W) is defined as

\[ W = \frac{C}{K}. \] (1)

The monthly average scavenging ratios (mass units) for Cd, Cu, Mn, Ni, Pb and V were calculated using the monthly average concentrations in precipitation (ng L⁻¹) and in air (ng m⁻³) and the density of air (1.2 kg m⁻³). It is assumed that the atmospheric concentrations measured by sampling for about 2 weeks in each month are equivalent to the monthly average concentrations. The calculated annual average scavenging ratio and standard deviation (1σ) for each element were as follows: Cd, 338 ± 395; Cu, 384 ± 422; Mn, 359 ± 429; Ni, 305 ± 404; Pb, 317 ± 359; V, 210 ± 251. There was large variation in the scavenging ratio of each element, which was within the range of values (200–2000) that are indicative of particle scavenging expected for the above elements (Guentzel et al., 1995). However, the relative standard deviation (mean ± 1σ) of scavenging ratios of trace elements in each month at each site was 52 ± 28%. Thus, the scavenging ratios in each month at each site are similar among all trace elements.

The correlation coefficient matrix for the amount of monthly precipitation and monthly average scavenging ratios of trace elements is indicated in Table 1. The monthly average scavenging ratios for trace elements other than V correlated inversely with the amount of monthly precipitation (P<0.05). A similar dependence on the amount of precipitation was observed for daily sulfate scavenging ratios reported by Barrie (1985). A tentative explanation is that the precipitation is more dilute at higher precipitation rates, probably because the amount of monthly precipitation is regarded as a measure of the average precipitation rate in each month (Barrie, 1985). Also, Table 1 shows that there is a significant correlation (P<0.001) among the scavenging ratios of trace elements. This suggests that the difference in the scavenging ratios is independent of any chemical characteristic of each element.

The dependence of the scavenging ratios of trace elements on the respective atmospheric concentrations was investigated. The relationship between the two parameters at urban and industrial, remote and background sites is shown in Fig. 4. The scavenging ratios for all trace elements tended to be approximately constant for higher atmospheric concentrations, but were scattered highly upwards when the atmospheric concentrations were lower. The scavenging ratios from background and remote sites were relatively high. At these sites, trace

<table>
<thead>
<tr>
<th>Precipitation</th>
<th>Cd</th>
<th>Cu</th>
<th>Mn</th>
<th>Ni</th>
<th>Pb</th>
<th>V</th>
</tr>
</thead>
<tbody>
<tr>
<td>Precipitation</td>
<td>1</td>
<td>0.22</td>
<td>1</td>
<td>0.34</td>
<td>1</td>
<td>0.49</td>
</tr>
<tr>
<td>Cd</td>
<td>-0.33</td>
<td>0.49</td>
<td>0.57</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cu</td>
<td>-0.19</td>
<td>0.42</td>
<td>0.46</td>
<td>0.45</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Mn</td>
<td>-0.32</td>
<td>0.68</td>
<td>0.57</td>
<td>0.78</td>
<td>0.44</td>
<td>1</td>
</tr>
<tr>
<td>Ni</td>
<td>-0.32</td>
<td>0.54</td>
<td>0.46</td>
<td>0.69</td>
<td>0.59</td>
<td>0.56</td>
</tr>
<tr>
<td>Pb</td>
<td>-0.14</td>
<td>0.54</td>
<td>0.46</td>
<td>0.69</td>
<td>0.59</td>
<td>0.56</td>
</tr>
<tr>
<td>V</td>
<td>-0.14</td>
<td>0.54</td>
<td>0.46</td>
<td>0.69</td>
<td>0.59</td>
<td>0.56</td>
</tr>
</tbody>
</table>
Fig. 4. Relationship between scavenging ratios (mass units) and atmospheric concentrations for Cd, Cu, Mn, Ni, Pb and V at urban and industrial, remote and background sites.
elements may be primarily associated with larger particles, such as soil particles, because the contribution from anthropogenic sources is small. It is known that larger particles are more efficiently scavenged by wet deposition (e.g., Warneck, 1988; Shimamura et al., 2006). However, whether this is true in the present case remains unknown, since there is no data for particle size distribution of trace elements at background and remote sites. Also, a possible cause of the scattered scavenging ratios is contamination during sample collection and handling and analysis, although this would not be the prime cause. The effect of contamination is expected to be markedly different among the trace elements. If the scattered scavenging ratios were due to contamination, the significant correlation among the scavenging ratios of trace elements, which was mentioned earlier (Table 1), would not be obtained.

4. Discussion

The RGM concentration in air is generally measured using an automatic mercury vapor analyzer with a KCl-coated annular denuder sampling unit (Landis et al., 2002). This makes it difficult to monitor the RGM concentrations at many sites over a long term. In contrast, the measurement of Hg(p) concentration based on the sampling of atmospheric particles by filtration is much easier. Hence, we estimated the contribution of precipitation scavenging of Hg(p) to mercury wet deposition in Japan using the measured Hg(p) concentrations and the assumed scavenging ratios of Hg(p). The principle behind this method is described below. The wet deposition flux of mercury, $F_w$, is given as the sum of the deposition fluxes due to Hg(p) scavenging, $F_{w,p}$, and RGM scavenging, $F_{w,RGM}$:

$$F_w = F_{w,p} + F_{w,RGM} = W_p K_p P + W_{RGM} K_{RGM} P,$$

where $W_p$ and $W_{RGM}$ are scavenging ratios for Hg(p) and RGM, and $K_p$ and $K_{RGM}$ are atmospheric concentrations for Hg(p) and RGM, respectively, and $P$ is the amount of precipitation. As described in the Results section, there was a significant correlation among the scavenging ratios of trace elements (Table 1), and their scavenging ratios in each month at each site were similar, making it possible to assume that the monthly average scavenging ratio of Hg(p) is equivalent to the mean value of other trace elements in each month. Then, the monthly wet deposition flux due to Hg(p) scavenging was calculated using the assumed scavenging ratio and measured atmospheric concentration in each month. The difference between this calculated flux due to Hg(p) scavenging and the measured mercury wet deposition flux corresponds to the wet deposition flux due to RGM scavenging. Particularly during winter and spring at sites C and G, however, the calculated flux due to Hg(p) scavenging often exceeded the measured flux, probably because of the large error. In such cases, the measured flux was assumed to be due entirely to Hg(p) scavenging. In this manner, the contribution of precipitation scavenging of Hg(p) and RGM to the monthly mercury wet deposition flux was estimated at each site. The results are indicated in Fig. 5.

At all sites, the mercury wet deposition during summer was dependent on mainly the precipitation scavenging of RGM. In contrast, the contribution of Hg(p) scavenging increased during spring and winter because of the higher Hg(p) levels (Fig. 2), and probably the lower RGM levels, in air during these seasons. Warm seasons generally favor the formation of RGM due to the higher air temperature, solar radiation intensity, and atmospheric O3 concentration (Lindberg and Stratton, 1998; Lynam and Keeler, 2005). The contribution of Hg(p) scavenging to the annual wet deposition flux of mercury at each site decreased in the order of site C, 46% > site G, 37% > site F, 34% ~ site B, 34% > site A, 21% ~ site J, 21% > site H, 20% > site D, 19% > site E, 15% > site I, 13%. Large fluxes due to Hg(p) scavenging were observed at a highly industrial site (site G) and at sites on the Japan Sea coast (sites C and F). Site G is strongly affected by local sources and sites C and F by the long-range transport from the Asian continent. The average contribution ($\pm 1\sigma$) of Hg(p) scavenging at 10 sites was 26±11%, showing that mercury wet deposition in Japan is dominated by precipitation scavenging of RGM.

There was a significant correlation between the annual wet deposition flux of mercury and the amount of annual precipitation at the 10 sites ($r^2 = 0.81, P<0.001$). This suggests that the contribution of local and regional sources to the mercury wet deposition is small. Hence, the scavenged RGM should mainly originate not from direct emissions from such sources but from in situ oxidation in the atmosphere. This is supported by
Fig. 5. Monthly changes in wet deposition fluxes by scavenging of reactive gaseous mercury and atmospheric particulate mercury at 10 sites (April 2004–March 2005).
the fact that in Japan, major mercury sources, such as municipal solid waste incineration and coal combustion facilities (Kida and Sakai, 2005), are generally equipped with a flue gas control system (e.g., wet scrubber) which effectively removes RGM but not Hg\textsuperscript{0} (Bergström, 1986; Lancia et al., 1993; Galbreath and Zygarlicke, 1996; Nishitani et al., 1996). Measurements of mercury species in flue gas from typical Japanese coal- and oil-fired power plants by the Ontario Hydro Method (ASTM D6784-02) revealed that only 5–17% of the total plants by the Ontario Hydro Method (ASTM from typical Japanese coal- and oil-fired power plants by the Ontario Hydro Method (ASTM D6784-02) revealed that only 5–17% of the total was in the form of RGM (Sakata et al., unpublished results). The annual average (±1\textsigma) concentration of total gaseous mercury (TGM, primarily Hg\textsuperscript{0}) at 211 sites in Japan was 2.2 ± 0.5 ng m\textsuperscript{-3} in fiscal year 2004 (Ministry of the Environment, Japan, 2006). Also, the average photochemical oxidant (primarily O\textsubscript{3}) concentration (±1\textsigma) at the pollution monitoring station nearest to each site was 0.032 ± 0.005 ppmv in the same year (Ministry of the Environment, Japan, 2006). Thus, there was a small difference in TGM and O\textsubscript{3} concentrations among sites. Therefore, we can expect that there is no marked difference in atmospheric RGM concentration or the scavenging ratio of RGM, which is inferred from Eq. (2), throughout the entire country.

A similar correlation between the annual wet deposition flux of mercury and the amount of annual precipitation has been observed at MDN sites in North America. The regression line (\textit{r}\textsuperscript{2} = 0.67, \textit{P}<0.001) based on data collected in 2004 (National Atmospheric Deposition Program, 2006) is given below.

Flux (\mu g m\textsuperscript{-2} yr\textsuperscript{-1})
\begin{equation}
= 0.0092 \times \text{Precipitation (mm)} + 0.2.
\end{equation}

Mercury wet deposition in North America appears to be governed by the same mechanisms as in Japan, i.e., precipitation scavenging of RGM formed by the in situ oxidation of Hg\textsuperscript{0} in the atmosphere. Previously, Guentzel et al. (1995) and Mason et al. (1997) demonstrated that in-cloud oxidation processes, rather than particulate mercury transport and scavenging, contribute to mercury wet deposition in southern Florida and the Chesapeake Bay watershed. Moreover, the model calculations by Seigneur et al. (2004) showed a small contribution from the local and regional sources of RGM and Hg(p) to mercury wet deposition in the United States, suggesting the importance of global background emissions.

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