Plant mercury accumulation and litter input to a Northern Sedge-dominated Peatland

Ting Sun[1,2,3] and Brian A. Branfireun[4]

[1]School of Environmental Science and Engineering, Shandong University, Qingdao 266237, PR China

[2]Institute of Eco-Environmental Forensics, Shandong University, Qingdao 266237, PR China

[3]Department of Earth Sciences, University of Western Ontario, 1151 Richmond Street, London, Ontario, Canada

[4]Department of Biology, University of Western Ontario, 1151 Richmond Street, London, Ontario, Canada

Correspondence to: Ting Sun (tsun64@uwo.ca)

Abstract

Plant foliage plays an essential role in accumulating mercury (Hg) from the atmosphere and transferring it to soils in terrestrial ecosystems, w~~W~~hile many studies have focused on forested ecosystems. Hg input from plants to northern peatland peat soils has not been nearly as well studied and is likely equally important from a mass balance perspective. In this study, we investigated the accumulation of atmospheric Hg by the dominant plant species, few-seeded sedge [*Carex oligosperma* Michx.], wire sedge [*Carex lasiocarpa* Ehrh], tussock sedge [*Carex stricta* Lamb.], and sweet gale [*Myrica gale* L.] in a boreal sedge-dominated peatland. Foliar Hg concentrations decreased early in the growing season due to growth dilution~~.~~, and after that ~~Foliar Hg concentrations~~ were subsequently positively correlated with leaf age (time). Hg concentrations were 1.4-1.7 times higher in sweet gale than in sedges. A leaching experiment showed that sweet gale leached less Hg but more bioaccessible dissolved organic matter (DOM) by mass than sedges. Leaching of Hg was positively related to the aromaticity of DOM in leachate, suggesting the importance of DOM with higher aromaticity in controlling Hg mobility. Annual inputs of Hg through senesced leaf material to peat soils were 9.88 mg/ha/yr, 1.62 mg/ha/yr, and 8.29 mg/ha/yr for sweet gale, tussock sedge, and few-seeded sedge/wire sedge, respectively. Future investigations into foliar Hg accumulation and input from other plant species to the sedge-dominated peatland are needed to estimate the annual Hg inputs precisely.

1 Introduction

Mercury (Hg), especially methylmercury (MeHg), is a global concern due to its potential toxicity and ubiquitous presence in the environment (Morel et al., 1998). Hg is emitted to the atmosphere from both natural (e.g., volcanoes, wildfires, geothermal activity) and anthropogenic sources (e.g., coal combustion, artisanal gold mining, incineration) (Schroeder and Munthe, 1998; Streets et al., 2011). Atmospheric Hg exists as gaseous elemental mercury (GEM, Hg(0)), reactive gaseous mercury (RGM, Hg(II)), and particulate-bound mercury (PBM, Hg_p) with GEM as the dominant species (> 95 %) (Schroeder and Munthe, 1998). RGM and PBM have shorter atmospheric residence time ranging from hours to days, whereas GEM has a longer atmospheric residence time of several months to a year and thus is transported globally (Schroeder and Munthe, 1998). These atmospheric Hg species are eventually deposited into aquatic and terrestrial ecosystems via wet deposition (precipitation, such as rain, snow, and fog) and dry deposition (particle settling or direct partitioning to vegetation, water, and soil surface, or direct absorption by vegetation foliage) (Lindberg et al., 2007). Hg dry deposition is a larger input than wet deposition to vegetated terrestrial landscapes, contributing 70 %~85 % of total Hg deposition (dry and wet deposition) in terrestrial ecosystems (Graydon et al., 2008; Risch et al., 2017; Risch et al., 2012; St. Louis et al., 2001; Wang et al., 2016; Zhang et al., 2016), and more than 70 % of Hg dry deposition is by vegetation litterfall/incorporation into soil organic matter (SOM) (Obrist et al., 2017; Wang et al., 2016).

Vegetation is generally considered a sink for atmospheric Hg, with the majority of Hg in vegetation leaves accumulated from the atmosphere (Jiskra et al., 2018; Obrist et al., 2017). Plant leaves accumulate Hg from the atmosphere mainly through stomatal uptake (Lindberg et al., 1992). Stamenkovic and Gustin (2009) suggested that the non-stomatal pathway of Hg deposition to the leaf cuticle and subsequently retention and incorporation into leaf tissue also plays an important role in accumulating atmospheric Hg. Plant roots are thought to generally act as a barrier of Hg transport from soils to shoots (Wang et al., 2015), and it has been shown that less than 10% of Hg in roots is transported to the aboveground portion of plants (Ericksen et al., 2003; Mao et al., 2013). Some studies have found that a great proportion of foliar Hg in halophytes in salt marshes was translocated from the root (Canário et al., 2017; Cabrita et al., 2019; Weis and Weis 2004). The plausible reason is that plants in the hydroponic growth system have fewer apoplastic barriers (i.e. Casparian bands and suberin lamellae) in root architecture than plants grown in contaminated soils (Redjala et al., 2011).

Forest ecosystems are important sinks of atmospheric Hg and have received widespread attention from researchers (Risch et al., 2012; St. Louis et al., 2001; Wang et al., 2016; Zhang et al., 2009); however, studies about foliar Hg accumulation in other plant

types in ~~or other ecosystems such as~~ boreal peatlands ecosystems ~~-~~ are few (see Moore et al., 1995) despite their critical role in the carbon (Gorham, 1991) and Hg cycles (Grigal, 2003). Boreal peatlands are a type of wetland that stores large amounts of Hg ~~store 500 ± 100 Gt of carbon as peat (partially decomposed vegetation matter) due to slow decomposition rates in their anaerobic and acidic conditions and low temperatures (Rydin and Jeglum, 2013). In addition, boreal peatlands are sinks for inorganic Hg~~ (Grigal 2003) ~~(St. Louis et al., 1994),~~ and can be major MeHg sources to downstream ecosystems (Branfireun et al., 1996; Mitchell et al., 2008; St. Louis et al., 1994), given their anaerobic conditions, non-limiting amounts of inorganic Hg, and often available but limited amounts of sulfate (Blodau et al., 2007; Schmalenberger et al., 2007) and bioaccessible carbon facilitating net MeHg production (Mitchell et al., 2008). Elucidation of foliar Hg input from the dominant plant types to boreal peatlands is important to further estimate the supply of bioavailable Hg(II) for net MeHg production.

~~Little information is available about the amount of atmospheric Hg accumulated in leaves in peatlands. Moore et al. (1995) reported that Hg levels in nonvascular plants (fungi, lichens, and mosses) are almost an order of magnitude higher than those in vascular plants in wetlands, and the Hg concentrations follow the sequence: grassland herbs < trees and shrubs < aquatic macrophytes < *Sphagnum* spp. mosses < lichens < fungi. However, the annual input of aboveground vegetation biomass to peatland soils as senesced litter from vascular plants is greater than from bryophytes (Frolking et al., 2001) in fen-type peatlands dominated by sedges and shrubs, so despite having lower Hg concentrations, the mass input may be significant. With more bioaccessible litter and leachate than bryophytes, vascular plant inputs may also decompose at faster rates, releasing Hg to the soil and/or facilitating net methylation (Hobbie, 1996; Lyons and Lindo, 2019).~~

Previous studies have found that the majority of Hg in plant leaves in wetlands was from the atmosphere (Brahmstedt et al., 2021; Enrico et al., 2016; Fay and Gustin 2007) and nonvascular plants (e.g., fungi, lichens, and mosses) had higher foliar Hg concentrations than vascular plants (Moore et al., 1995; Pech et al., 2022). Although foliar Hg concentration is lower in vascular plants than in nonvascular plants, Hg mass input to peatlands may be substantial, given the greater litter input from vascular plants than from nonvascular plants (Frolking et al., 2001). With more bioaccessible litter than bryophytes (Hobbie, 1996; Lyons and Lindo, 2019), vascular plants also have a faster initial decomposition rate (0.2 $y_{▲}^{-1}$) than bryophytes (0.05-0.08 $y_{▲}^{-1}$) (Frolking et al., 2001), leading to a rapid Hg release to the soil. Boreal peatlands are experiencing rising temperatures due to climate change (IPCC, 2018) which is likely to both increase aboveground biomass in vascular plant-dominated peatlands (Tian et al., 2020) and promote a shift from moss-dominated to more vascular plant-dominated plant communities (Buttler et al., 2015;

设置了格式: 上标
设置了格式: 上标

Dieleman et al., 2015; Weltzin et al., 2000) further affecting Hg deposition (Zhang et al., 2016). To date, the amount of atmospheric Hg accumulated in dominant plants in the vascular plant-dominated (i.e., graminoid plants and shrubs) peatlands, an important type of boreal wetlands (Rydin and Jeglum, 2013), is unknown.

~~Boreal peatlands are experiencing rising temperatures due to climate change (IPCC, 2018) that is likely to either increase aboveground biomass and leaf area in already vascular plant-dominated peatlands or potentially shift moss-dominated peatlands to more vascular plant-dominated. Changes in plant abundance and community composition may further affect Hg deposition (Zhang et al., 2016) and input through litterfall, given that foliar Hg concentrations (Moore et al., 1995) and annual vegetation biomass input (Frolking et al., 2001) are different among plant species. Weltzin et al. (2000) found that the productivity of vascular plants in peatlands increased under higher temperatures. A full factorial greenhouse laboratory experiment of increased temperature and elevated atmospheric CO_2 resulted in increased peatland graminoid productivity (both above and belowground) (Tian et al., 2020). Several studies have also shown that experimental warming of northern peatlands mesocosms altered plant community composition, increasing vascular plant abundance and biomass, and decreasing *Sphagnum* spp. cover (Buttler et al., 2015; Dieleman et al., 2015; Weltzin et al., 2000). Changes in plant abundance and biomass in northern peatlands are ecologically significant, given their role as a source of net MeHg production.~~

Foliar Hg eventually enters peat soils via litterfall and is expected to follow the sequence: (1) wash-off of aerosols, particles, and gases from leaf surfaces, (2) leaching of water-soluble components, and (3) incorporation into SOM after the microbial decomposition of litter. Leaching is the initial phase of litter breakdown in aquatic environments and can rapidly release up to 30 % dissolved matter, primarily dissolved organic matter (DOM) within 24 h after immersion of litter (Gessner et al., 1999). It has been established that dissolved organic matter (DOM) is closely related to Hg mobility in terrestrial and aquatic ecosystems (Haitzer et al., 2002; Ravichandran, 2004; Kneer et al., 2020), given the strong affinity between Hg and reduced sulfur groups (i.e., thiols) in DOM (Xia et al., 1999). DOM with higher aromaticity has more thiols ligands and has a stronger correlation with Hg (Dittman et al., 2009). The rapid and abundant leaching of DOM, especially those with higher aromaticity from litterfall may lead to large amounts of Hg leaching. The amount of rapidly released Hg during litter leaching is unknown and needs to be elucidated because more recently deposited Hg appears to be more readily methylated than “old” Hg in peat soils (Branfireun et al., 2005; Feng et al., 2014; Hintelmann et al., 2002). Despite previous studies showing that Hg mass in live leaf leachate is

insignificant compared to that on leaf surfaces and in SOM (Rea et al., 2001; Rea et al., 2000), litterfall generally lacks structural integrity and likely leaches more Hg compared to live leaves.

The overall objective of this study is to link the vascular plant community (i.e., sedges and shrubs) to the peatland Hg cycle in a vascular plant-dominated fen-type peatland. We use "sedge-dominated fen" instead of "vascular plant-dominated fen-type peatland" hereafter, given that sedges are the primarily dominant plants in this study site (Webster and McLaughlin, 2010). The specific objectives of this study are to:

(1) quantify the mass accumulation of atmospherically-derived Hg in leaves of dominant plant species in a sedge-dominated fen over a growing season;

(2) estimate the Hg input from the litter of different plant species and through litter leaching to peat soils;

(3) clarify the role of DOM characteristics in controlling Hg leaching;

(4) estimate the annual areal loading of foliar Hg of different plant species to peat soils.

2 Materials and methods

2.1 Study site

Samples were collected from a sedge-dominated fen (10.2 ha) located in an 817 ha sub-watershed of the Lake Superior basin near White River Ontario, Canada (48°21' N, 85°21' W). The growing season is roughly from June to September. The sedge-dominated fen is mostly open and the vegetation community is dominated by three sedge species: few-seeded sedge [*Carex oligosperma* Michx.]; wire sedge [*Carex lasiocarpa* Ehrh]; and tussock sedge [*Carex stricta* Lamb.] (Lyons and Lindo, 2019). Sweet gale [*Myrica gale* L.] is the dominant shrub at this site (Lyons and Lindo, 2019; Palozzi and Lindo, 2017). The mean species percent cover of few-seeded sedge, wire sedge, tussock sedge, and sweet gale from the sedge-dominated fen is 35.0 ± 21.79%, 0.3 ± 0.12%, 73.0 ± 18.81%, 44.8 ± 10.63% (average ± standard error (SE)), respectively (Palozzi and Lindo 2017). Details of the study site and the characteristics of these plants are provided in the Supporting Information (SI). In this study, few-seeded sedges and wire sedges were mixed during plant sample collection as they are indistinguishable in size and form from one another when not in flower/seed, and frequently co-occur.

2.2 Sample collection and analysis

Five locations several hundred meters apart were selected in the sedge-dominated fen to serve as within-site replicates to account for potential local-scale variability. These five locations were roughly evenly distributed over this study area. Approximately fifty whole leaves of each few-seeded sedge/wire sedge, tussock sedge, and sweet gale were collected from each location using a clean blade in the middle of June, July, August, and after senescence at the beginning of October 2018 in each location, totaling 60 samples. For the October sampling event, the sedge leaves were still standing with the lower sections green, and although senesced, shrub leaves were sampled from the branch to ensure that there was no mixing with previous years' fallen leaves. Disposable nitrile gloves were worn during the sample collection. All samples were double bagged with two polyethylene bags and transported to the lab using a clean cooler. Leaves of each species that were collected from each ~~plot~~ location in October 2018 were divided for foliar total Hg (THg) analyses and a foliar Hg leaching experiment. Leaves were stored frozen until they were returned to the university laboratory.

For estimation of annual biomass of senesced leaf, seven 0.25 m^2 (0.5 m × 0.5 m) plots several hundred meters apart were selected at the end of August 2019 during senescence and before leaf off. All aboveground biomass of few-seeded sedge/wire sedge and tussock sedge and all aboveground leaf biomass of sweet gale were collected separately using a clean blade from each 0.25 m^2 plot. All vegetation samples were stored by species in paper bags, transported to the lab, and then oven-dried at 60 °C for a minimum of 48 h. Dried leaves of each species in each plot were sorted and weighed to estimate senesced leaf biomass of each species for each plot. The senesced leaf biomass of each species per hectare per year was calculated and expressed as mg/ha/yr.

Foliar total mercury, C content and N content. In the laboratory, leaf samples for chemical analyses were rinsed three times with deionized water (18.2 MΩ cm), freeze-dried for 48 h, ground and homogenized, and then analyzed using a Milestone™ DMA-80 (EPA method 7473). Leaf C content (%C; w/w) and N content (%N; w/w) before and after the foliar Hg leaching experiment was analyzed using a CNSH analyzer (Vario Isotope Cube; Elementar). The ratio of leaf C content and N content (C:N) were calculated. Detailed information concerning analytical methods are described in the SI, including analysis of foliar total Hg, %C, and %N.

Foliar mercury leaching experiment. The foliar leaching experimental procedure followed the design of Rea et al. (2000) and Del Giudice and Lindo (2017). Senesced leaves of sedges and sweet gale collected in October 2018 were rinsed twice with 100

mL of deionized water (18.2 MΩ cm) to quantify particulate or loosely-bound Hg and DOM that can be easily removed/leached from the leaf surface. This water was reserved for subsequent analysis. After rinsing, the leaves were oven-dried at a low temperature (40 °C) for 48 h, and then the leaves of each species from each location were relatively evenly separated into three groups and weighed, totaling 45 groups. These oven-dried senesced leaf samples were immersed in 150 mL of deionized water in clean 250 mL PETG bottles. All PETG bottles were capped, double bagged, and incubated in the dark at room temperature (~21 °C) for 48 h. Senesced leaf materials were gently swirled at the beginning of the leaching experiment to ensure complete wetting. Following the leaching, the leachate was vacuum filtered through a 0.45 μm glass fiber filter into clean 250 mL PETG bottles. Leachate from each sample was split into two aliquots. One was preserved by acidifying to 0.5 % (vol/vol) with high-purity HCl for dissolved total Hg (THg_{aq}) analysis and stored in 250 mL PETG bottles; the other was stored in the clean 60 mL Amber glass bottles and analyzed within 2 d for the quantity and characteristics of DOM. All samples were stored in the dark at 4 °C for further analysis. Method blanks of the leaching experiment were performed at the same time following the same procedure.

Senesced leaf material was taken out of each PETG bottle, oven-dried at 40 °C for 48 h, and re-weighed after leaching. The dry leaf weight before and after the leaching process was used to calculate the mass loss. These re-dried senesced leaf samples after leaching were ground and homogenized before the measurement for %C and %N as described above.

The dissolved total Hg (THg_{aq}) concentrations in the rinse water and leachate were analyzed using Environmental Protection Agency (EPA) method 1631. Dissolved organic matter is quantified analytically as dissolved organic carbon (DOC). DOC concentrations in rinse water and leachate were measured using an iTOC Aurora 1030 (OI Analytical, College Station, TX, USA) using the persulfate wet oxidation method. Details on the analytical procedures and QA/QC data for concentrations of THg_{aq} and DOC are provided in the SI.

DOM in leachate was characterized as specific ultraviolet absorbance at a wavelength of 254 nm ($SUVA_{254}$), an indicator of the molecular weight (or size) and aromaticity (the content of aromatic molecules) of DOM (Weishaar et al., 2003). Higher $SUVA_{254}$ values suggest that DOM contains more high-molecular-weight and aromatic molecules (Weishaar et al., 2003). Fluorescence excitation-emission matrices (EEMs) were also collected for calculating informative optical indices that reflect differences in DOM characteristics in leachate. The reported EEMs were then converted to optical indices using R Software (R Core Team 2012). Three common indices were chosen in this study: the fluorescence index (FI), the humification index (HIX_{EM}), and the biological index or 'freshness' index (BIX). Lower FI values (< 1.2) indicate that DOM is terrestrially derived (resulting from

decomposition and leaching of plant and soil organic matter) and has higher aromaticity, while higher FI values (> 1.8) indicate that DOM is microbially derived (originating from processes as extracellular release and leachate of algae and bacteria) and has lower aromaticity (Fellman et al., 2010; McKnight et al., 2001). High HIX_{EM} (> 1.0) values reflect the high humification of DOM and DOM is composed of more highly condensed and higher molecular weight molecules (Fellman et al., 2010; Hansen et al., 2016; Huguet et al., 2009; Ohno, 2002). Higher BIX values (> 1.0) reflect that more low-molecular-weight DOM was recently produced by microbes (Fellman et al., 2010; Huguet et al., 2009). Details on the analytical procedures and QA/QC data for $SUVA_{254}$, FI, HIX_{EM}, and BIX are provided in the SI.

3 Statistical analysis

Results were analyzed using IBM SPSS statistics software (IBM SPSS Inc. 24.0). The repeated-measures ANOVA was performed to compare the difference in foliar THg concentrations among different plant species over the growing season and to analyze the effect of leaf age on foliar Hg concentrations. Linear regressions were analyzed to examine the relationship between foliar THg accumulation and leaf age. Differences in ~~the~~ foliage quality (%C, %N, and C:N) were analyzed using a multivariate ANOVA. One-way ANOVA was used to determine the effects of plant species on concentrations of THg_{aq} and DOM quantity and characteristics in leachate. The repeated-measures ANOVA, multivariate ANOVA, and one-way ANOVA were followed by a *post hoc* test (Bonferroni's significant difference; honestly significant difference at the 95 % confidence interval). Weighed least squares regression was used to examine the nature of the relationship between THg_{aq} concentrations and $SUVA_{254}$ in leachate. Data are presented as the mean ± standard deviation (SD). Coefficient of determination (R^2) and significance p-values (p) are presented for linear regression fits, and $p < 0.05$ was considered significant.

4 Results and discussion

4.1 Foliar mercury accumulation in peatland plants

Foliar THg concentrations were related to time/leaf age ($F_{(1.733,24.2636)} = 42.75108.86$, $p < 0.001$) and plant species ($F_{(1.232,1223.38)} = 29.3851.85$, $p < 0.001$) (Fig. 1). Based on *post hoc* tests, foliar THg concentrations were significantly different between plant species and between the sampling months, except that there was no significant difference in foliar THg concentrations between June and August. The mean foliar THg concentrations (n = 5) in June followed the sequence: few-seeded sedge/wire sedge < tussock sedge < sweet gale. In July foliar THg concentrations decreased by 30 % (few-seeded sedge/wire sedge), 40 % (tussock sedge), and 47 % (sweet gale), respectively. The decrease in THg concentrations is likely because of leaf growth dilution,

although changes in leaf biomass were not quantified as part of this study. Foliar THg concentrations were positively related to time after July (few-seeded sedge/wire sedge: $F_{(1,13)} = 185.79$, $p < 0.001$, $R^2 = 0.93$; tussock sedge: $F_{(1,13)} = 200.87$, $p < 0.001$, $R^2=0.94$; sweet gale: $F_{(1,13)} = 70.72$, $p < 0.001$, $R^2=0.84$). The mean foliar THg concentrations in October few-seeded sedge/wire sedge, tussock sedge, and sweet gale were 1.7, 1.3, and 2.0 times higher than the initial concentrations in June. This result showed a clear pattern of continuous THg accumulation in foliage in boreal peatland plant species ~~from the atmosphere~~ over time as has been shown for forests (Laacouri et al., 2013; Millhollen et al., 2006b; Rea et al., 2002), which can be attributed to foliar Hg accumulation from the air, given that plant roots act as a barrier of Hg transport from soils to shoots (Wang et al., 2015). Further studies are needed to quantify the contribution of atmospheric and soil Hg to foliar Hg.

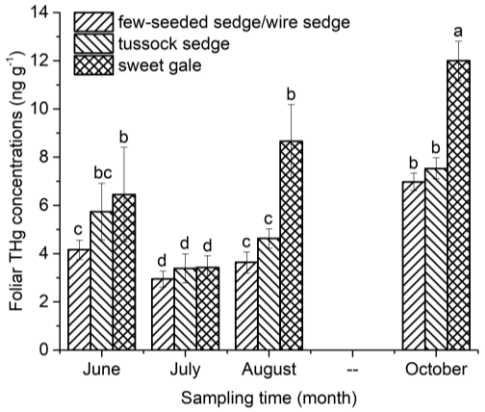


Figure 1 The intraseasonal trend in foliar total mercury (THg) concentrations (ng g^{-1}) of few-seeded sedge/wire sedge, tussock sedge, and sweet gale (ng g^{-1}). All concentrations are expressed in dry weight. Error bars are ± SD (n = 5 for each species for each time interval). The same letters above bars denote that values of foliar THg concentrations are not significantly different at the 0.05 levels.

Mercury accumulation in leaves is affected by many factors, such as atmospheric Hg concentration, environmental conditions (e.g., solar radiation and temperature), and biological factors (e.g., leaf age, plant species, leaf area, and leaf placement) (Blackwell and Driscoll, 2015; Ericksen et al., 2003; Ericksen and Gustin, 2004; Laacouri et al., 2013; Millhollen et al., 2006a). Since all samples were collected in the same location, factors such as atmospheric Hg concentration and environmental conditions were deemed the same, leaving only biological factors as an explanation for differences.

Leaf age. Leaf age is an important biological factor in controlling foliar concentrations (Ericksen et al., 2003; Laacouri et al., 2013). The positive relationship between foliar THg concentrations and time after July suggests that leaves of all species here

continued to assimilate atmospheric Hg over the growing season right up to senescence. Some studies have found that the rate of foliar Hg uptake decreased toward the end of the growing season (Ericksen et al., 2003; Laacouri et al., 2013; Poissant et al., 2008), which appears to be because of the decrease in~~of~~ photosynthetic activity at the end of the growing season (Koike et al., 2003). ~~Despite the decline of foliar Hg uptake during the late growing season~~In this study, foliar Hg concentrations continue to increase right up to senescence. Although foliar Hg can transport to other plant organs, such as tree rings (Arnold et al., 2018; McLagan et al., 2022), and/or can be re-emitted into the atmosphere (Zheng et al., 2016; Yu et al., 2016; Yuan et al., 2019), ~~because of the immobilization of~~ the majority of foliar Hg by mass is generally incorporated into leaf tissue (Laacouri et al., 2013; Lodenius et al., 2003; Stamenkovic and Gustin, 2009). In addition, it is likely that less than 10% of Hg in roots was transported to the leaves (Ericksen et al., 2003; Mao et al., 2013).

Plant species. Plant photosynthesis, transpiration, growth rates, and leaf area are different among plant species (Antúnez et al., 2001; Laacouri et al., 2013; Millhollen et al., 2006b), and given that these are important controls on Hg accumulation, the differences among species found in this study are not surprising. The mean foliar THg concentrations in tussock sedge were 1.2 times higher than that in few-seeded sedge/wire sedge, and although not measured as part of this study, tussock sedge has a larger leaf area than few-seeded sedge/wire sedge (Newmaster et al., 1997). A larger leaf has ~~more~~ a higher density of stomate~~s~~ and thus more leaf accumulation of atmospheric Hg (Laacouri et al., 2013; Millhollen et al., 2006; Stamenkovic and Gustin, 2009). A larger leaf area may also provide more adsorption sites for non-stomatal Hg uptake. Increased biomass corresponding with a bigger leaf area can offset the effects of stomate number on atmospheric Hg accumulation by leaves to a certain degree. A plausible explanation is that leaf biomass does not proportionally increase with leaf area and stomata, leading to a higher absolute Hg concentration in tussock sedge leaves than in few-seeded segde/wire sedge leaves. ~~The higher relative Hg concentrations in sweet gale (mean 1.7 and 1.4 times higher than few seeded/wire, and tussock sedge, respectively) is likely due in part to the same leaf area relationship. In addition,~~ Kozlowski and Pallardy (1997) reported that leaves near the top of the canopy generally have higher rates of photosynthesis and stomatal conductance than those near the bottom of the canopy due to light saturation. Sweet gale had potentially higher stomatal conductance due to higher incident radiation and vapor pressure deficits than sedges that are lower to the saturated ground with tightly packed vertical leaves.

Concentrations of Hg in senesced leaves of few-seeded sedge/wire sedge, tussock sedge, and sweet gale (6.58 ng g^{-1} to 12.77 ng g^{-1}) were lower than that reported in tree litter (~~17~~ 21 ng g^{-1} – ~~238~~ 78 ng g^{-1}) in North-America and Europe (Laacouri et al., 2013; Obrist et al., 2021; Poissant et al., 2008; Rea et al., 2002; Wang et al., 2016~~; Zhang et al., 2009~~) but similar to that previously

reported for ~~sedges~~ grasses and herbaceous plants~~and shrubs in Canada~~ (~~10.2 ± 6.8~~~10 ng g^{-1}) (Moore et al., 1995; Olson et al., 2019). The foliar Hg concentrations for plant species in this study increased 1.3-2.0 times over the growing season, which was smaller than that (3-11 fold) reported for trees (Laacouri et al., 2013; Poissant et al., 2008; Rea et al., 2002). The above results further confirm that foliar Hg concentrations differ among vegetation types (Demers et al., 2007; Moore et al., 1995; Obrist et al., 2012; Richardson and Friedland, 2015). It has been suggested that Hg previously retained in leaves can be photo reduced to Hg0 that is re-emitted to the atmosphere, and consistent Hg0 re-emission from the foliage is positively related to photosynthetically active radiation (PAR) (Yuan et al., 2019). The plants in open boreal peatlands lacking a tree overstorey like that in this study would receive very high exposure to ultraviolet (UV), which may result in a greater photoreduction of Hg previously retained in leaves and then Hg loss than tree leaves that are more often shaded. Moreover, despite angiosperms having higher stomatal conductance due to fewer stomata but more numbers (de Boer et al., 2016; Jordan et al., 2015), stomatal opening in dark-adapted leaves after light exposure was generally faster in gymnosperms than in angiosperms but stomatal closing upon the darkness of light-adapted leaves was faster in angiosperms than in gymnosperms (Xiong et al., 2018). This phenomenon may lead to a higher Hg concentration in trees (a type of gymnosperms) than in sedges and sweet gales (two types of angiosperms). More studies are needed to elucidate this mechanism of foliar Hg accumulation by different plant types.~~The foliar Hg concentrations for plant species in this study increased 1.3-2.0 times over the growing season, which was smaller than that (3-11 fold) reported for trees (Laacouri et al., 2013; Poissant et al., 2008; Rea et al., 2002). The above results suggested that foliar Hg concentrations differ among vegetation types (Demers et al., 2007; Moore et al., 1995; Obrist et al., 2012; Richardson and Friedland, 2015), which might be attributed to the larger leaf and higher stomatal density/ leaf placement in trees than sedges and shrubs.~~

域代码已更改
域代码已更改
域代码已更改
域代码已更改
域代码已更改
域代码已更改
域代码已更改

Leaf carbon, nitrogen and mercury. Leaf %C, %N, and C:N were significantly different among plant species ($F_{(6,104)} = 59.64$, $p < 0.001$) over the growing season ($F_{(9,124)} = 45.42$, $p < 0.001$) (Fig. 2). Based on *post hoc* tests, foliar %C, %N, and C:N was significantly different between sweet gale and sedges (few-seeded sedge/wire sedge and tussock sedge) but not between few-seeded sedge/wire sedge and tussock sedge. Foliar %C and %N were much lower in these sedges than sweet gale, which agrees well with a previous study that deciduous shrubs (i.e., sweet gale) generally have a higher foliar %C and %N than grasses (Wright et al., 2004). The fixation of nitrogen in sweet gale is in part attributed to sweet gale root nodules containing symbiotic nitrogen-fixing (Newmaster et al., 1997; Vitousek et al., 2002) with this greater amount of available N leading to higher photosynthetic capacity (Wright et al., 2004), thus, species containing a higher foliar %N are usually accompanied with a higher %C.

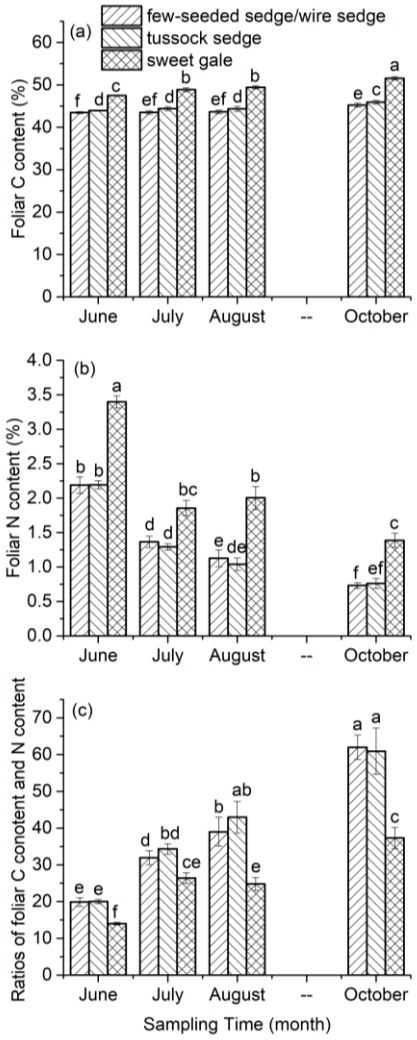
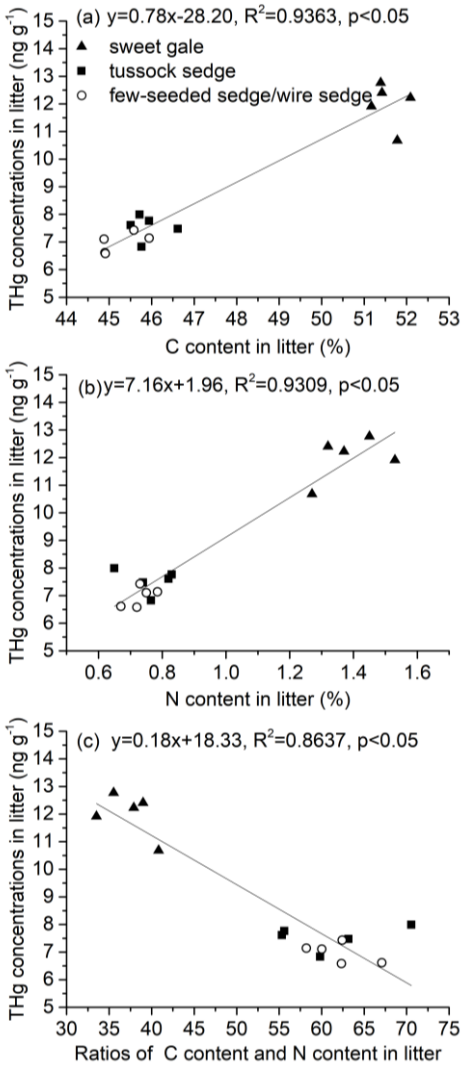


Figure 2 The carbon content (%C) (a), nitrogen content (%N) (b), and the ratio of carbon content to nitrogen content (C:N) (c) over the 2018 growing season. Vertical bars are mean ± SD (n = 5). The same letters above bars denote that values of foliar THg concentrations are not significantly different at the 0.05 levels.

There were significant increases in foliar %C (few-seeded sedge/wire sedge: $F_{(3,9)} = 25.98$, $p < 0.001$; tussock sedge: $F_{(3,9)} = 20.56$, $p < 0.001$; sweet gale: $F_{(3,9)} = 115.90$, $p < 0.001$) but sharp decreases ~~of~~ in foliar %N (few-seeded sedge/wire sedge: $F_{(1.34,4.03)} = 354.20$, $p < 0.001$; tussock sedge: $F_{(3,9)} = 252.36$, $p < 0.001$; sweet gale: $F_{(3,9)} = 170.43$, $p < 0.001$) over the growing season (Fig. 2). The strong decreases in foliar %N with leaf age can be attributed to the translocation of N from senescing leaves to new leaves (Wang et al., 2003). A study found that approximately 77 % N, 57 % phosphorus (P), and 44 % potassium (K) were translocated out of senescing leaves during mangrove leaf senescence (Wang et al., 2003). Foliar C is sequestrating continuously over the growing season (Kueh et al., 2013). The element re-translocation and C sequestration in leaves may lead to the foliar %C increase with time. The values of foliar C:N increased with time, which is a function of the decreases of foliar %N and the increases of foliar %C.

Senesced leaf tissue with higher foliar %C and %N had higher foliar THg concentrations (%C and Hg: $F_{(1,13)} = 191.09$, $p < 0.05$, $y = 0.78x - 28.20$, $R^2 = 0.94$;%N and Hg: $F_{(1,13)} = 82.38$, $p < 0.05$, $y = 7.16x - 1.96$, $R^2 = 0.93$) (Fig. 3a and 3b). THg concentrations were negatively related to foliar C:N during senescence ($F_{(1,13)} = 175.10$, $p < 0.05$, $y = 0.18x - 18.33$, $R^2 = 0.86$; Fig. 3c). A previous study found soil Hg concentrations were positively related to soil organic C and N, and then given a possible explanation that high C and N levels in soil reflect high vegetation productivity corresponding with high atmospheric Hg deposition via litterfall (Obrist et al., 2009). Although the mechanism of these relationships between Hg concentrations and contents of C and N in senesced leaves materials is still unclear, this study shows that Hg input via litterfall to soils can be affected by ~~higher~~ C and N content in senesced leaves. ~~indirectly indicates a higher input of Hg via litterfall to soils.~~ More studies and data are needed to draw predictive conclusions.



318

319 **Figure 3 Correlations between (a) THg concentrations and C contents, (b) THg concentrations and N contents, and (c) THg**
320 **concentrations and ratios of C content and N content (C:N) in litter. All linear correlations are statistically significant ($p < 0.05$).**

321 **4.2 Mercury leaching from senesced leaves**

322 **Surface-rinsable mercury.** The mean mass of Hg from the surface rinse of senesced leaf material (expressed per gram of dry
323 senesced leaf) was 0.02 ± 0.01 ng g^{-1} and 0.01± 0.00 ng g^{-1} (or 3.27 ± 1.68 ng L^{-1} and 1.39 ± 0.83 ng L^{-1}, expressed per liter of

rinse water (18.2 MΩ cm)), respectively, indicating that mass of Hg that was loosely bound on the leaf surface was small relative to the total senesced leaf tissue Hg concentration (8.83 ± 2.38 ng g^{-1}) representing on average only 0.4 % Hg (tussock sedge: 0.6 %; few-seeded sedge/wire sedge: 0.3 %; sweet gale: 0.3 %) of total THg mass.

Leachable mercury. The mean THg_{aq} mass per gram of senesced leaf had significant differences between plant species ($F_{(2,41)}$ = 11.55, $p < 0.001$; Fig. 4). Based on *post hoc* tests, there were significant differences in THg_{aq} mass per gram of senesced leaf between sweet gale and sedges (few-seeded sedge/wire sedge and tussock sedge) but not between few-seeded sedge/wire sedge and tussock sedge. The senesced leaf of sweet gale leached the least Hg among these plant species, which is likely due to their hydrophobic waxy cuticle that may both retain Hg, as well as protect the inner leaf material from leaching. Another plausible explanation is that N was more easily released from sedges than C and it was the opposite for sweet gale, based on changes in foliar %C and %N between before and after leaching (Table 1), whereas N groups in litter hinder the leaching of foliar Hg (Obrist et al., 2009). Foliar %N of sweet gale increased after leaching, which is likely attributed to a large amount of loss of other elements, such as K, Mg, and P, although they were not part of this experiment. Bessaad and Korboulewsky (2020) found that 60–79 % of K, 19–50 % of Mg, 22–30 % of P, and < 16 % of Ca and N were leached out from fully developed broadleaves (collected in summer) during rainfall.

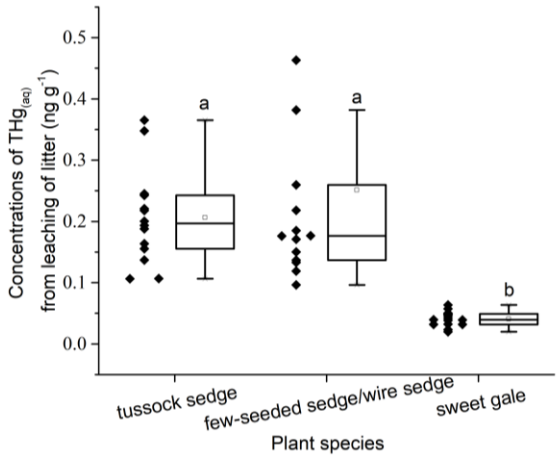


Figure 4 Mass of mercury leached per gram of senesced leaf material (ng g^{-1}). Boxplot displays median (50th percentile; the inside line of the box), first quartile (25th percentile; lower bound of the box), third quartile (75th percentile; upper bound of the box), whiskers (all measures between 5th percentile and 25th percentile and between 75th percentile and 95th percentile; the straight line below and above the box), and outliers (individual points outside of the percentile of 5th and 95th). n = 15.

Table 1 Changes of foliar carbon content (%C) and nitrogen content (%N) during leaching of litterfall. n = 15

	foliar %C			foliar %N		
	before leaching	after leaching	change percentage (%)	before leaching	after leaching	change percentage (%)
sweet gale	51.57 ± 0.36	51.03 ± 0.34	–1.05	1.39 ± 0.10	1.50 ± 0.07	–7.91
tussock sedge	45.91 ± 0.42	44.97 ± 0.54	–2.05	0.76 ± 0.07	0.68 ± 0.10	–10.53
few-seeded sedge/wire sedge	45.24 ± 0.49	43.83 ± 0.49	–3.12	0.73 ± 0.04	0.64 ± 0.02	–12.33

During experimental leaching, 3.0 %, 2.9 %, and 0.3 % of the total THg mass present in tussock sedge, few-seeded sedge/wire sedge, and sweet gale senesced leaf was leached, respectively. The percentages of Hg that leached from tussock sedge, few-seeded sedge/wire sedge leaves were 5.5 and 10.6 times higher than that from rinses, while the percentage of Hg that leached from sweet gale senesced leaf was similar to that from rinse water (0.3 %). Rea et al. (2000) reported that surface washoff of loosely bound and particulate Hg was a rapid and larger source of Hg in forest throughfall compared to continuously foliar Hg leaching from live leaves. It is likely because dry leaves lack structural integrity compared to live leaves in Rea et al.'s (2000) study, leading to more rapid leaching of soluble constituents (Gessner et al., 1999), including Hg, so the results of these prior studies are not directly comparable to this one. Further, although Hg leached from senesced leaf material was a small (< 5 % of foliar tissue Hg) but a measurable contributor to the mass balance, it is one that would be completely missed if material had been collected from a litter trap that had been exposed to rainfall for any period.

4.3 The roles of ~~Quantity and characteristics of leachate~~ dissolved organic matter properties in Hg mobility during litter leaching phase

The quantity and characteristics of DOM in leachate. The mean mass of DOC leached per gram of senesced leaf material and the mass loss during senesced leaf material leaching was significantly different between plant species (leached DOC mass: $F_{(2,42)} = 34.95$, $p < 0.001$; mass loss: $F_{(2,42)} = 11.62$, $p < 0.05$) with a same sequence following: few-seeded sedge/wire sedge < tussock sedge < sweet gale (Fig. 5). The same sequence is in part because the loss of soluble carbons accounted for the majority of the mass loss during litter leaching (Del Giudice and Lindo, 2017). Mass loss of sweet gale (17.7%) was significantly larger than sedges (few-seeded sedge/wire sedge (8.1%) and tussock sedge (11.5%)). The released DOC accounted for 22.96 ± 14.85%, 23.73 ± 12.95%, and 17.03 ± 6.68% of mass loss during senesced leaf material leaching for few-seeded sedge/wire sedge, tussock

带格式的: 居中

格式化表格

sedge, and sweet gale, respectively. Loss of other nutrients, such as dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) (Ong et al., 2017; Liu et al., 2018; Hensgens et al., 2020) and the inorganic components and other elemental organic matter (Lavery et al., 2013; Jiménez et al., 2017) also contribute to the mass loss, despite these nutrients not being measured.

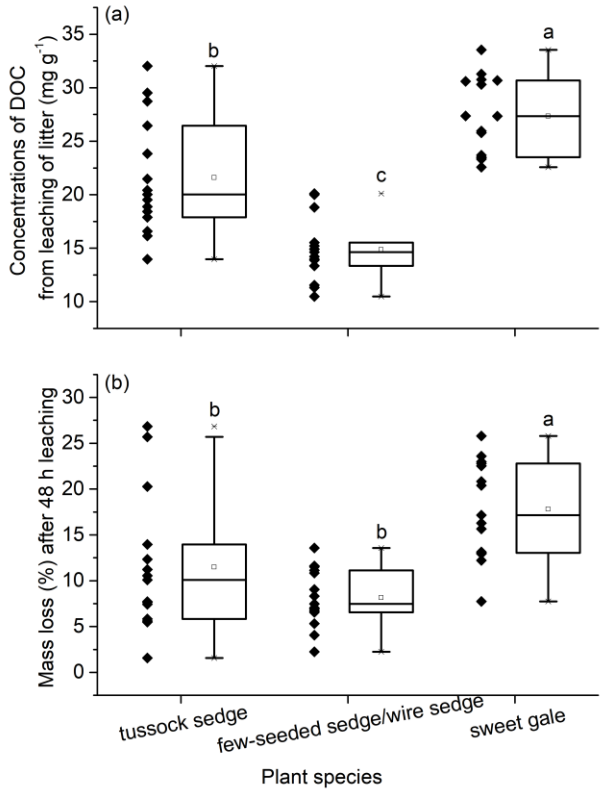


Figure 5 Mass of dissolved organic carbon (DOC) leached per gram of senesced leaf material (mg g^{-1}) (a), and mass loss after 48 h leaching (b). Boxplot displays median (50th percentile; the inside line of the box), first quartile (25th percentile; lower bound of the box), third quartile (75th percentile; upper bound of the box), whiskers (all measures between 5th percentile and 25th percentile and between 75th percentile and 95th percentile; the straight line below and above the box), and outliers (individual points outside of the percentile of 5th and 95th). n = 15.

Characteristics of DOM also varied among plant species ($SUVA_{254}$: $F_{(2,42)} = 24.02$, $p < 0.001$; HIX_{EM}:$F_{(2,42)} = 3.82$, $p < 0.05$; FI: $F_{(2,42)} = 11.24$, $p < 0.001$; BIX: $F_{(2,42)} = 125.48$, $p < 0.001$) (Fig. 6 and Table 2). Based on *post hoc* tests, there were significant

differences in $SUVA_{254}$ and FI between sweet gale and sedges (few-seeded sedge/wire sedge) only and in BIX among all plant species~~; there were no significant differences in HIX_{EM} among plant species~~. FI (1.2-1.8) and BIX (<1.0) reflected that DOM in leachate was generally of plant origin, suggesting that the microbially-derived OM was a smaller component. The mean value of $SUVA_{254}$ in leachate followed the sequence: tussock sedge ≈ few-seeded sedge/wire sedge > sweet gale leaves, respectively, indicating that leached DOM from tussock sedge and few-seeded sedge/wire sedge leaves had higher aromaticity and higher molecular weights than that from the sweet gale leaves. $SUVA_{254}$ was negatively related to DOM concentrations ($F_{(1,43)} = 48.37$, $p < 0.001$, $y = -0.69x + 3.93$, $R^2 = 0.53$) when all plant species were considered, suggesting that sweet gale prefers to release more amount of lower aromatic DOM.

Previous studies have found that characteristics of DOM controlled Hg mobility and methylation (Cui et al., 2022; Jiang et al., 2018; Ravichandran 2004; Xin et al., 2022; Wang et al., 2022). Hg is tightly and readily bound to reduced sulfur groups (i.e., thiols) in DOM (Ravichandran, 2004; Xia et al., 1999). Mercury weakly binds to carboxyl and phenol functional groups in DOM after all thiol groups are occupied at relatively high Hg concentrations (Drexel et al., 2002; Graham et al., 2012), which is atypical in most natural environments in which Hg concentrations are relatively low. Higher terrestrial (plant-derived) DOM had a greater DOM-Hg affinity (Wang et al., 2022). Additionally, DOM with higher aromaticity and molecular weight strongly bonded with Hg2+, potentially because these DOM provide more sulfidic groups such as thiols (Dittman et al., 2009; Wang et al., 2022). Therefore, terrestrial DOM and/or DOM with higher aromaticity and molecular weight may transport more Hg into peat soils during the litter leaching phase.

设置了格式: 下标
设置了格式: 下标
设置了格式: 下标
设置了格式: 上标

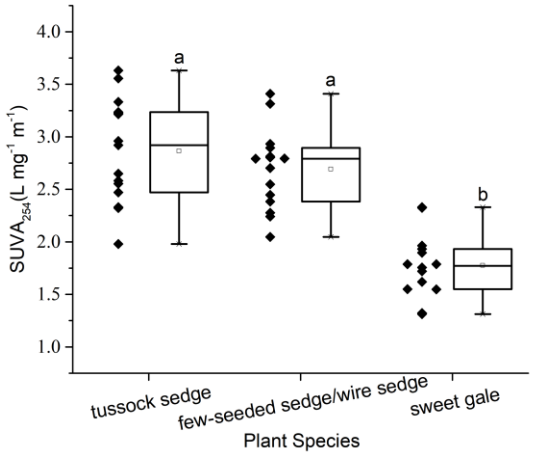


Figure 6 Dissolved organic matter characteristics as measured by specific ultraviolet absorbance at the wavelength 254 nm ($SUVA_{254}$), n = 15. Boxplot displays median (50th percentile; the inside line of the box), first quartile (25th percentile; lower bound of the box), third quartile (75th percentile; upper bound of the box), whiskers (all measures between 5th percentile and 25th percentile and between 75th percentile and 95th percentile; the straight line below and above the box), and outliers (individual points outside of the percentile of 5th and 95th).

Table 2 The mean fluorescence indices of dissolved organic matter characteristics[a]

Index	Tussock sedge	Few-seeded sedge/wire sedge	Sweet gale
FI	1.19 ± 0.10	1.31 ± 0.09	1.49 ± 0.27
HIX_{EM}	0.16 ± 0.03	0.16 ± 0.02	0.19 ± 0.03
BIX	0.53 ± 0.05	0.63 ± 0.06	0.35 ± 0.04

[a]Lower values of the FI (< 1.2) suggest dissolved organic matter (DOM) has higher aromaticity and is primarily composed of high-molecular-weight DOM, while high FI values (> 1.8) indicate that DOM has lower aromaticity and is mainly composed of low-molecular-weight DOM. DOM with high HIX_{EM} (> 1) values is composed of more highly condensed and higher molecular weight molecules. In contrast, higher BIX (> 1.0) values reflect that more low-molecular-weight DOM is recently produced, generally, by microbes. All indices are unitless, n = 15.

~~The mean value of $SUVA_{254}$ in leachate followed the sequence: tussock sedge > few-seeded sedge/wire sedge > sweet gale leaves, respectively, indicating that leached DOM from tussock sedge and few-seeded sedge/wire sedge leaves had higher aromaticity and less bioaccessible than that from the sweet gale leaves. These results are supported by indexes of FI and HIX_{EM}. DOM in senesced leaf material leachate of tussock sedge and few-seeded sedge/wire sedge had lower values of FI and HIX_{EM}~~

than that of sweet gale leaves, indicative of the presence of less bioaccessible and more aromatic DOM contents in sedges than in sweet gale. All BIX values (0.26–0.73) measured in this study were lower than 1.0, reflecting that DOM is mainly terrestrially derived (leaching from litterfall) in this study.

Although DOM leached from different litters has different characteristics, DOM leaching from litters is a substantial source to surrounding ecosystems (Davis et al., 2003; Davis et al., 2006; Del Giudice and Lindo, 2017). Importantly, the leached DOM (e.g., organic acids, sugars; amino acids) can provide energy and nutrients for microbes (Davis et al., 2003), which will subsequently stimulate biological degradation and Hg methylation.

4.4 Correlation between THg_{aq} concentrations and $SUVA_{254}$ values in leachate.

The concentrations of soluble THg_{aq} were significantly related to $SUVA_{254}$ values ($F_{(1,41)} = 52.06$, $p < 0.001$, $y = 0.09x - 0.09$, $R^2 = 0.55$; Fig. 7). Hg is tightly and readily bound to reduced sulfur groups (i.e., thiols) in DOM (Ravichandran, 2004; Xia et al., 1999), especially those with higher aromaticity that have more reduced sulfur groups (Dittman et al., 2009). Mercury weakly binds to carboxyl and phenol functional groups in DOM after all thiol groups are occupied at relatively high Hg concentrations (Drexel et al., 2002; Graham et al., 2012), which is atypical in most natural environments in which Hg concentrations are relatively low. This result suggested that agreed well with the literature indicating that DOM with higher aromaticity plays an important role in controlling Hg mobility, given that the number of reduced sulfur groups far exceeds the amount of Hg in natural environments (Ravichandran, 2004). The value of R^2 was only 0.55, which can be attributed that the number of reduced sulfur groups in DOM far exceeds the amount of Hg in natural environments and other factors, such as pH and sulfide may affect the binding between DOM and Hg (Ravichandran, 2004). In this study, DOM with higher aromaticity may transport more Hg from litter to soils, and senesced leaves of sedges had a higher potential in leaching Hg into peatland soils than the senesced leaves of sweet gales in this study.

带格式的: 正文, 段落间距段后: 0 磅

设置了格式: 字体: 10 磅

设置了格式: 字体: 10 磅, 上标

设置了格式: 字体: 10 磅

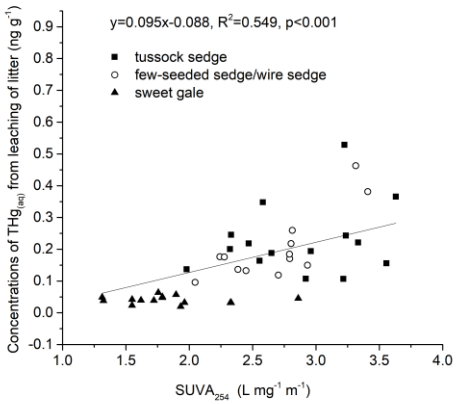


Figure 7 Correlations between the mass of mercury leached per gram of senesced leaf material (THg_{aq}) and the specific ultraviolet absorbance at the wavelength 254 nm ($SUVA_{254}$) in leachate.

4.5 Estimation of annual input of Hg via senesced leaves and rapid Leaching to peat soils

The annual input of leaf biomass (mg/ha/yr) of few-seeded sedge/wire sedge into peat soils was 5.55 and 1.41 fold higher than tussock sedge and sweet gale, while the annual inputs of Hg (mg/ha/yr) via sweet gale leaves were 6.29 and 1.22 fold higher than via tussock sedge and few-seeded/wire sedge leaves in the sedge-dominated fen (Table 3). Annual total Hg input through senesced leaves to peat soils were 1.56, 8.03, and 9.82 mg/ha/yr for tussock sedge, few-seeded sedge/wire sedge, and sweet gale, respectively. The input of surficial Hg and leachable Hg accounted for 0.64 % and 0.37 %, 0.31 % and 3.20 %, and 2.86 % and 0.30 % of total foliar Hg input to peat soils annually for tussock sedge, few-seeded sedge/wire sedge, and sweet gale, respectively. The majority of Hg in senesced leaves (> 96 %) was from the deposition of solid plant tissues in litter.

Table 3 Annual input of senesced leaves, and senesced leaf Hg, surficial Hg, and leached Hg during leaching into peat soils per hectare and per year in the sedge-dominated fen (mg/ha/yr).

Species	Senesced leaf biomass (mg/ha/yr)	Litter total Hg input (mg/ha/yr)	Washoff Hg input (mg/ha/yr)	Leachate Hg input (mg/ha/yr)
Tussock sedge	2.07×10^8	1.56	0.01	0.05
Few-seeded sedge/wire sedge	1.15×10^9	8.03	0.03	0.23
Sweet gale	8.18×10^8	9.82	0.03	0.03
Total	2.17×10^9	19.41	0.07	0.31

Based on the data from the study growing season, the annual input of Hg in total via senesced leaves (19.40 mg/ha/yr) was 5-22 % of litterfall in forest ecosystems (e.g., jack pine/black spruce/balsam fir forest, red maple/birch forest, Norway spruce forest; 86-372 mg/ha/yr) (St Louis et al., 2001; Graydon et al.,2008; Shanley and Bishop, 2012), which can be attributed to those forest ecosystems having both higher mean foliar Hg concentrations (21-51 ng g^{-1}) (Zhou and Obrist, 2021) and much greater aboveground biomass and litterfall inputs (2000-3488 kg/ha/yr) (Graydon et al., 2008) than plants in this study. The overall annual Hg inputs via these senesced leaves to peat soils in this sedge-dominated fen were 59 % of that via wet deposition using the mean precipitation Hg input estimates from the Experimental Lakes Area (33 mg/ha/yr) for the years 2001-2010, which is in the same general geographic region of Ontario (St Louis et al., 2019).

5 Conclusions

This study shows that the widely-observed pattern of foliar Hg ~~age~~ accumulation ~~of Hg~~ from the atmosphere and changes in foliar Hg concentrations over time are the same in peatland vascular plants as they are for forest trees and that the patterns are related to time/leaf age and plant species. ~~Although~~ The THg concentrations in ~~litterfall~~ senesced leaves in this study are relatively lower than that in the forest litterfall. ~~in this study are relatively lower than that in the forest litterfall, Hg input through litterfall to peatland soils cannot be neglected, given that peatlands are "hotspots" of MeHg production (Mitchell et al., 2008). Foliar leaching of lower molecular weight DOM from peatland shrubs such as sweet gale provides energy for bacteria (including~~

~~Hg methylators) and can enhance microbial metabolism.~~ Hg released from ubiquitous sedge litter during leaching is relatively more quickly than the much slower release of tissue-associated Hg through the decomposition of plant tissues. Thus, the supply of inorganic Hg to sites of methylation in peatlands has both fast and slow pathways that may shift under climate change, given that peatland plant species composition and biomass will certainly change under climate change.

■DATA AVAILABILITY

All data generated or analysed during this study are included in this published article and its supplementary information files.

■SUPPLEMENT

The supplement related to this article is available online.

■AUTHOR CONTRIBUTION

Ting Sun carried the project out, collected all samples, performed the leaching experiment, analyzed samples and data, and wrote the manuscript. Brian A. Branfireun designed the experiments, provided supervision, and edited the manuscript.

■COMPETING INTERESTS

The authors declare that they have no conflict of interest.

■DISCLAIMER

Publisher's note: Copernicus Publications remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

■ACKNOWLEDGMENTS

The research is supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) Strategic projects Grant (STPGP/479026-2015). We thank all members of Dr. Brian A. Branfireun and Dr. Zoë Lindo lab group for their help in the field.

■ABBREVIATIONS

Hg, mercury; MeHg, methylmercury; GEM, gaseous elemental mercury; RGM, reactive gaseous mercury; PBM, particulate-bound mercury; THg, total mercury; THg_{aq}, dissolved total mercury; SRB, sulfate-reducing bacteria; %C, carbon content; %N, nitrogen content; C:N, the ratio of leaf C content and N content; dissolved organic matter (DOM); DOC, dissolved organic carbon; $SUVA_{254}$, specific ultraviolet absorbance at a wavelength of 254 nm; EEMs, fluorescence excitation-emission matrices; FI, fluorescence index; HIX_{EM}, humification index; BIX, biological index; soil organic matter (SOM); CRM, certified reference material; RSD, relative standard deviation.

References

Antúnez, I., Retamosa, E. C., and Villar, R.: Relative growth rate in phylogenetically related deciduous and evergreen woody species, Oecologia, 128, 172-180, DOI:10.1007/s004420100645, 2001.

Arnold, J., Gustin, M. S., Weisberg, P. J.: Evidence for nonstomatal uptake of Hg by aspen and translocation of Hg from foliage to tree rings in Austrian pine, Environ. Sci. Technol., 52, 1174-1182, DOI10.1021/acs.est.7b04468, 2018.

Bessaad, A. and Korboulewsky, N.: How much does leaf leaching matter during the pre-drying period in a whole-tree harvesting system? Forest Ecol. Manag., 477, 118492, DOI:10.1016/j.foreco.2020.118492, 2020.

Blackwell, B. D. and Driscoll, C. T.: Deposition of mercury in forests along a montane elevation gradient. Environ. Sci. Technol., 49, 5363-5370, DOI:10.1021/es505928w, 2015.

Blodau, C., Mayer, B., Peiffer, S., Moore, T. R.: Support for an anaerobic sulfur cycle in two Canadian peatland soils, J. Geophy. Res-Biogeo., 112, G02004, DOI:10.1029/2006JG000364, 2007.

Brahmstedt, E. S., Crespo, C. N. A., Holsen, T. M., Twiss, M. R., Mercury distribution in an Upper St. Lawrence River wetland dominated by cattail (Typha angustifolia), Wetlands, 41, DOI10.1007/s13157-021-01511-9, 2021.

Branfireun, B. A., Heyes, A., and Roulet, N. T.: The hydrology and methylmercury dynamics of a Precambrian Shield headwater peatland, Water Resour. Res., 32, 1785-1794, DOI:10.1029/96WR00790, 1996.

Branfireun, B. A., Krabbenhoft, D. P., Hintelmann, H., Hunt, R. J., Hurley, J. P., Rudd, J. W. M.: Speciation and transport of newly deposited mercury in a boreal forest wetland: A stable mercury isotope approach. Water Resour. Res., 41, W06016, DOI:10.1029/2004WR003219, 2005.

Buttler, A., Robroek, B. J., Laggoun-Défarge, F., Jassey, V. E., Pochelon, C., Bernard, G., Delarue, F., Gogo, S., Mariotte, P., Mitchell, E. A. D.: Experimental warming interacts with soil moisture to discriminate plant responses in an ombrotrophic peatland. J. Veg. Sci., 26, 964-974, DOI:10.1111/jvs.12296, 2015.

Cabrita, M. T., Duarte, B., Cesario, R., Mendes, R., Hintelmann, H., Eckey, K., Dimock, B., Cacador, I., Canário, J.: Mercury mobility and effects in the salt-marsh plant Halimione portulacoides: Uptake, transport, and toxicity and tolerance mechanisms, Sci. Total Environ., 650, 111-120, DOI10.1016/j.scitotenv.2018.08.335, 2019.

Canário, J., Poissant, L., Pilote, M., Caetano, M., Hintelmann, H., O'Driscoll, N. J.: Salt-marsh plants as potential sources of Hg-0 into the atmosphere, Atmos. Environ.,152, 458-464, DOI10.1016/j.atmosenv.2017.01.011,2017.

Cui, H. Y., Zhao, Y., Zhao., L., Wei, Z. M.: Characterization of mercury binding to different molecular weight fractions of dissolved organic matter, J. Hazard. Mater., 431, DOI10.1016/j.jhazmat.2022.128593, 2022.

de Boer, H. J., Price, C. A., Wagner-Cremer, F., Dekker, S. C., Franks, P. J., Veneklaas, E. J.: Optimal allocation of leaf epidermal area for gas exchange, New Phytol., 210, 1219–1228, DOI10.1111/nph.13929, 2016.

~~Ong Demers, J. D., Driscoll, C. T., Fahey, T. J., Yavitt, J. B.: Mercury cycling in litter and soil in different forest types in the Adirondack region, New York, USA, Ecol. Appl., 17, 1341-1351, DOI:10.1890/06-1697.1, 2007.~~

~~Davis, S. E., Corronado-Molina, C., Childers, D. L., Day Jr, J. W.: Temporally dependent C, N, and P dynamics associated with the decay of *Rhizophora mangle* L. leaf litter in oligotrophic mangrove wetlands of the Southern Everglades, Aquat. Bot., 75, 199-215, DOI:10.1016/S0304-3770(02)00176-6, 2003.~~

~~Davis, S. E., Childers, D. L., and Noe, G. B. J. H.: The contribution of leaching to the rapid release of nutrients and carbon in the early decay of wetland vegetation. Hydrobiologia **569**, 87-97, DOI:10.1007/s10750-006-0124-1, 2006.~~

Demers, J. D., Driscoll, C. T., Fahey, T. J., Yavitt, J. B.: Mercury cycling in litter and soil in different forest types in the Adirondack region, New York, USA, Ecol. Appl., 17, 1341-1351, DOI:10.1890/06-1697.1, 2007.

Del Giudice, R. and Lindo, Z.: Short-term leaching dynamics of three peatland plant species reveals how shifts in plant communities may affect decomposition processes, Geoderma, 285, 110-116, DOI:10.1016/j.geoderma.2016.09.028,

2017.
Dieleman, C. M., Branfireun, B. A., McLaughlin, J. W., Lindo, Z.: Climate change drives a shift in peatland ecosystem plant community: implications for ecosystem function and stability, Glob. Change Biol., 21, 388-395, DOI:10.1111/gcb.12643, 2015.
Dittman, J. A., Shanley, J. B., Driscoll, C. T., Aiken, G. R., Chalmers, A. T., Towse, J. E.: Ultraviolet absorbance as a proxy for total dissolved mercury in streams, Environ. Pollut., 157, 1953-1956, DOI:10.1016/j.envpol.2009.01.031, 2009.
Drexel, R. T., Haitzer, M., Ryan, J. N., Aiken, G. R., Nagy, K. L.: Mercury (II) sorption to two Florida Everglades peats: evidence for strong and weak binding and competition by dissolved organic matter released from the peat, Environ. Sci. Technol., 36, 4058-4064, DOI:10.1021/es0114005, 2002.
Enrico, M., Le Roux, G., Marusczak, N., Heimburger, L. E., Claustres, A., Fu, X. W., Sun, R. Y., Sonke, J. E.: Atmospheric mercury transfer to peat bogs dominated by gaseous elemental mercury dry deposition, Environ. Sci. Technol., 50, 2405-2412, DOI10.1021/acs.est.5b06058, 2016.
Ericksen, J., Gustin, M., Schorran, D., Johnson, D., Lindberg, S., Coleman, J.: Accumulation of atmospheric mercury in forest foliage, Atmos. Environ., 37, 1613-1622, DOI:10.1016/S1352-2310(03)00008-6, 2003.
Ericksen, J. A. and Gustin, M.: Foliar exchange of mercury as a function of soil and air mercury concentrations, Sci. Total Environ., 324, 271-279, DOI10.1016/j.scitotenv.2003.10.034, 2004.
Fay, L., and Gustin, M. S.: Investigation of mercury accumulation in cattails growing in constructed wetland mesocosms, Wetlands, 27, 1056-1065, DOI10.1672/0277-5212(2007)27[1056:IOMAIC]2.0.CO;2, 2007.
Fellman, J. B., Hood, E., and Spencer, R. G.: Fluorescence spectroscopy opens new windows into dissolved organic matter dynamics in freshwater ecosystems: A review, Limnol. Oceanogr., 55, 2452-2462, DOI:10.4319/lo.2010.55.6.2452, 2010.
Frolking, S., Roulet, N. T., Moore, T. R., Richard, P. J., Lavoie, M., Muller, S. D.: Modeling northern peatland decomposition and peat accumulation, Ecosystems 4, 479-498, DOI10.1007/s10021-001-0105-1, 2001.
Gessner, M. O., Chauvet, E., and Dobson, M.: A perspective on leaf litter breakdown in streams, Oikos 85, 377-384, DOI10.2307/3546505, 1999.
Gorham, E.: Northern peatlands: Role in the carbon cycle and probable responses to climatic warming, Ecol. Appl., 1, 182-195, DOI:10.2307/1941811, 1991.
Graham, A. M., Aiken, G. R., and Gilmour, C. C.: Dissolved organic matter enhances microbial mercury methylation under sulfidic conditions, Environ. Sci. Technol., 46, 2715-2723, DOI:10.1021/es203658f, 2012.
Graydon, J. A., St. Louis, V. L., Hintelmann, H., Lindberg, S. E., Sandilands, K. A., Rudd, J. W., Kelly, C. A., Hall, B. D., Mowat, L. D.: Long-term wet and dry deposition of total and methyl mercury in the remote boreal ecoregion of Canada, Environ. Sci. Technol., 42, 8345-8351, DOI:10.1021/es801056j, 2008.
Haitzer, M., Aiken, G. R., and Ryan, J. N.: Binding of mercury (II) to dissolved organic matter: The role of the mercury-to-DOM concentration ratio, Environ. Sci. Technol., 36, 3564-3570, DOI:10.1021/es025699i, 2002.
~~Hansen, A. M., Kraus, T. E., Pellerin, B. A., Fleck, J. A., Downing, B. D., Bergamaschi, B. A.: Optical properties of dissolved organic matter (DOM): Effects of biological and photolytic degradation, Limnol. Oceanogr., 61, 1015-1032, DOI:10.1002/lno.10270, 2016.~~
Hensgens, G., Laudon, H., Peichl, M., Gil, I. A., Zhou, Q., Berggren, M.: The role of the understory in litter DOC and nutrient leaching in boreal forests, Biogeochemistry, 149, 87-103, DOI:10.1007/s10533-020-00668-5, 2020.
Hintelmann, H., Harris, R., Heyes, A., Hurley, J. P., Kelly, C. A., Krabbenhoft, D. P., Lindberg, S., Rudd, J. W., Scott, K. J., St. Louis, V. L.: Reactivity and mobility of new and old mercury deposition in a boreal forest ecosystem during the first year of the METAALICUS study, Environ. Sci. Technol., 36, 5034-5040, DOI:10.1021/es025572t, 2002.
Hobbie, S. E.: Temperature and plant species control over litter decomposition in Alaskan tundra, Ecol. Monogr., 66, 503-522,

DOI:10.2307/2963492, 1996.
Huguet, A., Vacher, L., Relexans, S., Saubusse, S., Froidefond, J. M., Parlanti, E.: Properties of fluorescent dissolved organic matter in the Gironde Estuary, Org. Geochem., 40, 706-719, DOI:10.1016/j.orggeochem.2009.03.002, 2009.
Jiang, T., Bravo, A. G., Skyllberg, U., Bjorn, E., Wang, D. Y., Yan, H. Y., Green, N. W.: Influence of dissolved organic matter (DOM) characteristics on dissolved mercury (Hg) species composition in sediment porewater of lakes from southwest China, Water Res., 146, 146-158, DOI10.1016/j.watres.2018.08.054, 2018.
Jiménez, M. A., Beltran, R., Traveset, A., Calleja, M. L., Delgado-Huertas, A., Marba, N.: Aeolian transport of seagrass (Posidonia oceanica) beach-cast to terrestrial systems, Estuar. Coast. Shelf S., 196, 31-44, DOI:10.1016/j.ecss.2017.06.035, 2017.
Jiskra, M., Sonke, J. E., Obrist, D., Bieser, J., Ebinghaus, R., Myhre, C. L., Pfaffhuber, K. A., Wängberg, I., Kyllönen, K., Worthy, D.: A vegetation control on seasonal variations in global atmospheric mercury concentrations, Nat. Geosc., 11, 244-250, DOI:10.1038/s41561-018-0078-8, 2018.
Jordan, G. J., Carpenter, R. J., Koutoulis, A., Price, A., & Brodribb, T. J.: Environmental adaptation in stomatal size independent of the effects of genome size. New Phytol., 205, 608–617, DOI10.1111/nph.13076, 2015.
Kneer, M. L., White, A., Rolfhus, K. R., Jeremiason, J. D., Johnson, N. W., Ginder-Vogel, M.: Impact of dissolved organic matter on porewater Hg and MeHg concentrations in St. Louis River estuary sediments, ACS Earch Space Chem., 4, 1386-1397, DOI10.1021/acsearthspacechem.0c00134, 2020.
Koike, T., Kitaoka, S., Ichie, T., Lei, T., Kitao, M.: Photosynthetic characteristics of mixed deciduous-broadleaf forests from leaf to stand, Terra, Publishing, Tokyo, Japan. 2003.
Kueh, J. H. R., Ab Majid, N. M., Seca, G., Ahmed, O. H.: Above ground biomass-carbon partitioning, storage and sequestration in a Rehabilitated forest, Bintulu, Sarawak, Malaysia, Sains Malays., 42, 1041-1050, 2013.
Laacouri, A., Nater, E. A., and Kolka, R. K.: Distribution and uptake dynamics of mercury in leaves of common deciduous tree species in Minnesota, USA, Environ. Sci.Technol., 47, 10462-10470, DOI:10.1021/es401357z, 2013.
Lavery, P. S., McMahon, K., Weyers, J., Boyce, M. C., Oldham, C. E.: Release of dissolved organic carbon from seagrass wrack and its implications for trophic connectivity, Mar. Ecol-Prog. Ser., 494, 121-133, DOI:10.3354/meps10554, 2013.
Lindberg, S., Bullock, R., Ebinghaus, R., Engstrom, D., Feng, X., Fitzgerald, W., Pirrone, N., Prestbo, E., Seigneur, C.: A synthesis of progress and uncertainties in attributing the sources of mercury in deposition, AMBIO, 36, 19-33, DOI:10.1579/0044-7447(2007)36[19:ASOPAU]2.0.CO;2, 2007.
Lindberg, S. E., Meyers, T. P., Taylor, G. E., Turner, R. R., Schroeder, W. H.: Atmosphere surface exchange of mercury in a forest: results of modeling and gradient approaches, J. Geophys. Res. Atmos., 97, 2519-2528, DOI10.1029/91JD02831, 1992.
Liu, S. L., Jiang, Z. J., Zhou, C. Y., Wu, Y. C., Arbi, I., Zhang, J. P., Huang, X. P., Trevathan-Tackett, S. M.: Leaching of dissolved organic matter from seagrass leaf litter and its biogeochemical implications, Acta Oceanol. Sin., 37, 84-90, DOI:10.1007/s13131-018-1233-1, 2018.
Lodenius, M., Tulisalo, E., and Soltanpour-Gargari, A.: Exchange of mercury between atmosphere and vegetation under contaminated conditions, Sci. Total Environ., 304, 169-174, DOI:10.1016/S0048-9697(02)00566-1, 2003.
Lyons, C. L. and Lindon, Z.: Above-and belowground community linkages in boreal peatlands, Plant Ecol., 221, 615-632, DOI:10.1007/s11258-020-01037-w, 2019.
Mao, Y. X., Li, Y. B., Richards, J., Cai, Y.: Investigating uptake and translocation of mercury species by Sawgrass (Cladium jamaicense) using a stable isotope tracer technique, Environ. Sci. Technol., 47, 9678-9684, DOI10.1021/es400546s, 2013.
McKnight, D. M., Boyer, E. W., Westerhoff, P. K., Doran, P. T., Kulbe, T., Andersen, D. T.: Spectrofluorometric characterization of dissolved organic matter for indication of precursor organic material and aromaticity. Limnol. Oceanogr., 46, 38-48,

带格式的: 两端对齐

DOI:10.4319/lo.2001.46.1.0038, 2001.
McLagan, D. S., Biester, H., Navratil, T., Kraemer, S. M., Schwab, L.: Internal tree cycling and atmospheric archiving of mercury: examination with concentration and stable isotope analyses, Biogeosciences, 19, 4415-4429, DOI10.5194/bg-19-4415-2022, 2022.
Millhollen, A., Obrist, D., and Gustin, M.: Mercury accumulation in grass and forb species as a function of atmospheric carbon dioxide concentrations and mercury exposures in air and soil, Chemosphere, 65, 889-897, DOI:10.1016/j.chemosphere.2006.03.008, 2006a.
Millhollen, A. G., Gustin, M. S., and Obrist, D.: Foliar mercury accumulation and exchange for three tree species, Environ. Sci. Technol., 40, 6001-6006, DOI:10.1021/es0609194, 2006b.
Mitchell, C. P. J., Branfireun, B. A., and Kolka, R. K.: Spatial characteristics of net methylmercury production hot spots in peatlands, Environ. Sci. Technol., 42, 1010-1016, DOI:10.1021/es0704986, 2008.
Moore, T., Bubier, J., Heyes, A., Flett, R.: Methyl and total mercury in boreal wetland plants, Experimental Lakes Area, Northwestern Ontario, J. Environ. Qual., 24, 845-850, DOI:10.2134/jeq1995.00472425002400050007x, 1995.
Morel, F. M., Kraepiel, A. M., and Amyot, M.: The chemical cycle and bioaccumulation of mercury, Annu. Rev. Ecol. Syst., 29, 543-566, DOI:10.1146/annurev.ecolsys.29.1.543, 1998.
Newmaster, S. G., Harris, A. G., and Kershaw, L. J.: Wetland plants of Ontario. Lone Pine Publishing, Edmonton, Canada, 1997.
Obrist, D., Agnan, Y., Jiskra, M., Olson, C., Colegrove, D., Hueber, J., Moore, C., Sonke, J., Helmig, D.: Tundra uptake of atmospheric elemental mercury drives Arctic mercury pollution, Nature, 547, 201–204, DOI:10.1038/nature22997, 2017.
Obrist, D., Johnson, D., and Lindberg, S.: Mercury concentrations and pools in four Sierra Nevada forest sites, and relationships to organic carbon and nitrogen, Biogeosciences, 6, 765-777, DOI:10.5194/bg-6-765-2009, 2009.
Obrist, D., Johnson, D. W., and Edmonds, R. L.: Effects of vegetation type on mercury concentrations and pools in two adjacent coniferous and deciduous forests, J. Plant Nutr. Soil Sc., 175, 68-77, DOI:10.1002/jpln.201000415, 2012.
Obrist, D., Roy, E. M., Harrison, J. L., Kwong, C. F., Munger, J. W., Moosmuller, H., Romero, C. D., Sun, S.W., Zhou, J., Commane, R.: Previously unaccounted atmospheric mercury deposition in a midlatitude deciduous forest, P. Natl. Acad. Sci. USA., 118, DOI:10.1073/pnas.2105477118, 2021.
~~Ohno, T.: Fluorescence inner-filtering correction for determining the humification index of dissolved organic matter, Environ. Sci. Technol., 36, 742-746, DOI:10.1021/es0155276, 2002.~~
~~Ong, C. S. P., Joon, C. J., and Yule, C. M.: The contribution of leaching to nutrient release from leaf litter of two emergent tree species in a Malaysian tropical peat swamp forest, Hydrobiologia, 794, 125-137, DOI:10.1007/s10750-017-3086-6, 2017.~~
Palozzi, J. E. and Lindo, Z.: Boreal peat properties link to plant functional traits of ecosystem engineers, Plant Soil, 418, 277-291, DOI:10.1007/s11104-017-3291-0, 2017.
Pech, P., Wojtun, B., Samecka-Cymerman, A., Polechonska, L., Kempers, A. J.: Metals in plant functional types of ombrotrophic peatlands in the Sudetes (SW Poland), Arch. Environ. Contam. Toxicol., 82, 506-519, DOI10.1007/s00244-022-00928-5, 2022.
Poissant, L., Pilote, M., Yumvihoze, E., Lean, D.: Mercury concentrations and foliage/atmosphere fluxes in a maple forest ecosystem in Quebec, Canada, J. Geophys. Res-Atmos., 113, DOI:10.1029/2007JD009510, 2008.
Ravichandran, M.: Interactions between mercury and dissolved organic matter: A review, Chemosphere, 55, 319-331, DOI:10.1016/j.chemosphere.2003.11.011, 2004.
Rea, A., Lindberg, S., Scherbatskoy, T., Keeler, G.J.: Mercury accumulation in foliage over time in two northern mixed-hardwood forests, Water Air Soil Poll., 133, 49-67, DOI:10.1023/A:1012919731598, 2002.
Rea, A. W., Lindberg, S. E., and Keeler, G. J.: Assessment of dry deposition and foliar leaching of mercury and selected trace elements based on washed foliar and surrogate surfaces, Environ. Sci. Technol., 34, 2418-2425, DOI:10.1021/es991305k,

2000.

Redjala, T., Zelko, I., Sterckeman, T., Legue, V., Lux, A.: Relationship between root structure and root cadmium uptake in maize, Environ. Exp. Bot., 71, 241-248, DOI10.1016/j.envexpbot.2010.12.010, 2011.

Richardson, J. and Friedland, A.: Mercury in coniferous and deciduous upland forests in northern New England, USA: implications of climate change, Biogeosciences, 12, 6737-6749, DOI:10.5194/bg-12-6737-2015, 2015.

Risch, M. R., DeWild, J. F., Gay, D. A., Zhang, L., Boyer, E. W., Krabbenhoft, D. P.: Atmospheric mercury deposition to forests in the eastern USA, Environ. pollut., 228, 8-18, DOI:10.1016/j.envpol.2017.05.004, 2017.

Risch, M. R., DeWild, J. F., Krabbenhoft, D. P., Kolka, R. K., Zhang, L. M.: Litterfall mercury dry deposition in the eastern USA, Environ. Pollut., 161, 284-290, DOI:10.1016/j.envpol.2011.06.005, 2012.

Rydin, H. and Jeglum, J. K.: The biology of peatlands. (2nd ed.), Oxford, United Kingdom, Oxford University Press. 2013.

Schaefer, J. K. and Morel, F. M.: High methylation rates of mercury bound to cysteine by *Geobacter sulfurreducens*, Nat. Geosci., 2, 123, DOI:10.1038/NGEO412, 2009.

Schaefer, J. K., Rocks, S. S., Zheng, W., Liang, L., Gu, B., Morel, F. M.: Active transport, substrate specificity, and methylation of Hg (II) in anaerobic bacteria, P. Natl. Acad. Sci. USA, 108, 8714-8719, DOI:10.1073/pnas.1105781108, 2011.

Schmalenberger, A., Drake, H. L., and Küsel, K.: High unique diversity of sulfate-reducing prokaryotes characterized in a depth gradient in an acidic fen, Environ. Microbiol., 9, 1317-1328, DOI:10.1111/j.1462-2920.2007.01251.x, 2007.

Schroeder, W. H. and Munthe, J.: Atmospheric mercury: An overview, Atmos. Environ., 32, 809-822, DOI:10.1016/S1352-2310(97)00293-8, 1998.

Shanley, J. B. and Bishop, K.: Mercury cycling in terrestrial watersheds. In: Banks, M. S. (Ed.), Mercury in the Environment: pattern and process, pp. 119-141. 2012.

Skyllberg, U., Xia, K., Bloom, P. R., Nater, E. A., Bleam, W. F.: Binding of mercury (II) to reduced sulfur in soil organic matter along upland-peat soil transects, J. Environ. Qual., 29, 855-865, DOI:10.2134/jeq2000.00472425002900030022x, 2000.

St. Louis, V. L., Graydon, J. A., Lehnherr, I., Amos, H. M., Sunderland, E. M., St. Pierre, K. A., Emmerton, C. A., Sandilands, K., Tate, M., Steffen, A., Humphreys, E. R.: Atmospheric concentrations and wet/dry loadings of mercury at the remote Experimental Lakes Area, Northwestern Ontario, Canada, Environ. Sci. Technol., 53, 8017-8026, DOI:10.1021/acs.est.9b01338, 2019.

St. Louis, V. L., Rudd, J. W., Kelly, C. A., Beaty, K. G., Bloom, N. S., Flett, R. J.: Importance of wetlands as sources of methyl mercury to boreal forest ecosystems, Can. J. Fish. Aquat. Sci., 51, 1065-1076, DOI:10.1139/f94-106, 1994.

St. Louis, V. L., Rudd, J. W., Kelly, C. A., Hall, B. D., Rolfhus, K. R., Scott, K. J., Lindberg, S. E., Dong, W.: Importance of the forest canopy to fluxes of methyl mercury and total mercury to boreal ecosystems, Environ. Sci. Technol., 35, 3089-3098, DOI:10.1021/es001924p, 2001.

Stamenkovic, J. and Gustin, M. S.: Nonstomatal versus stomatal uptake of atmospheric mercury, Environ. Sci. Technol., 43, 1367-1372, DOI:10.1021/es801583a, 2009.

Streets, D. G., Devane, M. K., Lu, Z., Bond, T. C., Sunderland, E. M., Jacob, D. J.: All-time releases of mercury to the atmosphere from human activities, Environ. Sci. Technol., 45, 10485-10491, DOI:10.1021/es202765m, 2011.

Tian, J., Branfireun, B. A., Lindo, Z.: Global change alters peatland carbon cycling through plant biomass allocation, Plant Soil, 455, 53-64, DOI10.1007/s11104-020-04664-4, 2020.

Ullrich, S. M., Tanton, T. W., and Abdrashitova, S. A.: Mercury in the aquatic environment: A review of factors affecting methylation, Crit. Rev. Env. Sci. Tec., 31, 241-293, DOI:10.1080/20016491089226, 2001.

Vitousek, P. M., Cassman, K., Cleveland, C., Crews, T., Field, C. B., Grimm, N. B., Howarth, R. W., Marino, R., Martinelli, L., Rastetter, E. B., Sprent, J. I.: Towards an ecological understanding of biological nitrogen fixation, Biogeochemistry, 57, 1-45, DOI:10.1023/A:1015798428743, 2002.

Wang, W. Q., Wang, M., and Lin, P.: Seasonal changes in element contents in mangrove element retranslocation during leaf senescene, Plant Soil, 252, 187-193, DOI:10.1023/A:1024704204037, 2003.

Wang, X., Bao, Z., Lin, C. J., Yuan, W., Feng, X.: Assessment of global mercury deposition through litterfall, Environ. Sci. Technol., 50, 8548-8557, DOI:10.1021/acs.est.5b06351, 2016.

Wang, X., Tam, N. F., He, H. D., Ye, Z. H.: The role of root anatomy, organic acids and iron plaque on mercury accumulation in rice, Plant Soil, 394, 301-313, DOI:10.1007/s11104-015-2537-y, 2015.

Wang, Y. Q., Liu, J., Van, L. N., Tian, S. Y., Zhang, S. Q., Wang, D. Y., Jiang, T.: Binding strength of mercury (II) to different dissolved organic matter: The roles of DOM properties and sources, Sci. Total Environ., 807, DOI10.1016/j.scitotenv.2021.150979, 2022.

Webster, K. L. and McLaughlin, J. W.: Importance of the water table in controlling dissolved carbon along a fen nutrient gradient, Soil Sci. Soc. Am. J., 74, 2254-2266, DOI:10.2136/sssaj2009.0111, 2010.

Weishaar, J. L., Aiken, G. R., Bergamaschi, B. A., Fram, M. S., Fujii, R., Mopper, K.: Evaluation of specific ultraviolet absorbance as an indicator of the chemical composition and reactivity of dissolved organic carbon, Environ. Sci. Technol., 37, 4702-4708, DOI:10.1021/es030360x, 2003.

Weltzin, J. F., Pastor, J., Harth, C., Bridgham, S. D., Updegraff, K., Chapin, C. T.: Response of bog and fen plant communities to warming and water-table manipulations, Ecology, 81, 3464-3478, DOI:10.1890/0012-9658(2000)081[3464:ROBAFP]2.0.CO;2, 2000.

Wright, I. J., Reich, P. B., Westoby, M., Ackerly, D. D., Baruch, Z., Bongers, F., Cavender-Bares, J., Chapin, T., Cornelissen, J. H. C., Diemer, M., Flexas, J., Garnier, E., Groom, P. K., Gulias, J., Hikosaka, K., Lamont, B. B., Lee, T., Lee, W., Lusk, C., Midgley, J. J., Navas, M. L., Niinemets, U., Oleksyn, J., Osada, N., Poorter, H., Poot, P., Prior, L., Pyankov, V. I., Roumet, C., Thomas, S. C., Tjoelker, M. G., Veneklaas, E. J., Villar, R.: The worldwide leaf economics spectrum, Nature, 428, 821-827, DOI:10.1038/nature02403, 2004.

Xia, K., Skyllberg, U., Bleam, W., Bloom, P., Nater, E., Helmke, P.: X-ray absorption spectroscopic evidence for the complexation of Hg (II) by reduced sulfur in soil humic substances, Environ. Sci. Technol., 33, 257-261, DOI:10.1021/es980433q, 1999.

Xin, Y., Zhang, X. H., Zheng, D. M., Zhang, Z. S., Jiang, M.: Impacts of spectral characteristics of dissolved organic matter on methylmercury contents in peatlands, Northeast China, Environ. Geochem. Hlth., DOI10.1007/s10653-022-01257-1, 2022.

Xiong, D. L., Douthe, C., Flexas, J.: Differential coordination of stomatal conductance, mesophyll conductance, and leaf hydraulic conductance in response to changing light across species, Plant Cell Environ., 41, 436-450, DOI10.1111/pce.13111, 2018.

Yu, B.; Fu, X. W.; Yin, R. S.; Zhang, H.; Wang, X.; Lin, C. J.; Wu, C. S.; Zhang, Y. P.; He, N. N.; Fu, P. Q.; Wang, Z. F.; Shang, L. H.; Sommar, J.; Sonke, J. E.; Maurice, L.; Guinot, B.; Feng, X. B.: Isotopic composition of atmospheric mercury in China: new evidence for sources and transformation processes in air and in vegetation, Environ. Sci. Technol., 50, 9262−9269, DOI10.1021/acs.est.6b01782, 2016.

Yuan, W., Sommar, J., Lin, C. J., Wang, X., Li, Kai, Liu, Y., Zhang, H., Lu, Z. Y., Wu, C. S., Feng, X. B.: Stable isotope evidence shows re-emission of elemental mercury vapor occurring after reductive loss from foliage, Environ. Sci. Technol., 53, 651-660, DOI10.1021/acs.est.8b04865, 2019.

Zhang, H., Holmes, C., and Wu, S.: Impacts of changes in climate, land use and land cover on atmospheric mercury, Atmos. Environ., 141, 230-244, DOI:10.1016/j.atmosenv.2016.06.056, 2016.

Zhang, L., Wright, L. P., and Blanchard, P.: A review of current knowledge concerning dry deposition of atmospheric mercury, Atmos. Environ., 43, 5853-5864, DOI:10.1016/j.atmosenv.2009.08.019, 2009.

Zheng, W., Obrist, D., Weis, D., Bergquist, B. A.: Mercury isotope compositions across North American forests, Global

带格式的: 两端对齐

752 Biogeochem Cyles, 30, 1475-1492, DOI10.1002/2015GB005323, 2016.
753
754