Litterfall Mercury Dry Deposition in the Eastern USA

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INTRODUCTION

Dry deposition of Hg species

Dry deposition to forests varies among the mercury (Hg) species gaseous oxidized Hg (GOM), particulate-bound Hg (PBM), and gaseous elemental Hg (GEM). GOM can be adsorbed rapidly on many surfaces due to its high solubility and reactivity. Cuticle, stomatal, and soil uptake all contribute to the removal of GOM from the air. Most PBM dry deposition to forest canopies is believed to be sorbed to the leaf surface as cuticle uptake. Stomatal uptake of PBM is probably limited, although possible for very small particles. Notably, GOM and PBM sorbed at leaf surfaces can sometimes traverse the cuticle and reach the epidermis (Cavallini et al., 1999; Stamenkovic and Gustin, 2009), and thus will not be washed away by precipitation. GEM has deposition velocities that are a fraction of those for GOM, but the ambient concentration of GEM is two orders of magnitude higher than GOM plus PBM. This means that the GEM contribution to total dry deposition over forest canopies can be substantial, despite the presence of bi-directional exchange through leaf and soil surfaces (Zhang et al., 2009 and references therein). There also is evidence that non-stomatal uptake of GEM can be as important as stomatal uptake (Stamenkovic and Gustin, 2009, and references therein; Converse et al., 2010). The scientific literature has not yet resolved the relative importance of new or recycled Hg, stomatal or non-stomatal Hg uptake, and proportions of GEM compared to GOM plus PBM in leaves (*see* reviews in Lindberg et al., 2007 and Stamenkovic and Gustin, 2009).

All three species probably contribute to the Hg mass in litterfall through both stomatal and non-stomatal pathways. A reasonable assumption is that PBM contributes least to litterfall due to its low ambient concentrations, capacity to be washed off in throughfall, and limited stomatal uptake. Whether GEM contributes more to litterfall Hg than GOM plus PBM is not fully known (Browne and Fang 1978; Millhollen et al., 2006; Stamenkovic and Gustin, 2009). A portion of the Hg mass in litterfall likely is from recycled Hg—that is, leaf incorporation of Hg reemitted from soil—(Zhang et al., 2009 and experimental evidence cited therein).

A modeling assessment by Zhang et al. (2011) included the northeastern USA, which is dominated by deciduous broadleaf forests, and showed that the annual dry deposition of GOM plus PBM is similar in magnitude to that of GEM—5 to 30 micrograms per square meter per year $[(\mu g/m^2)/yr]$—in most areas. The reemission of GEM from soils and leaves is 10 to 15 $(\mu g/m^2)/yr$, and the natural emission of GEM is less than 2 $(\mu g/m^2)/yr$. (Note that the reemission of GEM is from a portion of the combined dry and wet deposited GOM and PBM that can convert to GEM, not just from recently deposited GEM.) Net Hg deposition is the sum of GOM, PBM, and GEM deposition minus GEM reemission and natural emission. The model results indicated net Hg deposition ranges from 5 to more than 40 $(\mu g/m^2)/yr$.

Previous investigations of Hg in litterfall

Ericksen et al. (2003) showed that almost all of the Hg in leaf tissue of a deciduous species grown in a gas-exchange chamber originated from the atmosphere. Concentrations in the leaves increased as a function of leaf age, leveled off after 2 to 3 months, and were independent of concentrations in the soil. Approximately 80% of the total Hg accumulated in the above-ground biomass was in the leaves and approximately 1% of the Hg in the leaves was MeHg. Rea et al. (2002) reported Hg concentrations in live foliage increased 10-fold from spring bud break to autumn litterfall. Coupled with measurements of atmospheric Hg concentrations, they determined that dry deposition of only 25% of the available ambient GEM could explain all of the Hg in the foliage for the growing season. Hintelmann et al. (2002) sprayed an enriched stable isotope of Hg onto a boreal forest in the Experimental Lakes Area in western Ontario and found the enriched isotopic fraction bound to vegetation was much higher than native Hg, indicating that new atmospheric Hg mostly enters the soil after leaves die, fall, and decompose. Hall and St. Louis (2004) showed decomposing plant litter could gain or lose Hg in flooded and unflooded soils, depending on the original Hg content.

Previous investigations of Hg deposition in litterfall in North America included single- and multi-year studies in small-scale study areas. Early on, Johnson and Lindberg (1995) proposed that Hg in litterfall had an atmospheric

origin and could be a source of Hg deposition, based on data from a deciduous forest research site in Tennessee. Rea et al. (1996) did a short-term study in the Lake Champlain watershed in Vermont that showed the importance of litterfall in Hg deposition. Grigal et al. (2000) collected litterfall in a hardwood-dominant upland forest watershed and a coniferous peatland in northern Minnesota. Litterfall Hg concentrations from the two forest types were different, and the Hg flux in litterfall was approximately equal to the flux from combined throughfall and stemflow. St. Louis et al. (2001) collected litterfall beneath the canopy in a northern boreal forest in the Experimental Lakes Area in western Ontario during fall and spring and estimated fluxes of total Hg and MeHg in litterfall plus throughfall below the forest canopy were 2 to 3 times greater than annual fluxes by direct wet deposition. Sheehan et al. (2006) collected litterfall in two forested watersheds in Acadia National Park in Maine and estimated annual Hg deposition via litterfall was greater than Hg deposition via precipitation and was similar to or greater than the Hg deposition via throughfall. Demers et al. (2007) in a study in the Adirondacks of New York observed that litterfall dominated Hg fluxes to a deciduous forest and throughfall dominated Hg fluxes to the coniferous forest. They concluded the ultimate fate of Hg in the landscape depends on forest type and associated differences in Hg delivery and incorporation. Bushey et al. (2008) studied an upland deciduous forest ecosytem in the Adirondacks of New York and found annual litterfall was the largest Hg input, concluding litterfall Hg represents a largely new, rather than recycled, input to forests.

Hg monitoring networks

For many years, the primary data available to quantify Hg wet deposition have been from collection and analysis of "open-field" precipitation samples at sites in the Mercury Deposition Network (MDN) of the National Atmospheric Deposition Program (NADP). The MDN includes approximately 110 sites in North America (as of 2010) that use standardized procedures, supplies, and equipment. At these sites, weekly composite precipitation samples are collected with an automated sampler, and precipitation amounts are measured with a recording rain gage. Samples are retrieved by a trained operator on a routine schedule and are analyzed for total Hg at a central laboratory. Data on Hg concentration, precipitation amount, and Hg wet deposition are archived in a free, on-line data base maintained by the NADP. These data are used by scientists, regulators, and policymakers as a reliable measure of spatial patterns and temporal trends in Hg wet deposition for much of North America. The MDN provides a framework for litterfall Hg monitoring because it has long-term sites, a broad geographic coverage, capacity for supplementary sample collection, and weekly Hg wet-deposition measurements.

Continuous measurements of atmospheric Hg-species concentrations at multiple sites recently were integrated into a large-scale monitoring network (the Atmospheric Mercury Network (AMNet); National Atmospheric Deposition Program, 2010). Information from the MDN lacks a concurrent quantification of Hg dry deposition, although inferential modeling estimates can be made with data from the approximately 20 sites in the AMNet (as of 2010), most of which share a site with the MDN.

The need for litterfall Hg monitoring

The Madison Declaration (2007) stated "it can be reasonably inferred that emissions from natural surfaces have minimal effect on local atmospheric deposition near major source areas [and that] a significant portion of the Hg deposited onto the forest canopy is derived from the atmospheric pool of GEM." The Madison Declaration also noted that in the past decade, several studies in forest settings in Europe and North America have shown that Hg fluxes in litterfall plus throughfall (as a surrogate for total wet plus dry deposition) to the forest floor range from about 2- to 7-fold greater than Hg fluxes in wet deposition.

The need for litterfall Hg monitoring was supported by statements from two groups of Hg scientists. Lindberg et al. (2007), reflecting on progress and uncertainties in attributing the sources of Hg in deposition, observed that there are no long-term measurements of both wet and dry deposition for direct comparison. They stated that if litterfall and net throughfall reflect the net dry deposition of Hg, estimates of dry deposition are equal to or greater than open-field wet deposition. Furthermore, there is a need to develop accepted methods to measure dry deposition and to carry out long-term intercomparison studies with all methods at sites with ongoing wet-deposition measurements such as the MDN. Mason et al. (2005) outlined a mercury-monitoring strategy with a continental-scale network of long-term monitoring locations across different ecosystems. They noted the importance of dry deposition, stating that methods for the measurement of Hg in litterfall and throughfall need to be standardized and calibrated for the network.

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METHODS

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Litterfall Hg site selection

Site selection for our study was a multi-step process that used land cover, land use, and site-specific information to find MDN sites with nearby forest for a study plot. First, the land-cover types for the MDN sites active in 2007 were identified with the National Land Cover Database (NLCD) (Multi-Resolution Land Characteristics Consortium, 2001). The NLCD has 25 land-cover classes at a 30-meter (m) resolution for the USA. Of the 96 MDN sites in the USA, the locations of 32 sites were in areas the NLCD classified as forest, shrubland, or woody wetland. The land cover for the 30-m area did not always characterize the land cover in a smaller area around the MDN site. Second, site-specific information was used to indicate other MDN sites that were appropriate for litterfall monitoring. These other sites were in Federal or state parks, forests, and wildlife refuges, along with sites in the AMNet. Third, candidate sites were evaluated using the NADP's MDN site-survey data and photographs to determine if tracts of forest land were in view from the MDN site. Fourth, forest suitability for the study was verified with MDN site operators.

Litterfall collector placement

Our study plots were 16 by 16 m. To randomly place the collectors in a study plot, we composed a grid of 16 cells and assigned each cell a label by a random-number generator. Collectors were placed in the cells with the four lowest numbers and alternate cells were identified so that collectors could be located to avoid obstructions. The operators were provided a kit they used to delineate the study plot and to locate cells for the collectors. The same study plot was used each year, but new cells for the collectors were assigned each year.

Forest class and forest-cover type of study plots

The forest class of the study plots at the 23 sites in this study included 16 deciduous, 5 mixed, and 2 coniferous. These classifications were based on prevalent tree species reported by the site operator and those observed in the litterfall samples during processing. The forest-cover type for the area of the study plot was identified with a dataset (U.S. Geological Survey, 2000) based on a U.S. Forest Service inventory (Zhu and Evans, 1994) with a 1-kilometer (km) grid (Supplementary Data Fig. F1). The 23 sites in this study were in areas of 6 forest-cover types—oak-hickory, maple-beech-birch, aspen-birch, oak-gum-cypress, white-red-jack pine, and shortleaf pine. The forest class based on the species in the forest plot of 7 sites differed from the forest class based on the forest-cover type for the area. The number of study sites per forest-cover type can be compared to the area of these forest types in the eastern USA. Of the 6 forest-cover types, according to the forest inventory dataset, the largest area was oak-hickory (502,602 square kilometers, km^2), which had 9 sites; followed by maple-beech-birch (207,015 km^2) with 5 sites; oak-gum-cypress (120,347 km^2) with 2 sites; and aspen-birch (106,724 km^2) with 5 sites.

Assumptions

We made the following assumptions to justify the methods for site selection, sampling and analysis, and estimates of annual litterfall Hg dry deposition. Hg in a deciduous forest canopy reaches its annual maximum in autumn. The leaves in autumn litterfall constitute most annual Hg dry deposition. A square meter of 4 randomly placed passive collectors provides a representative sample of the autumn litterfall from a small study plot. Passive collectors that separate the litterfall samples from the forest floor inhibit substantial net Hg loss or gain from precipitation, decomposition, or soil contact.

Statistical methods

Statistical methods used to compare litterfall Hg and Hg in precipitation data from different years, sites, and groups were nonparametric tests completed on the data ranks rather than the data values to minimize the effect of outliers. A significance level of $\alpha = 0.05$ was used for the statistical tests, and a p-value less than 0.05 indicated a significant difference. The Wilcoxon rank-sum test (WRS) was used to evaluate whether the distributions of the data from two groups were different. The Kruskal-Wallis rank-sum test (KWRS) was used to evaluate whether the distributions of the data from more than two groups were different. The Tukey multiple comparison of medians of 95% confidence intervals (Tukey) was used to determine which groups were significantly different, regardless of the size of the group. Strengths of statistical correlations were evaluated with the Spearman rank correlation coefficient (*rho),* where near zero is weak and near one is strong correlation.

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SUPPLEMENTARY DATA TABLES AND FIGURE

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Table T1. Characteristics of Mercury Deposition Network sites in litterfall Hg study, 2007-2009.

MDN site number	Abbreviated site name	Latitude (decimal degrees)	Longitude (decimal degrees)	Elevation (meters)	Location	Location description	Co-located monitoring[a] at MDN site
GA09	Okefenokee	30.74	-82.13	47	Charlton County, GA	National Wildlife Refuge	NTN
IN20	Roush Lake	40.84	-85.46	244	Huntington County, IN	IN state recreation area	NTN
IN21	Clifty Falls	38.76	-85.42	256	Jefferson County, IN	State park	none
IN26	Fort Harrison	39.86	-86.02	260	Marion County, IN	State park	none
IN34	Indiana Dunes	41.63	-87.09	208	Porter County, IN	National Lakeshore	NTN
KY10	Mammoth Cave	37.13	-86.15	236	Edmonson County, KY	National Park	NTN
MD08	Piney Reservoir	39.71	-79.01	769	Garrett County, MD	University research site	NTN, AMNet
MD99	Beltsville	39.03	-76.82	46	Prince Georges County, MD	University research site	NTN, AMNet
MI48	Seney	46.29	-85.95	216	Schoolcraft County, MI	National Wildlife Refuge	NTN
MN16	Marcell	47.53	-93.47	431	Itasca County, MN	Forest research station	NTN
MN98	Blaine	45.14	-93.22	275	Anoka County, MN	Urban monitoring site	none
NY68	Biscuit Brook	41.99	-74.50	634	Ulster County, NY	Watershed research site	NTN, CASTNET
OH02	Athens Allegheny	39.31	-82.12	275	Athens County, OH	University research site	AMNet
PA13	Portage	40.46	-78.56	739	Cambria County, PA	National Historic Site	none
SC05	Cape Romaine	32.94	-79.66	3	Charleston County, SC	National Wildlife Refuge	NTN
TN11	Smoky Mountains	35.66	-83.59	640	Sevier County, TN	National Park	NTN, CASTNET
VA28	Shenandoah	38.52	-78.44	1,074	Madison County, VA	National Park	NTN, CASTNET
VT99	Underhill	44.53	-72.87	399	Chittenden County, VT	Forest research site	NTN, AMNet, AirMon
WI09	Popple River	45.80	-88.40	421	Florence County, WI	Forest reseach site	NTN
WI31	Devil's Lake	43.44	-89.68	389	Sauk County, WI	State park	none
WI36	Trout Lake	46.05	-89.65	501	Vilas County, WI	Watershed research site	NTN
WI99	Lake Geneva	42.58	-88.50	288	Walworth County, WI	Park and recreation area	NTN
WV99	Canaan Valley	39.06	-79.42	988	Tucker County, WV	State park	AMNet

[a]MDN, Mercury Deposition Network; NTN, National Trends Network; AMNet, Atmospheric Mercury Network; CASTNET, Clean Air Status and Trends Network; AirMon, Atmospheric Integrated Research Monitoring Network

Table T2. Tree species, forest-cover class, and forest-cover type for Mercury Deposition Network sites in litterfall Hg study, 2007-2009.

MDN site number	Prevalent tree species in study plot	Forest-cover class of study plot	Forest-cover type[a] for area of study plot	Study plot and area class	National Land Cover Database[b] class for study plot
GA09	pine, oak	mixed	oak-gum-cypress	similar	transitional
IN20	maple, poplar, ash, cherry	deciduous	maple-beech-birch	similar	row crops
IN21	maple, poplar, ash	deciduous	oak-hickory	similar	forest and residential
IN26	oak, maple, poplar, cherry	deciduous	oak-hickory	similar	urban
IN34	oak, maple, hickory	deciduous	oak-hickory	similar	pasture
KY10	oak, maple, hickory, ash	deciduous	oak-hickory	similar	pasture
MD08	oak, maple, cherry	deciduous	maple-beech-birch	similar	deciduous forest
MD99	oak, maple, beech, sweetgum	deciduous	loblolly-short leaf pine[c]	not similar	urban
MI48	red pine, jack pine, balsam fir, birch	coniferous	aspen-birch	not similar	woody wetland
MN16	aspen, maple	deciduous	aspen-birch	similar	deciduous forest
MN98	aspen, maple	deciduous	aspen-birch	similar	not determined
NY68	beech, maple	deciduous	oak-hickory	similar	deciduous forest
OH02	oak, maple, hickory, cherry	deciduous	oak-hickory	similar	pasture
PA13	oak, maple, beech, cherry	deciduous	oak-hickory	similar	commercial industrial
SC05	oak, hickory, sweetgum, pine	mixed	oak-gum-cypress	similar	residential
TN11	maple, poplar, birch, pine, hemlock	deciduous	oak-hickory	similar	mixed forest
VA28	birch, locust, oak, pine	mixed	white-red-jack pine	not similar	deciduous forest
VT99	birch, maple, beech, hemlock, spruce	mixed	maple-beech-birch	not similar	deciduous forest
WI09	birch, maple, aspen, fir	mixed	maple-beech-birch	not similar	herbaceous wetland
WI31	cottonwood, maple, oak, elm	deciduous	aspen-birch	similar	pasture
WI36	pine, fir, spruce	coniferous	aspen-birch	not similar	evergreen forest
WI99	oak, basswood, ash, elm	deciduous	oak-hickory	similar	deciduous forest
WV99	birch, maple, beech, hemlock, spruce	mixed	maple-beech-birch	not similar	pasture

[a] From the Forest Cover Types Data Set (U.S. Geological Survey, 2000), based on Zhu and Evans (1994).

[b] Multi-Resolution Land Characteristics Consortium (2001).

[c] Forest-cover type identified as urban deciduous for this analysis.

Table T3. Annual litterfall Hg concentrations, total sample catch, litterfall Hg deposition, and Hg wet deposition at Mercury Deposition Network sites, 2007-2009.

[MDN, Mercury Deposition Network; ng/g, nanogram per gram; g, gram dry weight; µg/m^2, microgram per square meter; n.a., not applicable]

MDN site number	2007				2008				2009			
	Litterfall Hg concentration (ng/g)[a]	Total sample catch (g)[b]	Litterfall Hg deposition (µg/m^2)[c]	Hg wet deposition (µg/m^2)[d]	Litterfall Hg concentration (ng/g)[a]	Total sample catch (g)[b]	Litterfall Hg deposition (µg/m^2)[c]	Hg wet deposition (µg/m^2)[d]	Litterfall Hg concentration (ng/g)[a]	Total sample catch (g)[b]	Litterfall Hg deposition (µg/m^2)[c]	Hg wet deposition (µg/m^2)[d]
GA09	no data	no data	no data	n.a.	34.0 ± 2.3	157.2	5.3	12.5	34.9 ± 3.5	*157.2*	*5.5*	13.3
IN20	33.4 ± 19.6[e]	588.2[f]	19.7	7.6	45.1 ± 3.6	273.5	12.3	9.5	41.3 ± 1.5	236.1	9.7	7.8
IN21	34.0 ± 22.3[e]	475.2[f]	16.1	12.0	53.4 ± 2.8	321.7	17.2	16.3	58.8 ± 1.0	251.6	14.8	16.5
IN26	48.2 ± 12.1[e]	444.8[f]	21.5	9.8	47.8 ± 0.6	408.6	19.5	19.7	42.8 ± 1.9	259.6	11.1	9.6
IN34	58.3 ± 11.0[e]	401.9[f]	23.4	10.6	45.4 ± 2.2	446.1	20.3	12.1	33.7 ± 3.3	367.8	12.4	10.0
KY10	36.1 ± 8.9[e]	348.6[f]	12.6	10.2	39.9 ± 1.9	318.4	12.7	10.2	45.2 ± 4.1	276.8	12.5	12.1
MD08	62.7 ± 1.6	*279.7*	*17.5*	8.3	53.7 ± 3.9	279.7	15.0	10.6	48.0 ± 2.8	*279.7*	*13.4*	6.1
MD99	no data	no data	n.a.	n.a.	49.8 ± 0.9	325.6	16.2	9.8	45.3 ± 4.1	326.7	14.8	10.8
MI48	no data	no data	n.a.	n.a.	39.0 ± 2.7	181.6	7.1	6.1	42.6 ± 4.6	*181.6*	*7.7*	5.2
MN16	no data	no data	n.a.	n.a.	21.4 ± 1.2	163.2	3.5	8.8	25.3 ± 1.3	*163.2*	*4.1*	4.5
MN98	no data	no data	n.a.	n.a.	38.5 ± 2.6	196.4	7.6	7.2	35.8 ± 1.1	222.6	8.0	5.9
NY68	no data	no data	n.a.	n.a.	43.9 ± 1.1	364.3	16.0	10.6	49.2 ± 1.8	*296.0*	*14.6*	8.5
OH02	50.1 ± 9.8[e]	431.4[f]	21.6	8.7	37.6 ± 3.8	499.2	18.8	9.8	49.0 ± 1.2	329.0	16.1	6.8
PA13	no data	no data	n.a.	n.a.	46.0 ± 2.3	356.3	16.4	9.7	41.1 ± 2.1	261.2	10.7	7.1
SC05	no data	no data	n.a.	n.a.	50.3 ± 5.0	181.9	9.1	9.0	51.9 ± 2.7	*181.9*	*9.4*	7.4
TN11	32.6 ± 12.4[e]	367.7[f]	12.0	9.9	43.9 ± 4.6	463.0	20.3	11.3	44.9 ± 2.9	273.0	12.3	11.6
VA28	no data	no data	n.a.	n.a.	38.6 ± 2.9	214.7	8.3	12.4	33.4 ± 3.5	*214.7*	*7.2*	6.8
VT99	no data	no data	n.a.	n.a.	38.6 ± 1.2	324.7	12.5	9.2	34.5 ± 2.2	288.8	10.0	6.3
WI09	39.6 ± 10.2	*272.0*	*10.8*	8.9	29.0 ± 1.6	317.4	9.2	5.3	no data	no data	n.a.	n.a.
WI31	40.5 ± 7.9	*129.3*	*5.2*	9.6	39.1 ± 5.1	129.3	5.1	12.9	26.8 ± 7.7	*129.3*	*3.5*	6.0
WI36	31.4 ± 21.2[e]	294.3[f]	9.2	6.3	38.6 ± 3.8	180.4	7.0	6.4	36.9 ± 6.8	*180.4*	*6.6*	4.4
WI99	51.0 ± 15.9[e]	293.0[f]	14.9	8.8	53.5 ± 3.3	284.6	15.2	11.6	44.9 ± 5.5	*284.6*	*12.8*	10.4
WV99	no data	no data	n.a.	n.a.	40.1 ± 2.6	250.9	10.1	11.6	38.7 ± 1.7	*250.9*	*9.7*	7.7

[a]Annual mean Hg concentration in litterfall samples from 4 collectors ± standard deviation of mean.
[b]Annual sum of the sample catch from 4 collectors; adjusted total sample catch in *italics*.
[c]Annual litterfall Hg deposition computed as the product of annual mean Hg concentration and total sample catch; deposition computed with adjusted total sample catch in *italics*.
[d]Annual Hg wet deposition (from National Atmospheric Deposition Program, 2008, 2009, 2010).
[e]Annual mean Hg concentration in litterfall samples from 8 collectors ± standard deviation of mean.
[f]Annual sum of the sample catch from 8 collectors.

Table T4. Methylmercury concentrations in composite litterfall samples at Mercury Deposition Network sites, 2007.

[MeHg, methylmercury; ng/g, nanogram per gram]

MDN site number	First sample MeHg (ng/g)	Second sample MeHg (ng/g)	Mean MeHg (ng/g)	First sample Hg (ng/g)	Second sample Hg (ng/g)	Mean Hg (ng/g)	Percentage MeHg/Hg[a]
IN20	0.31	0.70	0.50	49.9	57.7	53.8	0.9
IN21	0.26	0.47	0.36	50.1	60.1	55.1	0.7
IN26	0.49	0.36	0.42	46.1	42.0	44.0	1.0
IN34	0.52	0.40	0.46	59.9	49.9	54.9	0.8
KY10	0.25	0.31	0.28	40.9	38.2	39.5	0.7
MD08	0.33	ND	0.33	58.5	ND	58.5	0.6
OH02	0.56	0.35	0.46	51.5	49.0	50.3	0.9
TN11	0.25	0.24	0.25	40.9	30.1	35.5	0.7
WI09	0.29	ND	0.29	37.8	ND	37.8	0.8
WI31	0.50	ND	0.50	32.8	ND	32.8	1.5
WI36	0.52	0.31	0.42	53.1	39.6	46.3	0.9
WI99	0.33	0.38	0.35	51.8	38.0	44.9	0.8

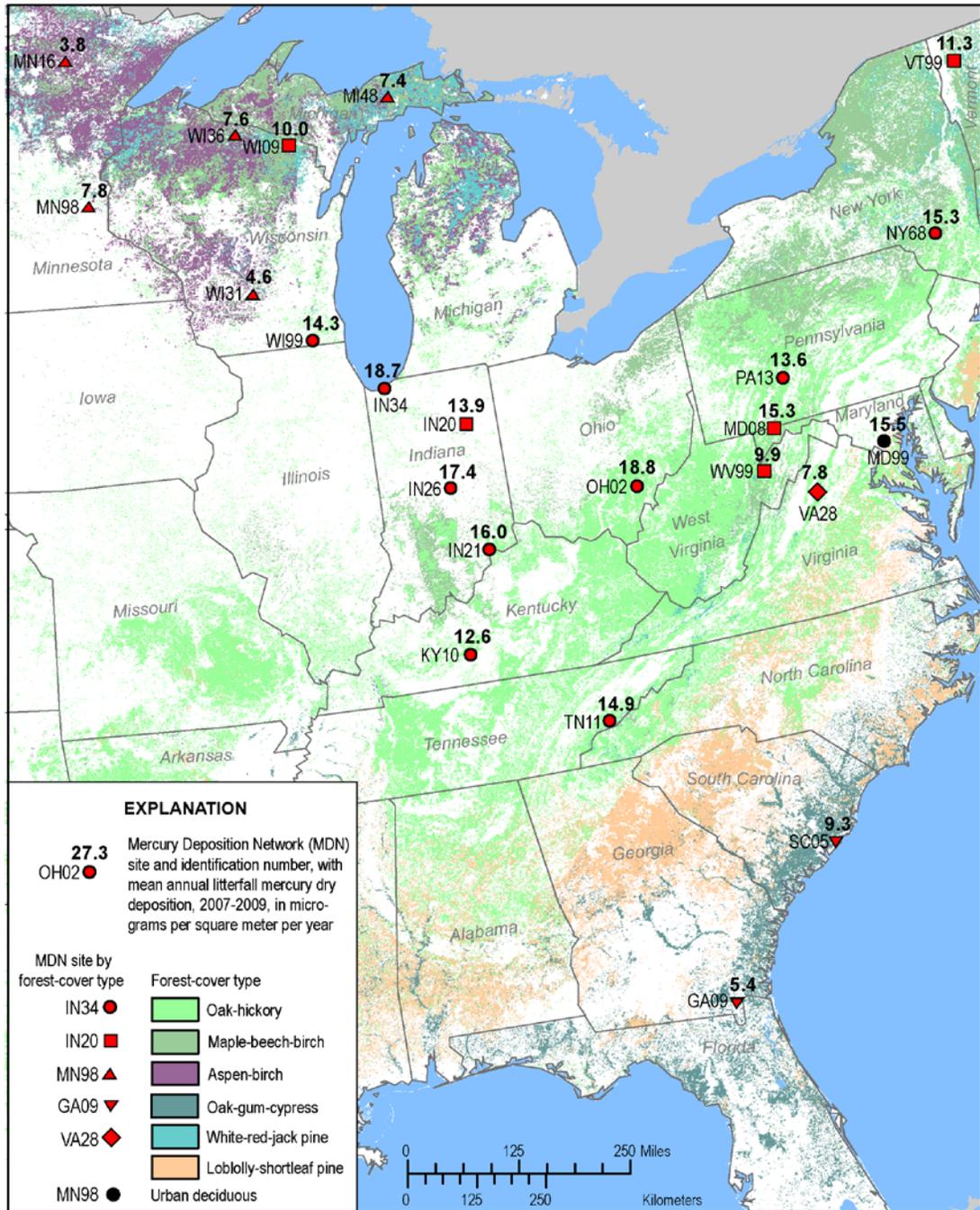[a]Ratio of mean MeHg to mean Hg concentration, as a percentage.

Table T5. Median values of annual litterfall Hg concentration, total sample catch, and annual Hg deposition by forest cover, 2007-2009.

[ng/g, nanogram per gram; g, gram; $\mu g/m^2$ microgram per square meter]

Forest-cover type/ Forest-cover class	Median of annual values for study sites					
	Litterfall Hg concentration (ng/g)	Total sample catch (g)[a]	Litterfall Hg deposition ($\mu g/m^2$)	Hg wet deposition ($\mu g/m^2$)	Total deposition ($\mu g/m^2$)[b]	Number of annual values
Aspen-birch	37.7	180	6.8	6.2	13.3	12
Maple-beech-birch	39.8	280	11.6	8.1	20.7	12
Oak-hickory	45.2	349	15.2	10.2	26.1	25
Oak-gum-cypress	42.6	170	7.3	10.8	18.0	4
White-red-jack pine	36.0	215	7.7	9.6	17.3	2
Urban deciduous	47.6	326	15.5	10.3	25.8	2
Deciduous	44.9	295	14.7	9.8	23.8	40
Mixed	38.6	233	9.3	9.0	18.0	12
Coniferous	38.6	182	7.0	6.1	13.2	5

[a]Includes adjusted values.

[b]Sum of litterfall Hg deposition and Hg wet deposition.

Fig. F1. Forest-cover types in the eastern USA with Mercury Deposition Network sites in litterfall study, showing mean annual litterfall Hg dry deposition, 2007–2009.

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