Fate of mercury in the Degerö Stormyr peatland, Northern Sweden

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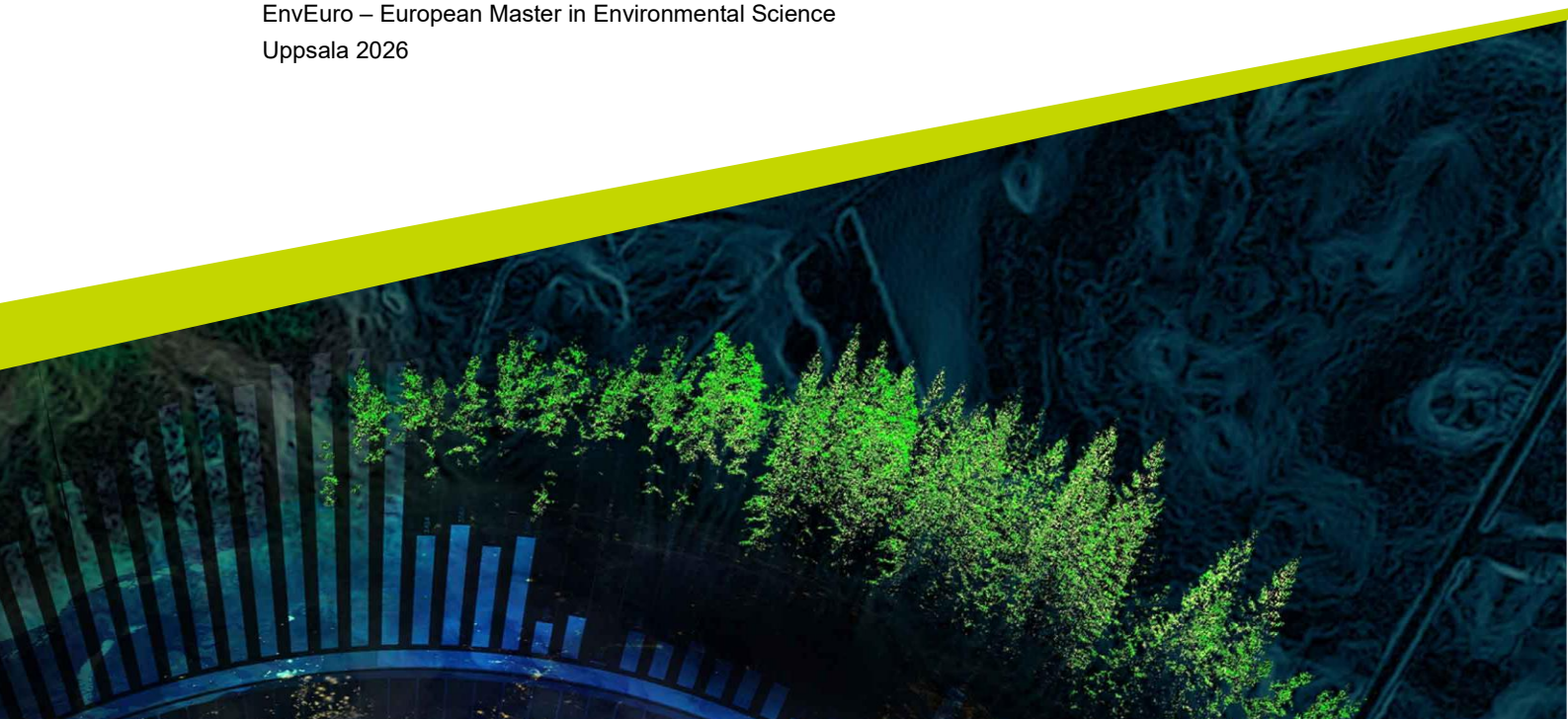

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Abstract

Mercury (Hg) is a neurotoxic heavy metal that can be taken up by vegetation from the atmosphere. Peatlands, formed by decomposed vegetation, store significant amounts of Hg, part of which is transported to downstream aquatic ecosystems. Peatlands also serve as a natural archive of atmospheric Hg concentrations in a chronological order, even though post-deposition processes can mobilize Hg from the peat. Understanding the fate of mercury in peatlands is important for predicting how ecosystems respond to changing atmospheric Hg concentrations. In this study, Hg concentrations in eight peat cores collected in 2025 from Degerö Stormyr, Sweden, were analyzed and compared with those in peat cores sampled in 2015 and 2020. We found that the Hg pool decreased over the last decade, and that different peat cores can exhibit very distinct biogeochemical properties even if sampled from the same site, separated by only a few meters. The vegetation growing on the peatland also showed variable Hg concentrations depending on species and the parts of plants analyzed. Our findings of large intra-variability in Hg levels highlight the importance of investigating multiple sites and cores to monitor Hg concentrations within the same peatland and of analyzing several cores from the same sites. It is also significant to include an analysis of plants, both surface vegetation and of plant macrofossils in the peat. They are to ensure a comprehensive investigation of Hg mobility in peatlands and a robust evaluation of temporal changes in atmospheric Hg levels using peat records.

Keywords: mercury, peatlands, vegetation, mobility

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Abbreviations

Abbreviation	Description
EPA	Environmental Protection Agency
SLU	Swedish University of Agricultural Sciences
UNEP	United Nations Environment Programme
WHO	World Health Organization

1. Introduction

Mercury (Hg) is a toxic heavy metal, considered by the Worl Health Organization (WHO) to be one of the top ten chemicals of major public health concern (WHO, 2024). Being exposed to this element can result in health impairments such as acute damage to the nervous system, as well as disorders of the immune and reproductive systems or kidney and heart diseases (Ashar *et al*., 2025).

1.1 Sources of mercury in the environment

Mercury is a trace element naturally present in the Earth's crust. The most common mercury mineral and most stable mercury compound is cinnabar (HgS) (Kozin and Hansen, 2013). Mercury from the Earth's crust can be released to the atmosphere as a result of volcanic eruptions, which are the only natural source of Hg in the free troposphere and stratosphere (Pyle and Mather, 2003). Other natural sources include wildfires, which release deposited Hg from soils and vegetation (Francisco López, Heckenauer Barrón and Bello Bugallo, 2022). In addition to natural sources, recent anthropogenic activities also contribute substantial amounts of mercury to the atmosphere, including fossil fuel combustion, particularly coal burning, which causes higher emissions of Hg than the burning of both oil and natural gas. Moreover, the process of coal washing also releases mercury to the aquatic environment (Charvát *et al*., 2020). Another significant source of this substance is artisanal and small-scale gold mining, in which mercury is used to extract gold from the ore as a stable amalgam. After the extraction, the gold is isolated by heating the amalgam and evaporating the mercury (Esdaile and Chalker, 2018). According to the United Nations Environment Programme, this process is responsible for as much as 37 % of anthropogenic emissions of Hg into the environment, while 24 % of the emissions are a result of coal combustion (United Nations Environment Programme, 2013). With the aim of reducing mercury pollution worldwide, the Minamata Convention was signed by representatives of over 140 countries in 2013. The convention requires that its parties reduce or phase out the usage of mercury from several industrial activities and products, such as batteries, cosmetics, pesticides, lights or the production of dental amalgam, vinyl chloride monomer, and acetaldehyde (U.S. Environmental Protection Agency (EPA), 2026).

1.2 The biogeochemical cycle of mercury

Mercury can be found in the environment in three oxidation states: Hg^0 referred to as elemental mercury, Hg^+ (mercurous), and Hg^{2+} (mercuric) (Figure 1). In the atmosphere, mercury mainly exists in the Hg^0 form (O'Connor *et al*., 2019),

which can account for as much as 90 % of total Hg (Valente *et al*., 2007; Lyman and Gustin, 2008). Hg^0 has an atmospheric lifetime against oxidation to Hg^{2+} (mainly by Br-atoms) of ca. 2.7 months (Horowitz *et al*., 2017). Oxidation in the atmosphere can be also caused by reactions with O_3, OH, and NO_3 (Gonzalez-Raymat *et al*., 2017). Hg^0 can also be taken up by vegetation, where it is oxidized to Hg^{2+} by various enzymatic reactions, thereby becoming biochemically fixed (Liu *et al*., 2022). Hg^0 is the most important source of Hg deposited in the soils of the Arctic tundra. The deposition occurs all year round, with enhancements in the summer driven by vegetation uptake (Obrist *et al*., 2017). An analysis of $\Delta^{200}Hg$ isotope data suggested that ocean Hg^0 uptake is more significant than previously thought and can be as high as the deposition of Hg^{2+} into the ocean (Jiskra *et al*., 2021). Hg^+ is unstable in environmental conditions and forms Hg^{2+} ions, whereas Hg^{2+} tends to be removed from the atmosphere by deposition into soil or ocean or reduced in the atmosphere by e.g. photoreduction (Horowitz *et al*., 2017). Hg^{2+} deposited into soils can form inorganic Hg (II) compounds, such as $HgCl_2$ and $Hg(OH)_2$ (EPA, 1997).

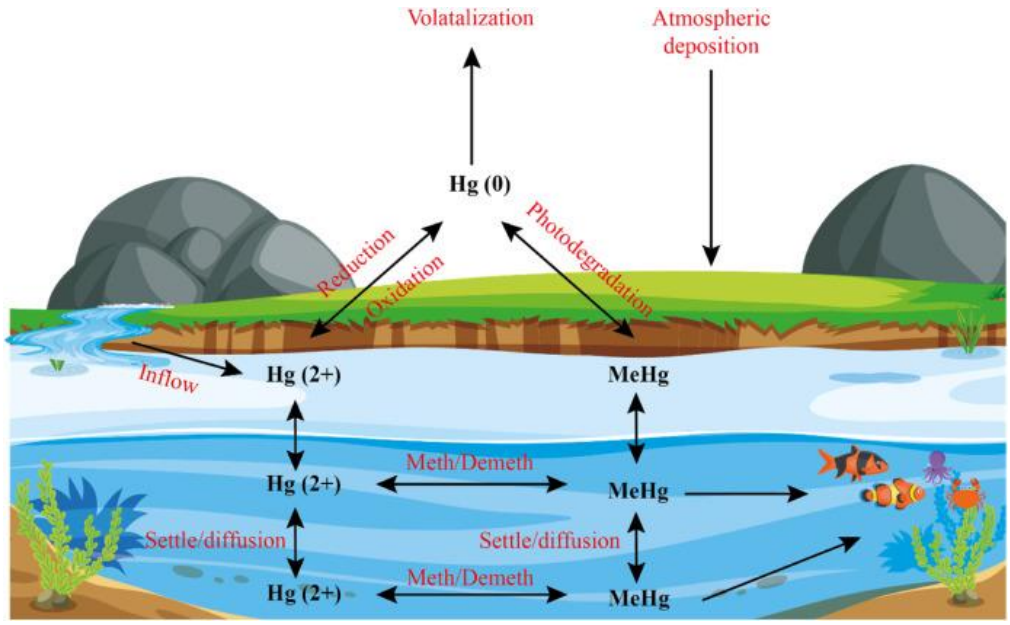

Figure 1. The geochemical cycle of mercury. (Pavithra et al., 2023).

The most toxic species of mercury is its methylated form (CH_3Hg^+) (Gonzalez-Raymat *et al.*, 2017). This compound is synthesized by microorganisms that carry the hgcA and hgcB genes (Parks *et al.*, 2013). Mercury can also be methylated via the abiotic pathway, e.g. reaction with methyl iodide (CH_3I) after this compound dissociates into CH_3^+ and I^- (Gonzalez-Raymat *et al.*, 2017). Peatlands accumulate Hg due to vegetation Hg^0 uptake and Hg^{2+} rain deposition. The methylation of the accumulated mercury is then facilitated by the availability of electron acceptors like sulphate and by anoxic conditions (Haynes *et al.*, 2017). Methyl-Hg is

exported with runoff into watercourses and lakes, where it enters the food chain and undergoes biomagnification (Lavoie *et al.*, 2013). It has been observed that the deposition of sulphate to peatlands and lakes causes an increase in the levels of MeHg, because several sulphate reducing bacteria are capable of methylating Hg (Strickman *et al.*, 2016). Hu *et al*., (2020) showed that sulphate reduction is indeed the main driver of Hg methylation in young peatlands, but methanogenic and syntrophic organisms are more important in older systems with a lower pH and nutrient availability (Hu *et al.*, 2020) Interestingly, also non-Hg methylating microorganisms can have an influence on the formation of MeHg, because they can modulate the bioavailability of Hg that can undergo methylation and also have an impact on the population of the microbes that actually methylate MeHg. Some of them also reduce MeHg levels during demethylation processes (Wang *et al.*, 2023).

1.3 Mercury in aquatic ecosystems

As a result of MeHg being present in the waters downstream of peatlands, many species of freshwater fish in the Nordic countries have high Hg concentrations that exceed EU-guidelines (Munthe *et al*., 2007). Organisms in high latitudes also appear to be more prone to Hg biomagnification than those in tropical regions (Lavoie *et al*., 2013), possibly because they have lower growth rates than organisms in lower latitudes and therefore have higher Hg concentrations per unit of body mass (Simoneau *et al*., 2005). Other possible explanations include the fact that elimination of MeHg in fish is positively correlated with temperature (Trudel and Rasmussen, 1997). Food webs are also less diverse in higher latitudes, which could increase the efficiency of trophic Hg transfer, because consumers have a narrower choice of prey (Lavoie *et al*., 2013). Moreover, MeHg is eliminated from fish at a lower rate than inorganic Hg (Trudel and Rasmussen, 1997), while the trophic transfer efficiency from food is significantly higher in the case of MeHg than inorganic Hg. One of the reasons for this is the fact that zooplankton consumed by fish assimilates MeHg from the cell cytoplasm of phytoplankton organisms it consumes four times more efficiently than it assimilates inorganic Hg from phytoplankton membranes (Mason, Reinfelder and Morel, 1996). Hence, despite the high nutritional value of fish meat, fish consumption on a regular basis might result in adverse health effects arising from MeHg and, as a consequence, serious diseases (Dragan *et al*., 2022). Åkerblom *et al*., (2014) found that fish Hg concentrations in Swedish waters have been declining since 1970 by ca 1 % per year, but still remain high in certain regions and health advisory guidelines are in effect across Fenno-Scandia (Åkerblom *et al*., 2014). Braaten *et al*., (2019) also found a decreasing trend in Hg concentrations in fish in boreal and subarctic lakes in the years 1965-2015, but pointed out that the Hg concentrations still exceed the

Water Frame Directive Environmental Quality Standard of 0.02 ppm, which is set to protect wildlife, with the highest concentrations being observed in the predatory pike (Braaten *et al*., 2019). The consumption of freshwater fish has been reported to cause an increase in hair mercury levels of members of an angling society in Hagfors, Sweden (Johnsson *et al*., 2004). The Swedish National Food Agency (Livsmedelsverket) advises pregnant women to consume the following species of freshwater fish not more frequently than 2-3 times per year to avoid hazardous uptake of mercury: pike (*Esox lucius*), perch (*Perca fluviatilis*), and pikeperch (*Sander lucioperca*) (Swedish National Food Agency, 2008). It is desirable to reduce Hg levels in fish so that restricting consumption is unnecessary (Johnson *et al*., 2004).

1.4 Mercury in terrestrial ecosystems

In terrestrial ecosystems, vegetation uptake from the atmosphere is responsible for 60-90 % of Hg deposition. The uptake of Hg by vegetation reduces the deposition into the ocean. The atmospheric Hg captured by vegetation, especially wetlands, is still an important source of Hg for downstream freshwater organisms. This is because Hg can be transferred from soils to freshwater and coastal ecosystems. In vascular plants, Hg concentrations are the highest in leaves and in needles (Zhou *et al*., 2021). Foliar uptake is a more significant pathway of accumulation than surficial adsorption on leaves (Liu *et al*., 2022). Given that they take up Hg from the atmosphere, these parts of plants are highly sensitive to fluctuations in Hg atmospheric concentrations (Zhou *et al*., 2021). A study conducted in Hebei Province, China, demonstrated that Hg concentrations in foliage of maize and wheat are significantly correlated with Hg concentrations in air, whereas Hg concentrations in roots are significantly correlated with the concentrations of mercury in soil (Niu *et al*., 2011). Non-vascular vegetation, like mosses and lichens, tends to show significantly higher Hg concentration than vascular plants (Zhou *et al*., 2021). Lichens can also be considered as good biomonitors for analyzing contaminated regions (López Berdonces *et al*., 2017).

Mercury stored in peatlands is of global significance, given that Hg has a strong affinity for organic matter, and peatland ecosystems store 15-30 % of terrestrial organic carbon. Apart from Hg^0 taken up by vegetation, they also accumulate a less significant amount of Hg^{2+} that is deposited from rain. Part of the deposited Hg is reduced to Hg^0 and released back into the atmosphere, which likely contributes to the reduction of bioavailable Hg for MeHg production (Li *et al*., 2023).

1.5 Peatland ecosystems as an archive of atmospheric Hg concentrations

Despite a proportion of deposited Hg returning to the atmosphere, peatlands still remain a valuable natural archive of atmospheric Hg concentrations, as they preserve Hg deposited in a chronological order and can be used to investigate the response of ecosystems to fluctuating atmospheric Hg concentrations. A peat core from Estibere, Spain, showed an increase in the Hg accumulation rates in the Middle Ages and preindustrial times, and a decrease in the years 1990-2011 (Enrico *et al*., 2017).

Interpreting Hg records in peatlands is, however, not straightforward, because the concentration of Hg in peat is also affected by the post-deposition processes, in addition to the atmospheric Hg concentration (Li *et al*., 2025). For instance, the Hg concentrations in peat increase when peat undergoes mass loss during humification. Hence, variations in mercury concentrations in peat are not always a result of changes in Hg supply rates from the atmosphere (Biester *et al*., 2003).

However, even before one considers post-depositional processes, it is important to know what was deposited in the first place. A key factor affecting the Hg deposition rate in peat may be the vegetation type and plant species composition at the surface. Li *et al*., 2023, found that hummocks have higher Hg accumulation rates than lawns, even if they are as close as 5 m apart from each other. The average Hg concentration in surface vegetation is also higher on hummocks than on lawns (Li *et al*., 2023). A shift in the vegetation composition as well as changes in the hydrological regime can increase the accumulation of mercury in peat and enhance its transport downstream (Haynes *et al*., 2017). This is one of the reasons why mercury concentrations and mobility in peatlands might be impacted by climate change.

Since Hg accumulation in peat varies with shifts in the vegetation composition, it is important to integrate botanical evidence from the surface and macrofossils from the peat core with Hg concentrations in different species of plants, and to determine which shifts in Hg accumulation were caused by a change in vegetation composition and which by fluctuating Hg concentrations in the atmosphere. Another factor that impacts Hg accumulation rates can be variations in primary productivity (Li *et al*., 2025).

Osterwalder *et al*., (2017) also found that during the period June 2013 – June 2014, Degerö Stormyr peatland in Northern Sweden had a net atmospheric evasion of mercury that exceeded rainfall inputs of Hg. They assumed that Hg evasion from the surface of the peatland increased when atmospheric Hg concentrations above Europe were halved between ca. 1980 and 2000. Should this trend be sustained, the Hg pool in the peatland will decrease at a much faster

rate than previously anticipated, and the transport of Hg to the downstream aquatic ecosystem will also decline (Osterwalder et al., 2017).

However, isotope analyses of Li *et al*., (2023) suggested that Hg^0 does not diffuse to the atmosphere via soil pores in the peat, and no evidence of Hg^0 being produced from Hg^{2+} stored below the water table in the peat by a process called dark microbial reduction was found either. Instead, Hg^{2+} from rainfall likely was the main source of Hg^0 in the groundwater. When rainwater resides at the surface of the peatland, it is exposed to sunlight, which enables photoreduction of Hg^{2+} to Hg^0. The largest source of Hg^0 input to the Degerö Stormyr peatland, however, is the direct uptake of atmospheric Hg^0 by plants. This pool of Hg, after being oxidized to Hg^{2+}, is less available to photoreduction and evasion than Hg^{2+} deposited by rain, because it is bound to the plant tissues (Li *et al*., 2023).

1.6 Research questions

The aim of this thesis is to analyze the Hg concentration in eight peat cores from Degerö Stormyr sampled in August 2025 in order to compare it to data from eight peat cores collected in 2015. The results will contribute to our understanding of the fate of mercury after deposition to peat. The thesis aims to answer the following research questions:

- Previous research predicted an evasion of Hg from Degerö Stormyr based on one year of exchange measurements (June 2013 – June 2014, Osterwalder *et al*., 2017). Did the Hg pool in the peat decrease between 2015 and 2025?

- Could shifts in surface vegetation composition influence the input of Hg in the peat?

2. Materials and methods

2.1 Site description

The peat samples analysed in this thesis were collected from the Degerö Stormyr mire complex (Figure 2) (64°11'N, 19°33'E), located in the county of Västerbotten in the north east of Sweden. The mire is located at an elevation of 270 m a.s.l. between the rivers Umeälven and Vindelälven. The closest town to the site is Vindeln. The distance from the Gulf of Bothnia is ca. 70 km (Nilsson *et al*., 2008). The catchment spans an area of 1.9 km^2. 70 % of the soils consist of peat and 30 % of mineral soil (glacial till) (Osterwalder *et al*., 2017). The deepest peat layers reach an age of ca. 5,700 years. The peat can be as deep as 8 m, although it most often reaches the depth of 3-4 m (Nilsson *et al*., 2008). The bedrock consists mainly of gneiss. The mire is rather poor in nutrients, which is reflected by the plant communities that are dominated by sphagnum mosses and grasses (Swedish University of Agricultural Sciences (SLU), 2025). The climate of the site can be described as cold humid, with a 60 cm deep snow cover lasting for approximately 6 months (Nilsson *et al*., 2008). This site is one of the most extensively studied mires in the entire Northern Hemisphere (SLU, 2025).

Figure 2. A view of Degerö Stormyr. © Jacob Smeds.

2.2 Sampling of the peat cores

Eight cores, each of 40 cm, were sampled by Jacob Smeds in the summer of 2025. They were sampled from four sites from Degerö Stormyr: North, East, South, and West, with a 50 m radius between them. A circular stainless steel corer was used to pre-drill the surface peat to facilitate the insertion of 15.1 ⌀ cm PVC plastic tubes (Figure 3). The corer and tubes were always cleaned before reuse. Two cores were sampled from each site. After sampling, the cores were sealed by watertight caps with the purpose of preventing water leakage. The cores were held upright during transport to the laboratory. They were stored in a -18 °C freezer and subsequently sliced into 2 cm discs using a bandsaw with a stainless-steel blade. This yielded 160 samples (8 cores, 20 samples each). The samples were placed in plastic bags and air-dried at 60 °C for 96 h. Air-drying can result in a median Hg deficit of 4.2 % (Smeds *et al*., 2022). Even though part of the Hg might have escaped from the peat during oven-drying, a median loss of 4.2 % as was found by Smeds *et al*., (2022) would not have a significant impact on the results of the study.

Figure 3. Corer (background) and PVC tube (foreground). © Jacob Smeds.

The data from the 2025 cores were compared with those from eight 34 cm cores sampled in Autumn 2015 from the same peatland by Stefan Åkerblom. Each 2025

core was collected within 10 meters of one of the cores sampled in 2015, which ensures a high degree of comparability. All cores sampled in both 2025 and 2015 come from lawn sites. The 2015 cores were cut in 2 cm increments and freeze-dried for five days (Osterwalder *et al*., 2017).

A 100 cm core sampled in 2020 was also used for comparison. However, it was sampled ca. 200 m away from the 2025 and 2015 cores. The data from 2020 come from the Supplementary Information of Li *et al*., 2023.

2.3 Grinding of the peat samples

The 2025 samples were pre-homogenized to enable subsampling for milling. An IKA Tube Mill Control (IKA, Staufen, Germany) with single-use milling chambers was then used to obtain a fine powder. However, some of the samples did not pass the replicate check. Therefore, all the samples were re-grinded with the Mixer Mill MM 400 manufactured by Retsch GmgH. Each sample was ground for two minutes at a frequency of 25 s^{-1}. This procedure ensured that the samples passed the replicate check and narrowed the results' standard deviation.

2.4 Hg concentration analysis

The Hg concentration was determined with a Milestone Direct Mercury Analyzer (DMA) 80 (Figure 4). Dried apple leaves (NIST 1515) and the lichen *Pseudevernia furfuracea* (BCR-482) were used for calibration. The analysis was run by Andreas Gulde in the Mercury Laboratory of the Department of Aquatic Sciences and Assessment at SLU, Uppsala. The DMA-80 operates on the principle of thermal decomposition, catalytic conversion, amalgamation, and atomic adsorption spectrophotometry. The samples are introduced into a quartz tube and subsequently dried and thermally decomposed. The decomposition products are carried by a continuous flow of oxygen through a hot catalyst bed which traps halogens, nitrogen and sulphur oxides. All species of Hg undergo reduction to Hg^0 and are transported with reaction gases to a gold amalgamator which traps mercury selectively, while the continuous flow of gas flushes out all non-mercury vapors and products of decomposition. Subsequently, the amalgamator is heated and all trapped Hg is released to the double beam, fixed wavelength atomic absorption spectrophotometer, where the absorbance is measured as a function of Hg content at a wavelength of 253.7 nm (Milestone Srl, 2019).

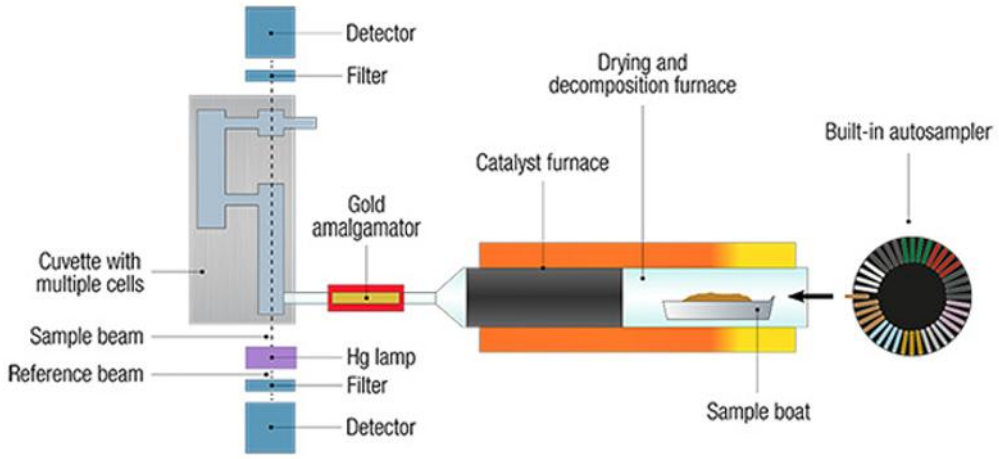

Figure 4. The structure of DMA-80. (Milestone Srl, 2019).

For the peat samples collected in 2015, certified reference lake sediment (IAEA SL1 [130 ng g^{-1}] and pine needle material (PINE1575a [40 ng g^{-1}] were used for calibration. The Hg content was analyzed with a SMS100 (Perkin Elmer, Waltham, USA) through thermal decomposition atomic absorption spectrometry according to EPA method 7473 (Osterwalder *et al*., 2017).

2.5 Accuracy of the Certified Reference Materials ®

NIST 1515

A unit of NIST 1515 consists of 50 g of dried apple leaves. The certified value is 0.0432 ± 0.0023 mg kg^{-1} (NIST, 2022). All the measured values were slightly higher, which means the recovery rate was above 100 % (Figure 5). The mean recovery rate was 106.7 %, and all the measured values were within the 100-115 % interval, which is considered acceptable.

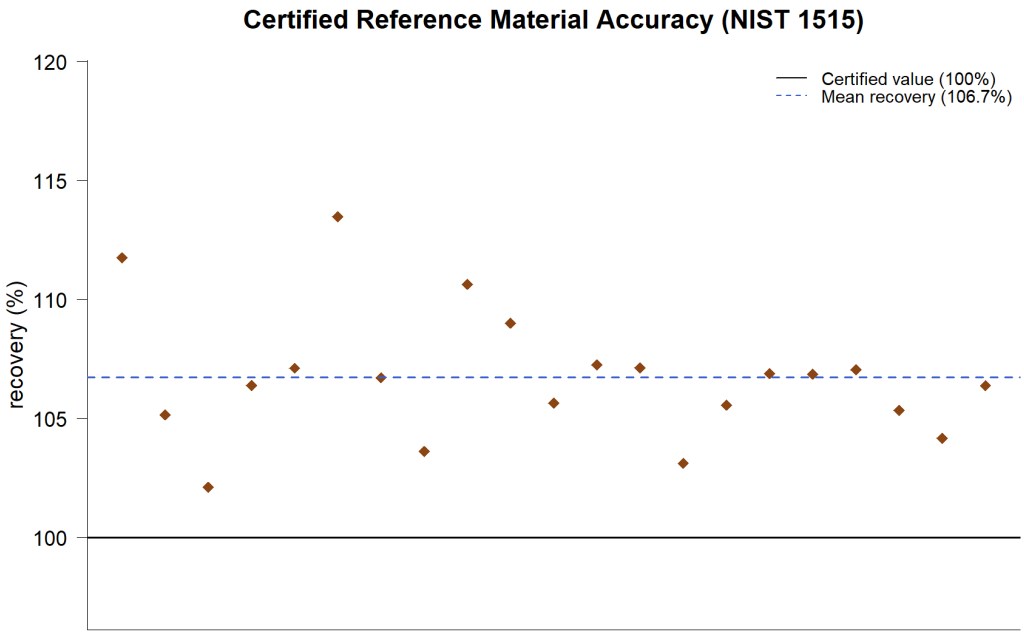

Figure 5. Accuracy of the Certified Reference Material ® (NIST 1515).

BCR-482

The BCR-482 Certified Reference Material ® is prepared from the lichen *Pseudevernia furfuracea*. The certified value is 0.48 mg kg^{-1} ± 0.02 mg kg^{-1} (Quevauviller, Herzig and Muntau, 1996). All the measured values are within the 95-105 % interval (Figure 6), which is considered acceptable.

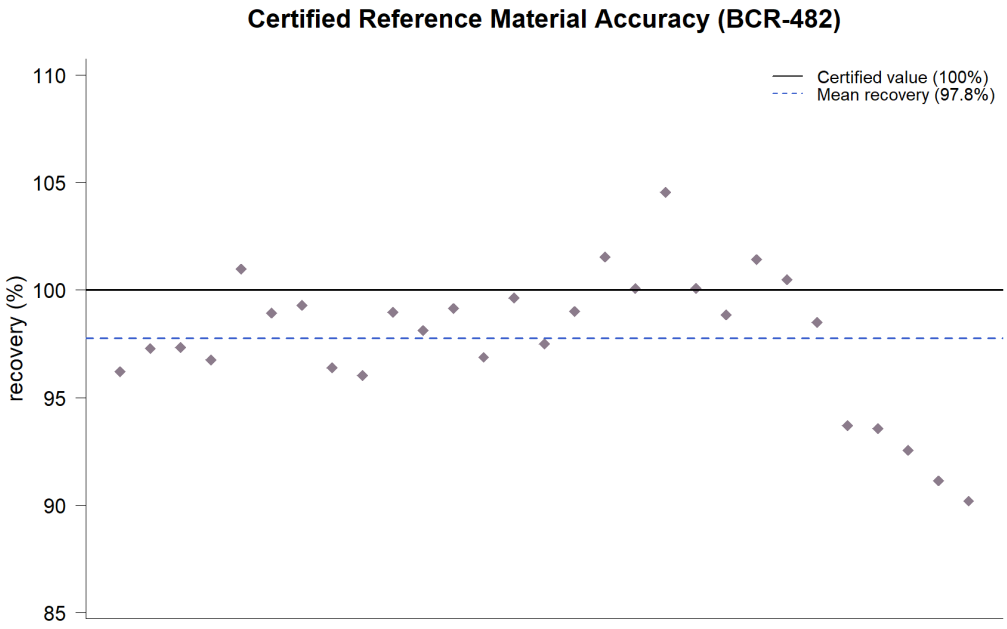

Figure 6. Accuracy of the Certified Reference Material (BCR-482).

The mean recovery value of all the measurements is 101.53 %, which is very close to 100 %.

2.6 Vegetation sampling

The vegetation samples were collected by Kevin Bishop during summer 2025 from sites S10, S18, S26, and S70. These sites are located ca. 50 km southeast of Degerö Stormyr. The Hg concentrations were compared with concentrations in vegetation sampled from Degerö Stormyr in 2020. The 2020 data come from the Supplementary Information of Li *et al*., (2023).

2.7 Statistical analysis

The statistical analysis was conducted with RStudio, Version 2026.01.1+403. Pairs of peat profiles from the same sites and the same sampling years were compared to each other with a paired Wilcoxon test. The same test was applied to compare average profiles from the same sites and different sampling years. Hg concentration profiles, peat dry bulk density profiles, and Hg inventory profiles were used for comparison. Since 34 cm cores were available from 2015, only the uppermost 34 cm of the 40 cm 2025 cores were used for the statistical tests.

The Hg inventory (μg m^{-2}) in each core was calculated using the following formula:

$$\frac{Hg\ concentration\ (ng\ g^{-1})}{1000} * \ dry\ bulk\ density\ (g\ m^{-3}) * 2 * 10{,}000$$

3. Results

3.1 Hg concentrations in the peat samples

There is a general increase in Hg concentration with the depth in the peat profiles (Figure 7). The increase is the strongest in the middle part of the core. The variation in concentration is most likely due to peat decomposing with time and losing up to 90 % of its initial organic matter, (Li *et al*., 2025), while the amount of Hg remains the same. Therefore, there is more Hg per unit of peat in the lower part of the profile where the peat has undergone decomposition for a longer period of time.

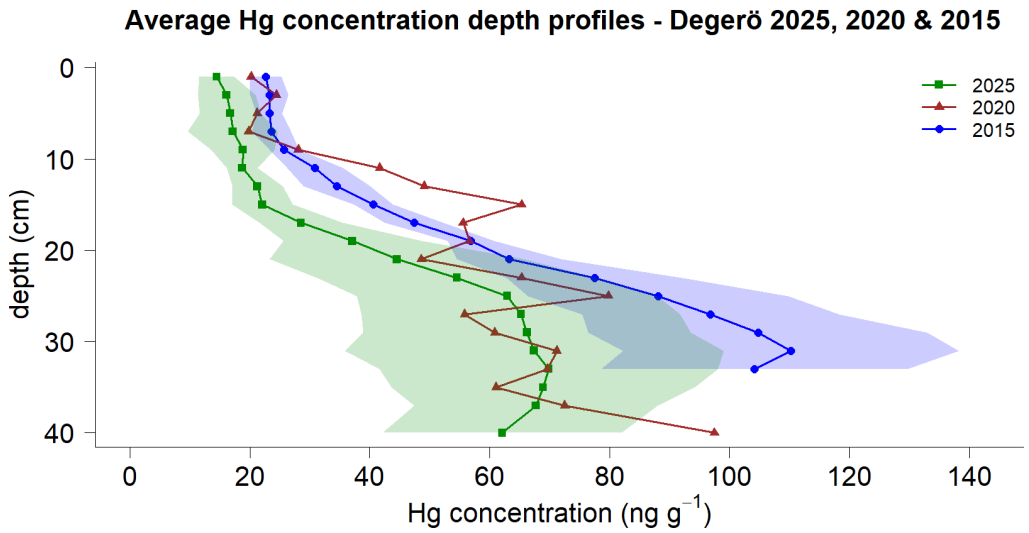

Figure 7. Average (± SD) Hg concentration in the 2025, 2020, and 2015 peat cores[1].

Average Hg concentrations in the peat profile are significantly lower in 2025 compared to 2015. The values from 2015 partially overlap with those from 2020, and so do values from 2020 with those from 2025. The higher standard deviation in the lower section of the 2015 and 2025 cores indicates a greater heterogeneity in the deeper peat layers. A paired Wilcoxon test yielded a p-value < 0.001, indicating a significant difference in the Hg concentrations between 2015 and 2025.

[1] Data from 2015 and 2025 come from 8 peat cores, whereas data from 2020 come from one peat core. The measurements in 2020 were taken at different depth intervals than in 2015 and 2025, so the average of the measurements taken on each 2 cm interval of the 2020 core was used. No standard deviation is presented for 2020 data, since the measurements come from one core. Only the uppermost 40 cm of the 2020 core are presented.

When the 2025 concentration profiles of each core are analyzed separately, the high variability between them is evident (Figure 8). The biggest difference occurs in Site North, where the Hg concentrations are quite close to each other at the top of the two profiles, but very distinct in the lowermost half. The differences between the concentration profiles are less striking at the other sites, but the two profiles remain quite distinct in the lowermost part of the cores from Site West and in the central part of those from Site East. The Hg concentrations in the two profiles in Site South overlap the most. A paired Wilcoxon test was conducted between the pairs of peat cores from each site sampled in the same year. The obtained p-values for the pairs of 2025 cores were consistently below 0.05, confirming a significant difference in the concentration profiles. However, the test for the pairs of 2015 cores yielded p-values greater than 0.05, indicating that the Hg concentration profiles of cores sampled at the same site in 2015 do not differ significantly.

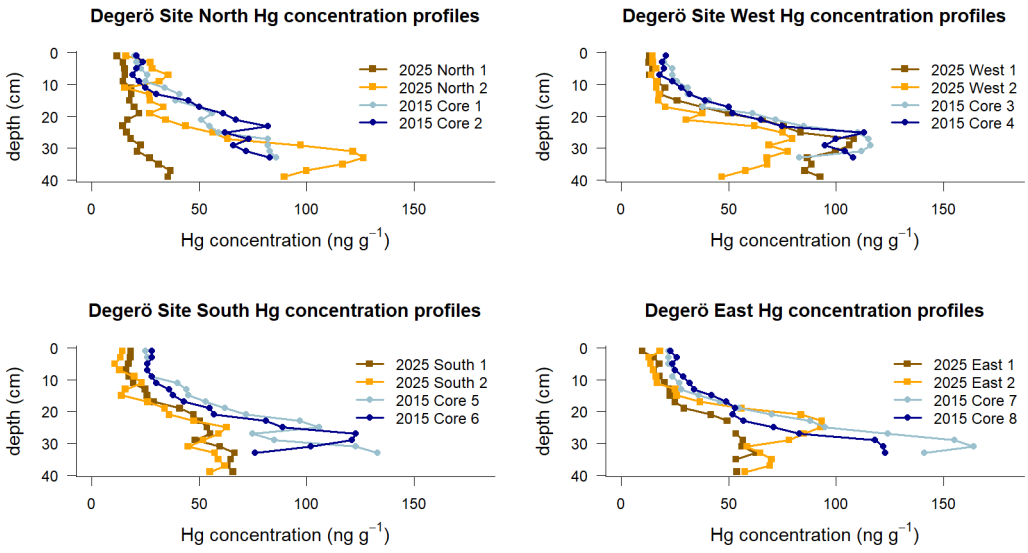

Figure 8. Hg concentration depth profiles of 2025 and 2015 peat cores of Degerö Stormyr.[2]

A comparison of Hg concentrations in the four sites of Degerö Stormyr also shows a decrease in the Hg pool between 2015 and 2025. Each site shows a lower Hg concentration in 2025 than in 2015. The difference is especially evident at the Southern Site, where the average measured Hg concentration was higher in 2015 than in 2025 across the profile (Figure 8). The values from 2015 and 2025 also differ at the Western Site, but Hg concentrations of Core West 1 converge in the middle of the profile with concentrations of Core 5 and Core 6 from 2015. In the Northern Site, the concentrations slightly overlap in the uppermost part. Core

[2] Intervals 24-26 cm and 36-38 cm missing from core 2025 East 1 due to wrong labelling of the samples.

North 2 shows a strong increase around 30 cm depth, exceeding the concentrations from both 2015 cores. It is, however, unknown what the 2015 concentration profiles look like below 34 cm. Core North 1 has a lower Hg concentration than both 2015 cores throughout the profile. Core East 2 has Hg concentrations that overlap in the middle intervals with the Hg concentrations of the two 2015 cores. Despite these small overlaps, the overall concentration was higher in 2015 than in 2025 in all four sites. A significant difference between the average Hg concentration profiles from both years is confirmed by the results of a paired Wilcoxon test, which yielded p-values < 0.001 at each site.

3.2 Dry bulk density of the peat

The dry bulk density of the peat cores also highlights the intra-variability of the peatland. On average, the dry bulk density increases from 0.009 g m^{-3} at the top of the cores to 0.07 g m^{-3} at the bottom (Figure 9). The standard deviation is higher in the lowermost part of the cores. This shows that the values differ more between cores at the bottom than at the top. The average dry bulk density of the 2025 cores overlaps to a certain extent with that of the 2015 cores. A paired Wilcoxon test yielded a p-value > 0.05, indicating no significant difference between the average dry bulk densities of the peat cores sampled in 2015 and 2025. The lawn core from 2020, however, has a significantly higher density in its uppermost part. Its density values begin to overlap with those of the 2015 and 2025 cores in the middle of the profile. This core, however, was sampled ca. 200 m from the sites from which the 2015 and 2025 cores were obtained.

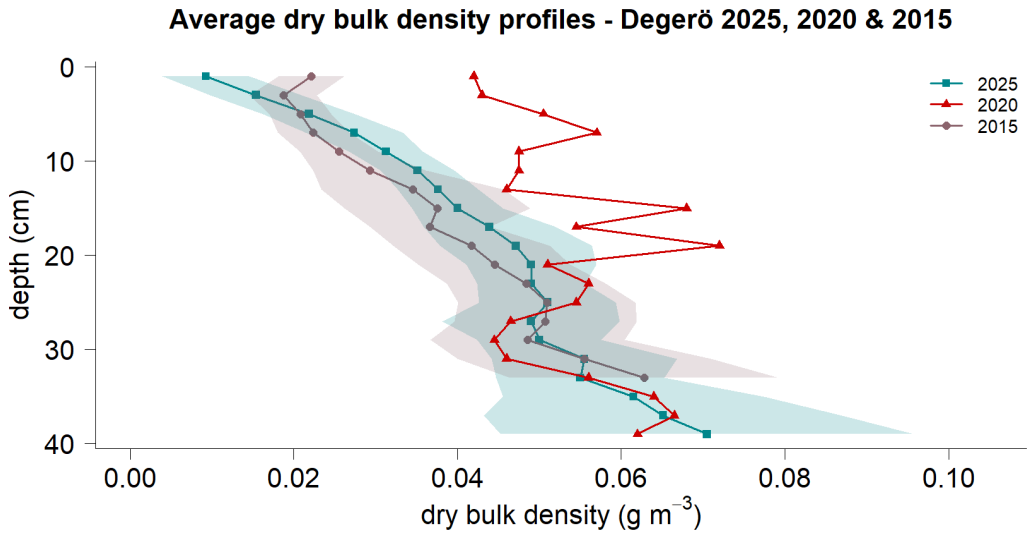

Figure 9. Average (± SD) dry bulk density profiles of the 2025, 2020, and 2015 peat cores.[3]

Additionally, for the dry bulk density of the 2025 cores, the largest difference occurs between the two cores from Site North (Figure 10). This is especially evident in the lowermost part of the cores, where the density of core North 2 increases sharply, while that of core North 1 remains nearly constant. The densities of the other pairs of cores are also more distinct from each other closer to the bottom of the cores than at the top. For instance, the density of core West 1 increases by a very strong amount in the lowermost 10 cm of the core, whereas it is not evident in core West 2. A paired Wilcoxon test was also conducted for each pair of cores. The p-value for the 2025 cores from site West was 0.21, indicating that the sudden increase in the density of core West 1 is insufficient to make the two cores statistically different. For the other sites, the test for 2025 cores always yielded a p-value < 0.01. For the 2015 cores, the Wilcoxon test yielded a p-value higher than 0.05 for sites South and West, although the value for site South was very close to 0.05. The test for pairs of cores from sites North and East yielded p-values significantly less than 0.05, confirming a significant difference between them.

[3] Data from 2015 and 2025 come from 8 peat cores, whereas data from 2020 come from one peat core. The measurements in 2020 were taken at different depth intervals than in 2015 and 2025, so the average of the measurements taken on each 2 cm interval of the 2020 core was used. No standard deviation is presented for 2020 data, since the measurements come from one core

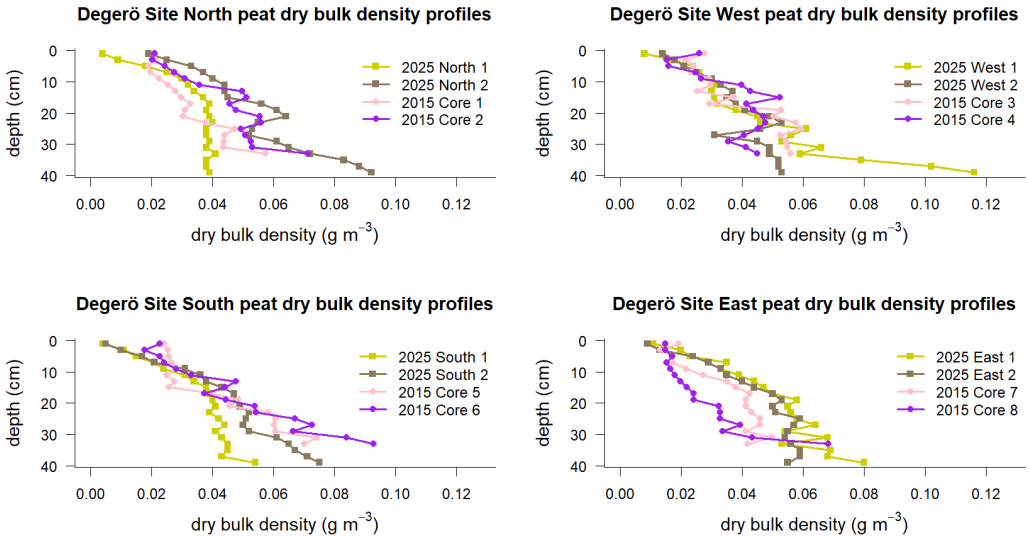

Figure 10. Dry bulk density depth profiles of the 2025 and 2015 peat cores.

It was also verified whether the dry bulk density differs between the cores from the 2025 and 2015 field campaigns, as this parameter influences the calculated Hg inventory. The only site in which a higher bulk density was measured in 2015 than in 2025 was Site South, where the bulk density of the two cores sampled in 2015 strongly increases in the lowermost 10 cm (Figure 10). The cores sampled in 2015 at the Eastern Site have a lower average dry bulk density than those sampled in 2025. At the Northern and Western sites, there is no significant difference in the average dry bulk density between cores sampled in 2015 and 2025. The paired Wilcoxon test also yielded p-values > 0.05 for these two sites. For the Southern and Eastern sites, the p-values were < 0.01, confirming a significant difference in average dry bulk density.

3.3 Hg inventory

The total average Hg inventory in the 2025 cores was calculated as 862 µg m^{-2}. The standard deviation is higher near the bottom of the cores, indicating greater internal variability there (Figure 11). Internal variability can also be observed when inventory profiles are analyzed separately (Figure 12).

A decrease in the Hg pool between 2025 and 2015 is evident in the average inventory analysis (Figure 11). The average inventory calculated for the 34 cm cores sampled in 2015 was 882 µg m^{-2}. However, the average inventory in the uppermost 34 cm of the 2025 cores was 594 µg m^{-2}. Hence, the inventory decreased by an average of 33 %. However, if Hg was mobilized, some of it might have moved to deeper parts of the profile. It is also possible that the peat in the 2025 cores was younger, thereby lowering the Hg inventory. Dating the peat would help determine its age and Hg accumulation rates. The 2020 core was

excluded from the inventory analysis because its higher dry bulk density would have negatively impacted the comparability of the data.

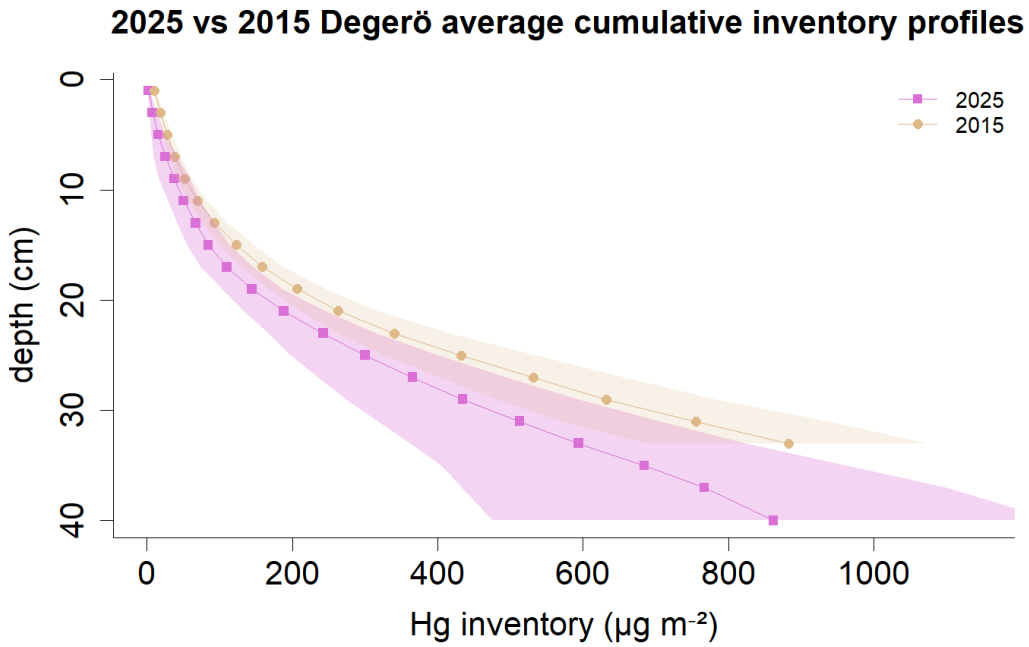

Figure 11. Average (± SD) cumulative Hg inventory depth profiles for 2025 and 2015 (n per year = 8).

At all sites, the pairs of cores have inventories that are close together in the uppermost parts of the cores and farther apart in the lower parts (Figure 12). The biggest difference is evident between the inventory profiles of the two 2025 cores from the Northern Site. Core North 2 has the largest inventory of all the cores, while core North 1 has the lowest. There is also a significant difference between the 2025 cores from the Western Site. The total Hg inventory of core West 1 is nearly as high as that of core North 2. Core West 2 has a larger total inventory than core North 1, though. Among the four sites, the inventory profiles of the 2025 cores from the Southern Site have the closest values.

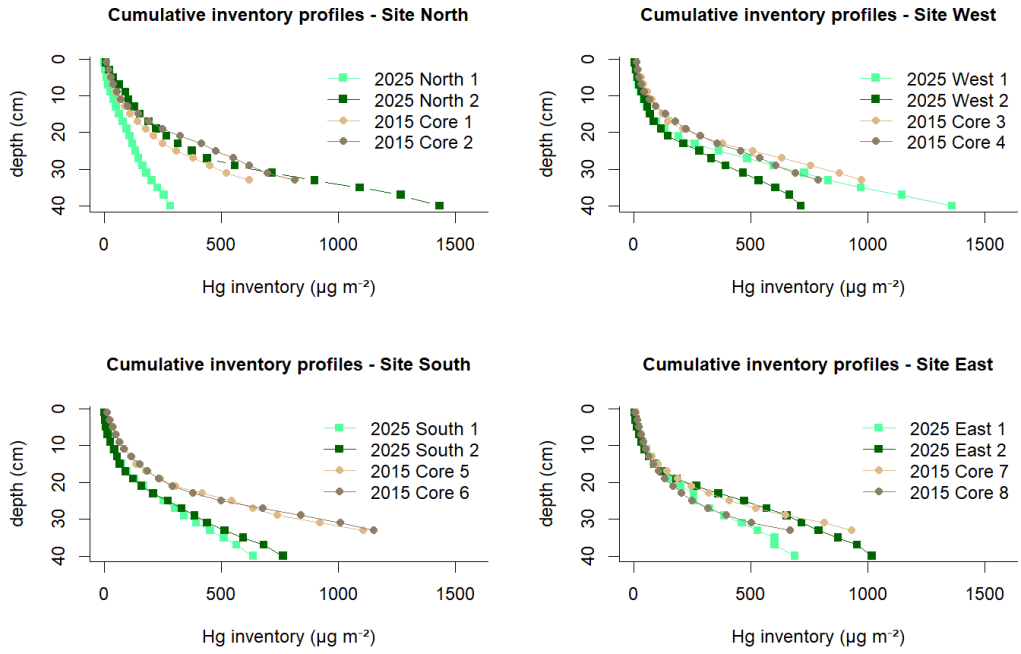

Figure 12. Cumulative Hg inventory depth profiles of the 2025 and 2015 peat cores.[4]

A paired Wilcoxon test was conducted between pairs of cumulative inventory profiles sampled in the same year. For pairs of both 2025 and 2015 cores, the p-value was less than 0.05 for the North, West, and East sites, indicating a significant difference between the profiles. The p-value was especially low for Site North. The test for Site South yielded a p-value of 0.15 for the 2025 cores, which can be tentatively interpreted as an indication of no significant difference in the inventory profiles of core South 1 and South 2. For the 2015 cores from Site South the value yielded by the Wilcoxon test was 0.85, showing that the inventory profiles of these cores are not significantly different from each other.

A comparison of the average Hg inventories in 2015 and 2025 in the four sites of Degerö Stormyr highlights the intra-variability of the peatland. In the Northern, Western, and Eastern sites, the average inventory decreased by ca. 20 % between 2015 and 2025, while the Southern Site shows a decrease of ca. 230 % (Figure 12). The reason for such different results is not only a lower measured Hg concentration in the Southern Site in 2025 than in 2015 (Figure 8), but also a lower measured dry bulk density (Figure 10). A paired Wilcoxon test comparing inventory profiles in 2015 and 2025 at each site yielded p-values < 0.01 for the Northern, Southern, and Western Sites. In the Eastern Site, the p-value was 0.6, indicating that the difference between the profiles is not significant.

[4] Two samples are missing from core 2025 East 1 due to wrong labelling of the samples, which influences the calculation of the cumulative inventory

3.4 Vegetation data

Different plant species have different Hg uptake capacities, which means that a shift in vegetation on the surface of a peatland is likely to be reflected in the Hg concentrations in peat (Li *et al*., 2023). Hence, it is important to integrate vegetation data into an analysis of peat Hg concentration, to understand how vegetation composition might have affected the Hg concentration in the analyzed peat.

3.4.1 Vegetation type

The Hg content in 13 plant species and one lichen genus was analyzed (Figure 13). Hg concentrations vary widely across species. The highest Hg concentration was found in the genus *Cladina*, which is the only analyzed lichen. Three species of the non-vascular *Sphagnum* were also analyzed, and they had higher Hg concentrations than most vascular plants. *S. majus* exceeded the concentrations of all vascular plants, while the concentrations in *S. balticum* and *S. fuscum* are very close to those of *Rubus chamaemorus*, *Calluna vulgaris*, and *Eriophorum vaginatum*, but exceed Hg concentrations in the remaining vascular plants. The plant species with the lowest mean Hg concentrations are *Menyanthes trifoliata* and *Carex limosa*. It is worth noting that different parts of plants were analyzed and this can have an impact on the measured Hg concentrations. Moreover, some of the plants grew on lawns, while others grew on hummocks. The sites from which the plants were sampled also differed in age (young: < 1000 y, intermediate: 1000-2000 y, old: > 2000 y). All these factors can influence the Hg concentrations.

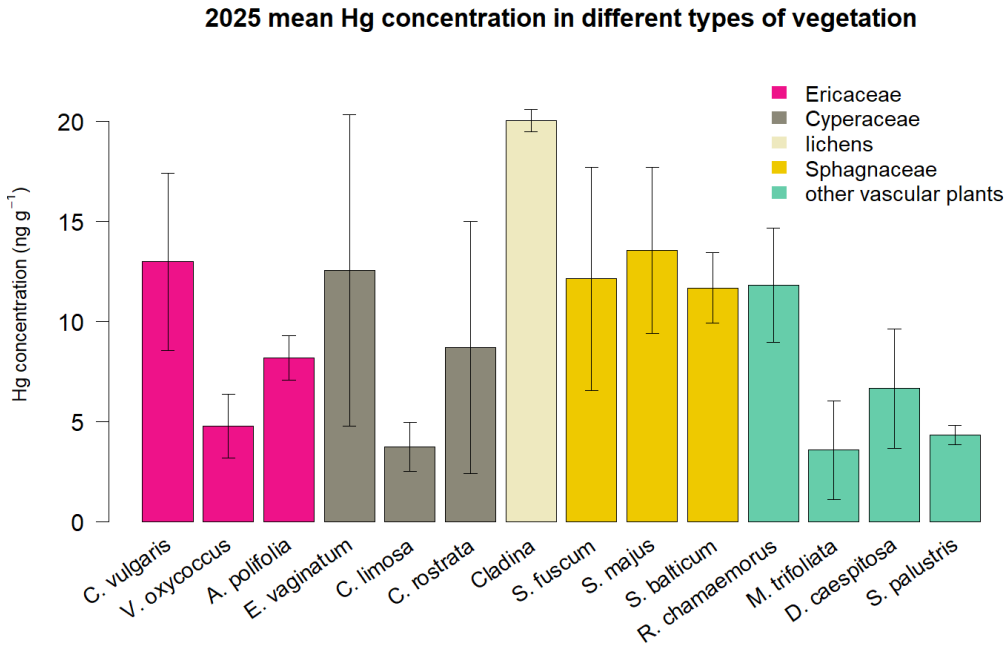

Figure 13. Mean Hg concentrations in different types of vegetation.[5]

3.4.2 Hg concentrations in lawns and hummocks

Hummocks tend to have higher Hg concentrations than lawns, even if they are not distant from each other (Li *et al*., 2023). This conclusion can also be drawn from the data used in this thesis (Figure 14).

[5] The error bar assigned to the genus Cladina does not present the standard deviation, because only one sample of this genus was analyzed. Instead, it depicts the analytical uncertainty of the CRM NIST 1515 (n = 21), which was used for analyzing the Hg concentration.

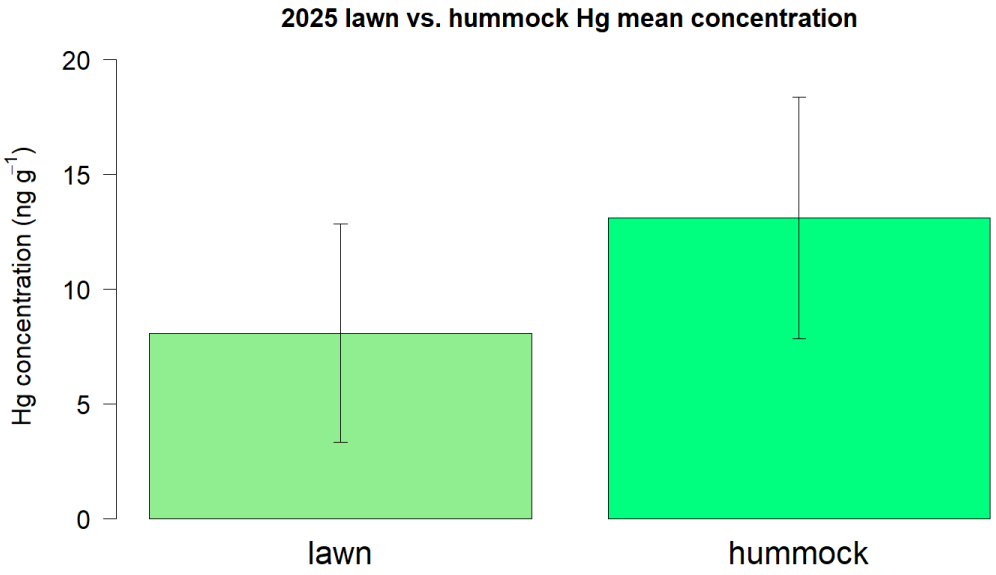

Figure 14. Mean Hg concentration (± SD) in the vegetation of lawns and hummocks.

Table 1. List of vegetation used for the analysis of the Hg concentration in lawns and hummocks.

Lawn vegetation	Hummock vegetation
Andromeda polifolia	*Calluna vulgaris*
Carex limosa	*Cladina*
Carex rostrata	*Eriophorum vaginatum*
Deschampia caespitosa	*Rubius chamaemorus*
Eriophorum vaginatum	*Sphagnum fuscum*
Menyanthes trifoliata	
Sphagnum balticum	
Sphagnum majus	
Scheuchzeria palustris	
Vaccinium oxycoccus	

The only species that was found in both lawns and hummocks is *Eriophorum vaginatum*.

3.4.3 Hg concentrations in different parts of plants

Vascular plants tend to have the highest Hg concentrations in leaves and needles (Zhou *et al*., 2021). Nine species of plants were used to compare the Hg concentration in leaves with the Hg concentration in roots. The mean leaf Hg concentration slightly exceeded the mean root Hg concentration (Figure 15).

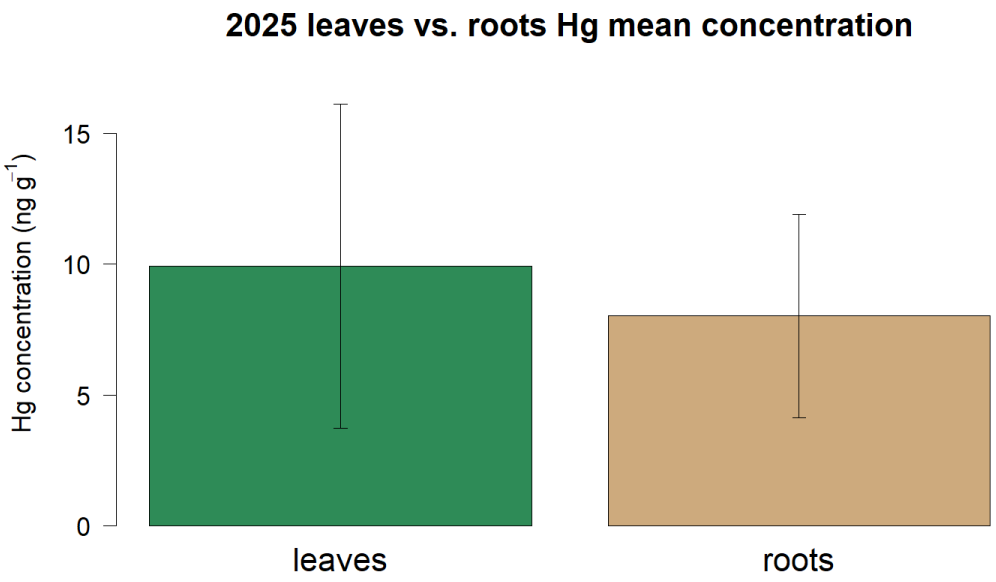

Figure 15. Mean Hg concentrations in leaves and roots of the analyzed plants. List of species used for analysis: Andromeda polifolia, Calluna vulgaris, Carex rostrata, Deschampia caespitosa, Eriophorum vaginatum, Menyanthes trifoliata, Rubus chamaemorus, Scheuchzeria palustris, Vaccinium oxycoccus.

The species were also analyzed separately. Most of them had a higher Hg concentration in leaves than in roots (Figure 16). However, the Hg concentration in the roots of *R. chamaemorus*, *A. polifolia*, and *D. caespitosa* exceeded that in the leaves of these plants.

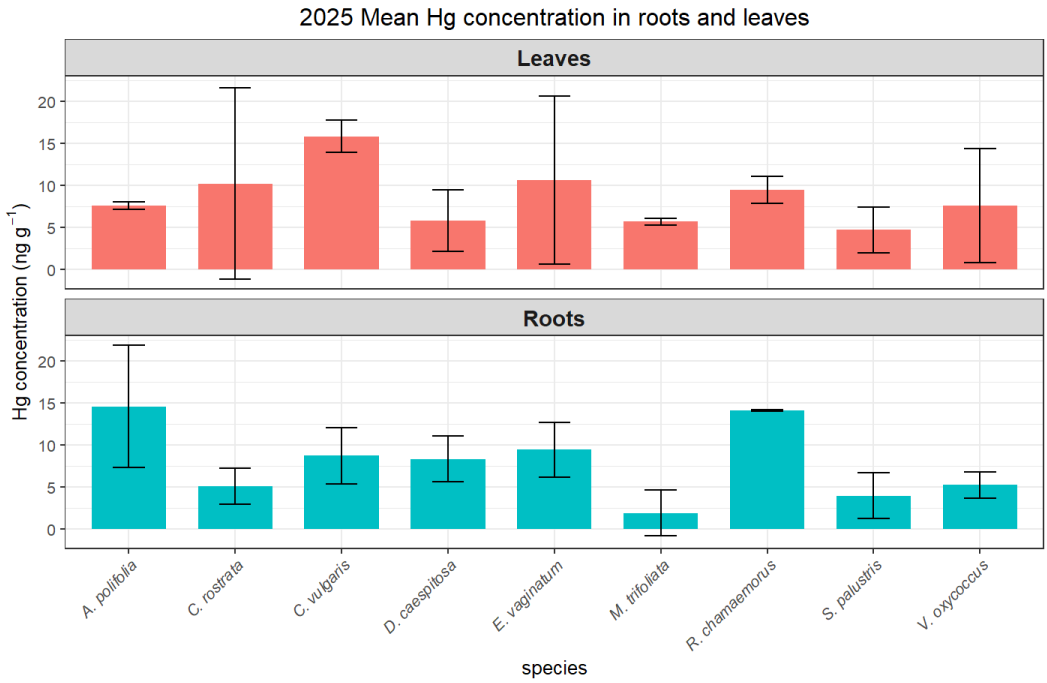

Figure 16. Mean Hg concentrations (± SD) in the leaves and roots of the analyzed plants.[6]

3.4.4 Comparison of 2025 and 2020 vegetation data

The Hg concentrations from 2025 were compared with Hg concentration measured in 2020 in Degerö in the leaves coming from four species of vascular plants: *Scheuchzeria palustris*, *Vaccinium oxycoccus*, *Andromeda polifolia*, and *Eriophorum vaginatum*. The non-vascular moss species *Sphagnum balticum*, *S. fuscum*, *S. majus* (only data from 2025), and *S. lindbergii* (only data from 2020) were also used for comparison.

The differences between the Hg concentrations in the vascular plants between 2020 and 2025 are all within the uncertainty range (Figure 17). It can therefore be assumed that the concentrations did not change significantly between the years.

[6] The error bar assigned to the roots of *D. caespitosa*, *M. trifoliata*, and *S. palustris*, and to the leaves of *S. palustris* does not present the standard deviation, because only one sample of each type was analyzed. Instead, it depicts the analytical uncertainty of the CRM NIST 1515 (n = 21), which was used for analyzing the Hg concentration.

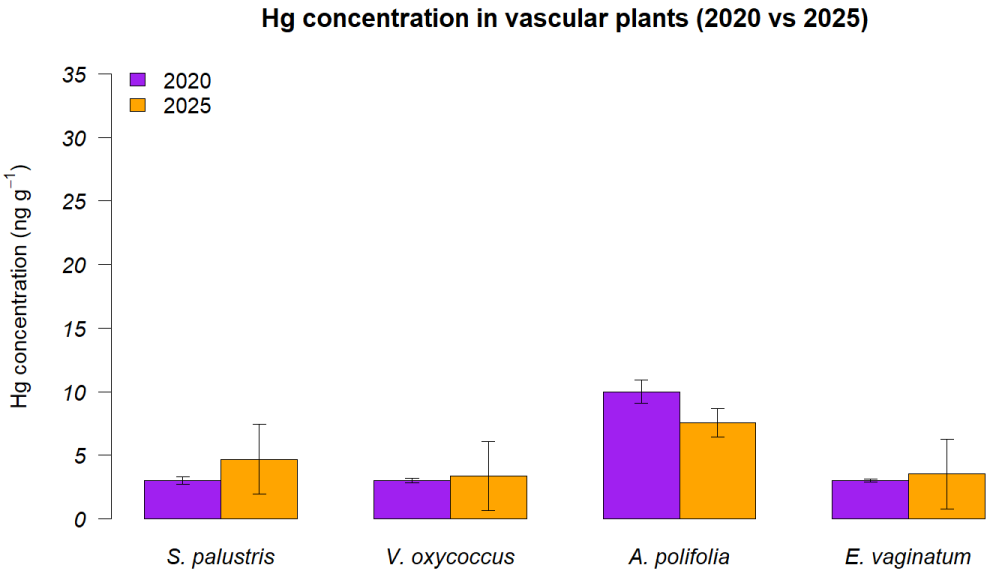

Figure 17. Comparison of Hg concentrations (± SD) in vascular plants in 2020 and 2025.[7]

The mosses, on the other hand, had lower Hg concentrations in 2025 than in 2020 (Figure 18). Only two species could be compared between both 2025 and 2020, though. The highest Hg concentration in 2015 was found in *Sphagnum lindbergii*, which was not available for comparison in 2025. It would be valuable to conduct a detailed analysis of Hg concentrations in different species of moss on Degerö Stormyr.

[7] The error bar for the 2025 samples of *S. palustris*, *V. oxycoccus*, and *E. vaginatum* does not present the standard deviation, because only one sample of each type was analyzed. Instead, it depicts the analytical uncertainty of the CRM NIST 1515 (n = 21), which was used for analyzing the Hg concentration.

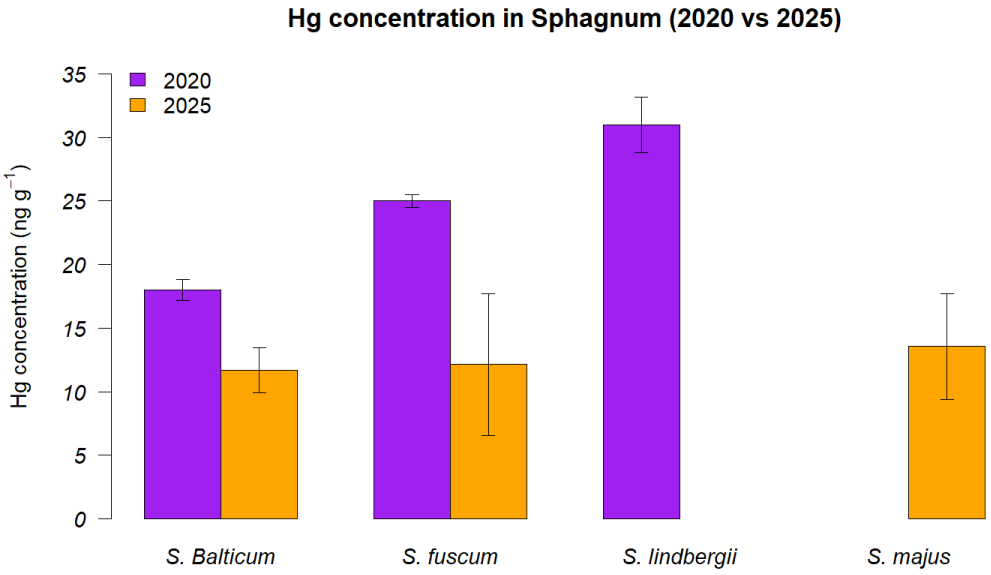

Figure 18. Comparison of Hg concentrations in Sphagnum species in 2020 and 2025.

4. Discussion

4.1 Fluctuations of the Hg pool

Hg concentrations in peat cores are significantly different between years. There are several possible explanations for such differences. Hg evasion from the peatland seems to be a contributing factor. Osterwalder *et al*., (2017) found that the annual Hg mass balance at Degerö was dominated by evasion of Hg^0 between June 2013 and June 2014. However, isotope data do not show any evidence of diffusion of Hg^0 through the unsaturated zone of the peat and back to the atmosphere. Instead, photoreduction of Hg^{2+} is supposed to be the main reason for the mobility of Hg in the peatland (Li *et al*., 2023). Diffusion via peat pores is thus unlikely to be the most important reason for the decrease in the Hg concentration.

A potential reason for the distinct Hg concentrations could be differences in the plant species growing on the surface. Differences in vegetation type and plant species can affect Hg sequestration in the peat record. Changes in vegetation type and species composition can affect the Hg record recorded in peat (Rydberg *et al*., 2010). If the cores sampled in 2015 were overgrown by plants that take up more Hg from the atmosphere, they might have accumulated more Hg than cores sampled in 2025 (see the next section on a detailed explanation on vegetation).

Water-table fluctuations might be a factor regulating Hg mobility. Experiments conducted by Haynes *et al*., (2017) showed that lowering the water table can increase the Hg concentrations in peat pore water and the downstream transport of Hg (Haynes *et al*., 2017). If the water table dropped in Degerö between 2015 and 2025, it could have led to higher Hg concentrations in pore waters and lower Hg concentrations in the peat itself due to Hg mobilization. Discharge of Hg^{2+} complexes with dissolved organic matter is an additional pathway of Hg loss from the peatland, accounting for about half of the Hg loss caused by photoreduction (Li *et al*., 2023).

Another possible explanation for the difference in Hg peat concentrations might be differences in peat decomposition rates. Hg concentrations can increase significantly as peat loses mass during humification (Biester *et al*., 2003). Lower concentrations in peat from 2025 might thus be due to lower humification rates, rather than a lower Hg supply to the peat. A C:N isotope analysis would provide an overview of the decomposition rates of the peat.

Some of the peat cores from 2025 have a quite strong correlation between the Hg concentration and dry bulk density, like the cores from the Southern and Eastern sites (Figure 19). A slightly weaker correlation was found in cores West 1 and North 2. Core West 2 shows a moderate correlation between the two parameters, but only a weak correlation can be seen in core North 1. The lack of

correlation is especially evident in the lower part of the core, where the Hg concentration rises (Figure 8) but dry bulk density remains nearly constant (Figure 10).

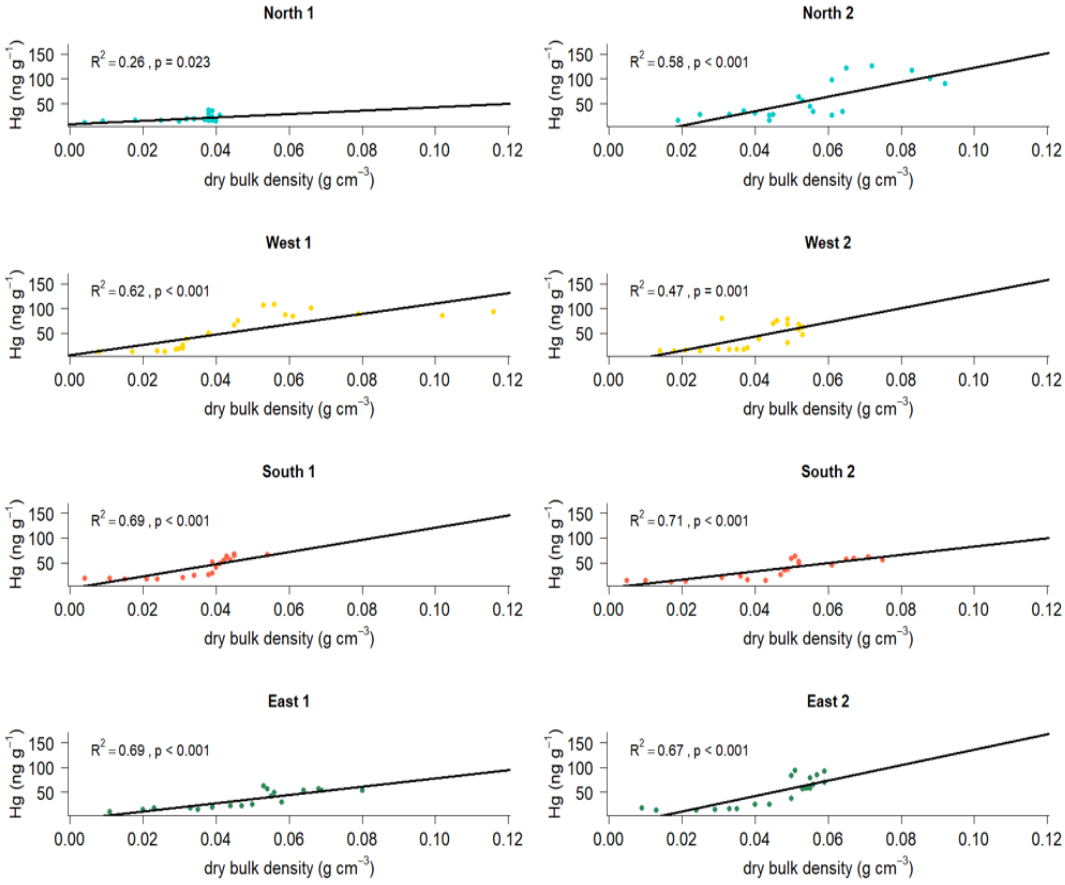

Figure 19. Correlation plots of Hg concentration and dry bulk density of the 2025 peat cores.

Cores sampled from the Northern Site have a strong correlation between the two parameters (Figure 20), which is not the case in the 2025 cores from the same site. Cores from the Southern Site also have a strong correlation, especially Core 5, but very different results were yielded by cores from the Western Site. A very strong correlation is evident in Core 5, but only a very low correlation in Core 6, even though both cores come from the same sites. This shows how different the properties of peat can be, even if the cores were collected close to one another.

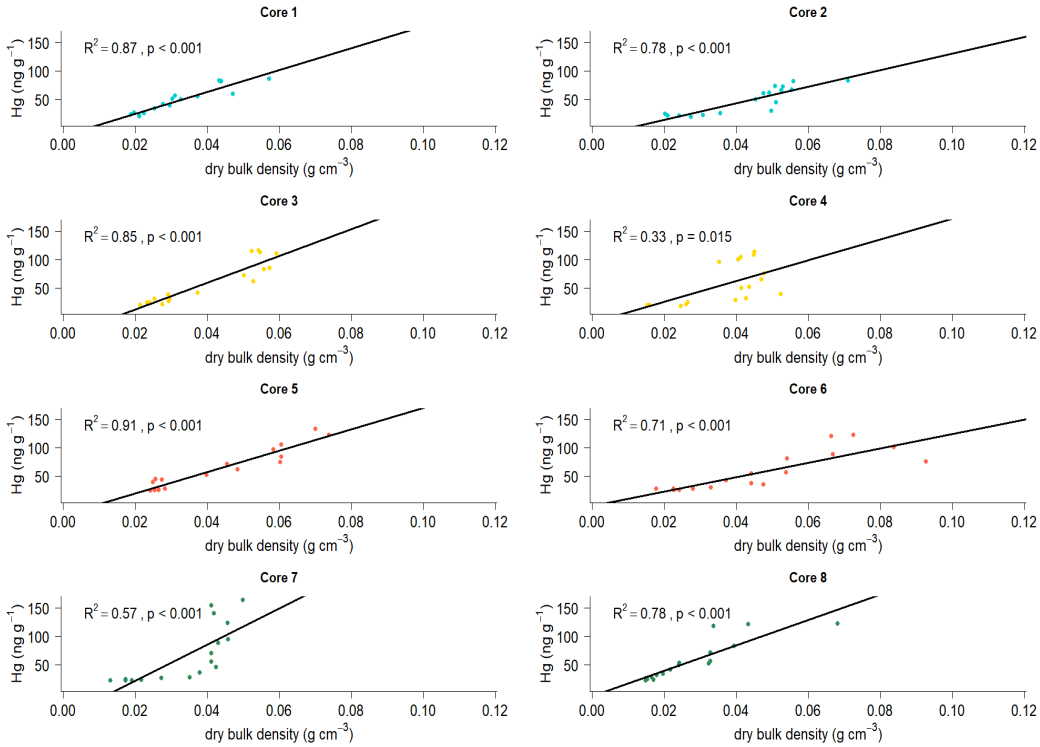

Figure 20. Correlation plots of Hg concentration and dry bulk density of the 2015 peat cores.

The correlation between the average Hg concentration and average dry bulk density is slightly lower in the 2025 cores than in the 2015 cores (Figure 21). Despite the decrease in correlation, it is still quite strong in the 2025 cores. However, it seems that there is no correlation between the Hg concentration and dry bulk density of the lawn core sampled in 2020. However, this single core was sampled ca. 200 m away from the sites where the 2015 and 2025 cores were collected. This high intra-variability underscores the importance of sampling new cores as close as possible to where old cores were sampled when analyzing trends across years.

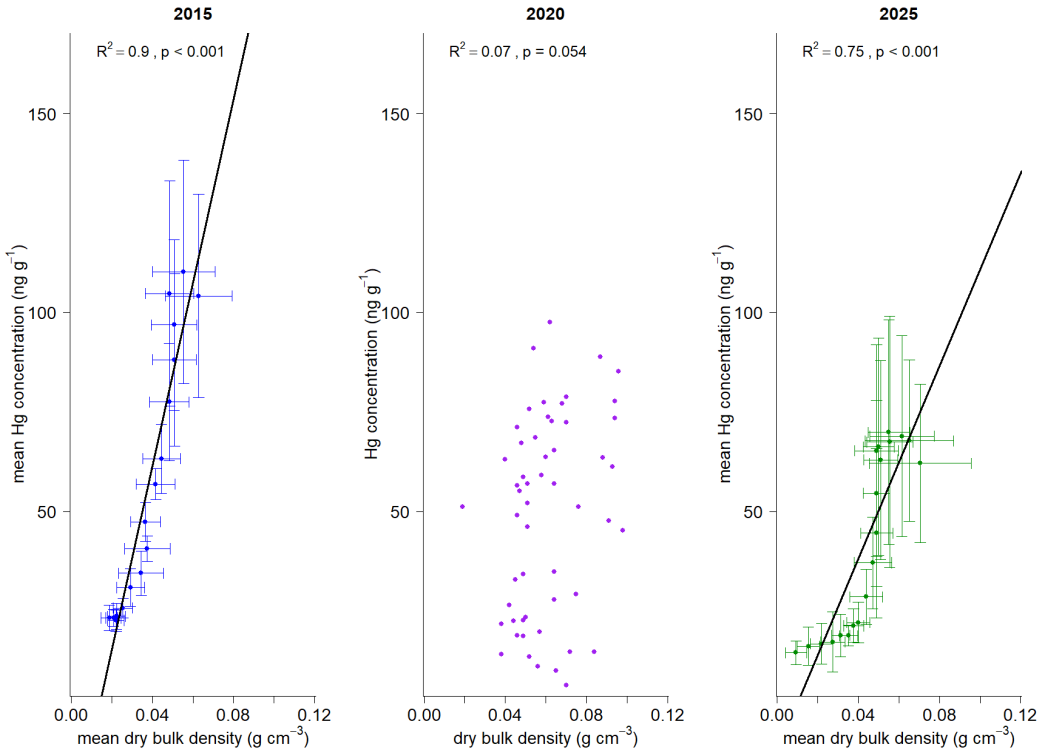

Figure 21. Correlation plots of the mean Hg concentration and dry bulk density between the 2015, 2020, and 2025 peat cores.[8]

4.2 Variability of Hg concentrations among plant species and tissues

The vegetation data show that Hg concentrations vary substantially across plant species and tissues. Leaves generally had higher Hg concentrations than roots of the same species, but three species of plants had a higher Hg concentration in roots (*Rubus chamaemorus*, *Andromeda polifolia*, and *Deschampia caespitosa*). The highest Hg concentration was detected in the lichen *Cladina*. This is in accordance with the findings of Zhou *et al*., (2021), whose database shows that lichens and mosses generally have higher Hg concentrations than vascular plants. Lichens have also been found to have higher Hg concentrations than flowering plants in the Canadian Arctic (Choy *et al*., 2010). Three species of the moss genus *Sphagnum* were also analyzed, and their Hg concentrations also exceed those of most of the analyzed vascular plants. Further monitoring of vegetation on Degerö Stormyr, combined with Hg concentration measurements, could provide further insight into the link between the dominant surface plant species and Hg concentrations in peat.

[8] 2015 samples come from eight 34 cm cores. 2020 samples come from one 100 cm core. 2025 samples come from eight 40 cm cores.

4.3 Implications

The outcome of this study shows that the Hg pool in a peatland is dynamic and can significantly change over a decade. The results also highlight how variable peat properties can be, even when the peat is sampled from the same site. This shows the importance of collecting several cores when analyzing the Hg peat archive. Properties such as Hg concentration or dry bulk density can vary significantly between different cores. Many of the analyzed cores showed a correlation between dry bulk density and Hg, but this was not the case in all of them. The vegetation data also show how variable Hg concentrations can be across plant species. In some cases, even species within the same family can differ in Hg uptake capacity, such as *Andromeda polifolia* and *Vaccinium oxycoccus* within the Ericaceae family. The Hg concentration often depends on the part of the plant analyzed. (Figure 16).

4.4 Outlook

The results show that it is important to conduct regular sampling campaigns and Hg analyses in order to monitor the fate of Hg in peatlands. It would be of advantage to verify whether fluctuating Hg concentrations in peatlands can be reflected in changes in Hg concentrations in downstream water courses and lakes, as well as aquatic organisms susceptible to Hg biomagnification. Should the Hg supply to Swedish waters be reduced, consuming fish from Swedish lakes could become safer in the future. It is also important to measure Hg accumulation rates, which show deposition and post-depositional losses of Hg (Li *et al*., 2023). This is possible once the age of the peat is known.

The data obtained from analyzing Hg concentrations in the vegetation samples also show that it would be beneficial to include an analysis of plant macrofossils from each analyzed peat core. This would provide information on which type of vegetation was most abundant during peat formation. Monitoring surface vegetation would also contribute to our understanding of which species are present on the peatland surface and how they might affect the quantities of Hg taken up.

5. Conclusion

The results of this thesis show that the Hg pool of Degerö Stormyr significantly decreased between 2015 and 2025. Possible explanations include evasion of Hg, different vegetation species growing on the peat, different age or decomposition rates of the peat, and possibly also Hg mobility within the peat profile. The input of Hg into the peat can be influenced by shifts in vegetation composition, as different plant and lichen species have different Hg concentrations. The properties of peat can be highly variable, even when cores are sampled close to each other, underscoring the importance of sampling several cores for Hg analysis. A detailed analysis of both peat properties and vegetation composition enables a better understanding of Hg cycling in peatlands.

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Popular science summary

Mercury (Hg) is a toxic substance that can cause severe health problems. Large amounts of mercury are stored in Swedish peatlands, from where it is transported to rivers and lakes and pollutes fish making them unsafe to eat. It is important that we know how much mercury is stored in peatlands and if the amount changes with varying concentrations of mercury in the atmosphere. This study analyzed Hg concentrations in peat cores collected in 2015, 2020, and 2025 from a peatland called Degerö Stormyr in the north east of Sweden. The results show that peat sampled in 2025 has a lower Hg concentration than that sampled in 2015 and 2020. They also show that peat can have variable properties, even if the cores were sampled close to each other. The analyses of Hg content in plants growing on the surface of the peat show that different species of plants contain different amounts of Hg. It is important to monitor Hg concentrations in peat regularly to have an overview of how the concentrations change with time.

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