BIOGEOCHEMICAL FACTORS AFFECTING MERCURY METHYLATION IN SEDIMENTS OF THE VENICE LAGOON, ITALY

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Abstract—Mercury methylation and sulfate reduction rates, total Hg, and monomethyl Hg in the sediments of the Venice Lagoon (Italy) were measured in June 2005 in order to identify the factors affecting the methylation of inorganic Hg. While the rates of Hg methylation and sulfate reduction were generally higher in the surface layers (0–2.5 cm), the correlation between Hg methylation and sulfate reduction rates was not significant when considering all depths and sites. This discrepancy is discussed considering two factors: the activity of sulfate-reducing bacteria and Hg solubility. The former factor is important in determining the Hg methylation rate in comparable geochemical conditions as evidenced by similar vertical profiles of Hg methylation and sulfate reduction rates in each sediment core. The latter factor was assessed by comparing the Hg methylation rate with the particle–water partition coefficient of Hg. The Hg methylation rates normalized to sulfate reduction rates showed a negative linear correlation with the logarithm of the particle–water partition coefficient of Hg. The Hg methylation rates normalized to sulfate reduction rates showed a negative linear correlation with the logarithm of the particle–water partition coefficient of Hg, suggesting that the availability of dissolved Hg is a critical factor affecting Hg methylation. Solid FeS seems to play an important role in controlling the solubility of Hg in Venice Lagoon sediments, where sulfate and iron reductions are the dominant electron-accepting processes. Overall, the production of monomethyl Hg in the Venice Lagoon is controlled by a fine balance between microbial and geochemical processes with key factors being the microbial sulfate reduction rate and the availability of dissolved Hg.

Keywords—Mercury Methylation Sulfate reduction Sediment Venice Lagoon

INTRODUCTION

Monomethyl Hg (MMHg) can be toxic to marine and freshwater organisms yet is often retained and biomagnified along the food chain to a greater extent than inorganic Hg, with sometimes no apparent harm to the contaminated organism [1]. This accumulation can be deleterious for more sensitive consumers and poses a potential health risk for humans through the consumption of MMHg-contaminated fish [2]. In marine systems, principal sources of MMHg are estuarine and coastal sediments where most of the methylation of inorganic Hg occurs [3]. The bioaccumulation of MMHg in benthic invertebrates and subsequent transfer to higher trophic levels couples sedimentary MMHg to marine food chains [4].

Although methylation processes in anoxic sediments are not completely understood, numerous studies have demonstrated that sulfate reduction involving microbial processes is tightly coupled to Hg methylation [5–7]. Initial evidence was provided by the inhibition of Hg methylation by molybdate, a metabolic inhibitor of sulfate reduction [5]. More convincing evidence was obtained by the addition of sulfate to boreal peatland pore water and sediment, which caused an increase in the MMHg concentrations up to certain sulfate levels [6]. Increasing the in situ concentration of sulfate in the peat and peat pore water by 2 or 20 times resulted in an increase of pore water MMHg by a factor of three to four. The increase in MMHg was not proportional to the added amount of sulfate, probably because of limitations in bacterial population size and availability of organic matter [6]. Experiments with pure cultures of sulfate-reducing bacteria (SRB) demonstrated that acetate-utilizing microbial groups (e.g., the family Desulfovibrio bacteriaceae) show higher Hg methylation rates than microbial groups that are unable to use acetate (e.g., the family Desulfobulbaceae), suggesting that a genetic component and/or carbon metabolism is related to the efficiency of Hg methylation [8]. The activity and composition of SRB have been considered major factors affecting Hg methylation [9].

IN SUMMARY, the bioavailability of inorganic Hg is another critical factor that determines the Hg methylation rate. Benoit et al. [3,11] suggested that the passive uptake of the neutral HgS0 complex controls the bioavailability of Hg to methylating bacteria. In their study [3], a chemical equilibrium model was set to include the interaction of Hg with solids containing one or two sulfide groups. The modeled decline in HgS0 with increasing dissolved sulfide in pore water was consistent with the observed decline in sedimentary MMHg. In low-sulfide coastal sediments, the particle–water partition coefficient, Kp = Cw / Cw0 (mole/Kg) / (mole/dm3), was suggested to be an important factor regulating the availability of Hg to methylating bacteria [12]. In the same
Understanding Hg methylation in estuarine sediments is challenging because the biological and geochemical processes involved affect each other as well as the Hg methylation process. In the present study, Hg methylation rates were determined along with microbial sulfate reduction rates and conventional geochemical parameters in order to investigate the factors affecting the methylation of inorganic Hg in the sediments of the Venice Lagoon.

MATERIALS AND METHODS

Study areas

Venice Lagoon, located in the northern part of the Adriatic Sea, is the largest lagoon in the Mediterranean (Fig. 1). It has an average depth of 1 m and a surface area of 549 km², 40% of which consists of tidal marshes, islets, and fish farms [13]. Venice Lagoon connects to the sea through three inlets, Lido, Malamocco, and Chioggia. Tidal flow through these inlets determines water circulation patterns that hydrologically isolate the northern basin, north of Malamocco inlet, from the southern part of the lagoon. It has been reported that the northern basin is highly contaminated by trace elements, such as Zn, Pb, Cu, Ni, and Cr [14]. Depth profiles of concentrations and fluxes of trace metals (e.g., Zn, Cu, Cr, Ni, and Pb) in 60-cm sediment cores showed a significant increase of these metals from the 1920s, with maximum inputs between the 1930s and 1970s. Because of the reduced input in recent years, trace metal concentrations have decreased in surface sediments starting approximately 5-cm depth [14]. Industrially discharged trace metals from the petrochemical zone of Porto Marghera are distributed in the sediments of the northern basin as a result of suspension of polluted sediment and its transport via a strong tidal flow [15,16]. Continuous erosion and resuspension of sediment due to anthropogenic activity, such as speed boat-
Table 1. Summary of quality assurance for the measurement of total Hg and monomethyl Hg (MMHg) in sediments and pore waters of the Venice Lagoon (Italy)

<table>
<thead>
<tr>
<th>Measure</th>
<th>Parameter</th>
<th>Sediment</th>
<th>Pore water</th>
</tr>
</thead>
<tbody>
<tr>
<td>Method blanks</td>
<td>Mean</td>
<td>0.73 (ng/g)</td>
<td>0.019 (ng/g)</td>
</tr>
<tr>
<td>CV (%)</td>
<td>119</td>
<td>13</td>
<td>44</td>
</tr>
<tr>
<td>n</td>
<td>6</td>
<td>6</td>
<td>8</td>
</tr>
<tr>
<td>CRM recovery</td>
<td>Mean (%)</td>
<td>99.5</td>
<td>102</td>
</tr>
<tr>
<td>CV (%)</td>
<td>5.4</td>
<td>17</td>
<td>5.0</td>
</tr>
<tr>
<td>n</td>
<td>19</td>
<td>7</td>
<td>7</td>
</tr>
<tr>
<td>Matrix spike recovery</td>
<td>Mean (%)</td>
<td>101</td>
<td>110</td>
</tr>
<tr>
<td>CV (%)</td>
<td>8.1</td>
<td>5.6</td>
<td>—</td>
</tr>
<tr>
<td>n</td>
<td>8</td>
<td>6</td>
<td>—</td>
</tr>
</tbody>
</table>

* CV = coefficient of variation.
* MDL = method detection limit.
* CRM = certified reference material.

Monomethyl Hg in sediments was extracted as described in Choe et al. [19]. Approximately 1 g wet sediment was mixed with 5 cm³ acidic KBr solution, 1 cm³ of 1 M CuSO₄ solution, and 10 cm³ of CH₂Cl₂. After 2 h of reaction time at room temperature, vigorous shaking was carried out for an additional hour using a mechanical shaker. After centrifugation for 20 min at 3,000 rpm, a 2 cm³ aliquot of the CH₂Cl₂ layer was pipetted into acid-cleaned 60 cm³ Teflon distillation vials, and then 40 cm³ of Milli-Q water were added to each vial. Distillation vials were placed into a heating block held at 45°C, and solutions were purged with N₂ until complete volatilization of CH₂Cl₂. Monomethyl Hg measurements in pore-water samples were conducted using aqueous-phase distillation for purification. Mercury species was quantified by cold vapor atomic fluorescence spectrometry [19]. Monomethyl Hg extracted from sediment samples was quantified in the same way without the distillation step. The detection limit (estimated as three times the standard deviation of the method blank), the matrix spike recovery, and the certified reference material recovery are presented in Table 1.

For the measurement of Hg methylation rates, the tracer method using Hg stable isotopes was used in the present study with a cold vapor generation system interfaced to an inductively coupled plasma/mass spectrometer to allow a greater sensitivity [12]. In our method, 200Hg working solution diluted from 200Hg(NO₃)₂ with filtered overlying water was added to sediment samples in concentrations of 2 ng 200Hg/g wet sediment under an N₂-saturated atmosphere. This concentration range corresponds to 0.5 to 2% (weight/weight) of natural Hg concentrations in Venice Lagoon sediments. The sediment samples were incubated under anoxic conditions for 4 h at field temperatures (21°C), after which sediment samples were frozen at −80°C. The concentration of MM200Hg in incubated sediment samples was detected using an inductively coupled plasma/mass spectrometer. Methylation of added 200Hg was evaluated as the excess concentration of 200Hg versus 198Hg in the total concentration of MM 200Hg (the sum of MM 200Hg produced from tracer injection, MM 200Hgtracer is the concentration of MMHg produced from tracer (200Hg) injection, ΣMM198Hg is the total concentration of MM 198Hg in tracer solution.

\[
MM_{200Hg_{\text{tracer}}} = \left[ \text{ΣMM}^{198\text{Hg}} - \text{ΣMM}^{200\text{Hg}} \times R_{\text{natural}} \right] + \left[ (R_{\text{tracer}} - R_{\text{natural}}) \times A_{200} \right]
\]

(1)

Hg methylation rate (%/h) = \[\frac{MM^{200}\text{Hg}_{\text{tracer}} \times 100}{(200\text{Hg}_{\text{added}} \times \text{hours of incubation})}\]

(2)

Where MM^{200}\text{Hg}_{\text{tracer}} is the concentration of MMHg produced from tracer (200Hg) injection, ΣMM^{198\text{Hg}} is the total concentration of MM 198Hg, ΣMM^{200\text{Hg}} is the total concentration of MM 200Hg (the sum of MM 200Hg produced from tracer injection), and A_{200} is the abundance of 200Hg in tracer solution. The R_{tracer} and R_{natural} represent the ratio of 198Hg/200Hg in tracer solution and natural ratio of 198Hg/200Hg, respectively.

The detection limit of MM200Hg produced is a function of the ambient MMHg concentration (~0.5 ng MMHg/g dry sediment), the natural abundance of 200Hg, and the precision in the measurement of 198Hg/200Hg. In this study, the detection limit averaged 1.27 pg/g with 1.1% CV in the measurement of 198Hg/200Hg. In most cases, the calculated concentrations of MM200Hg were greater than the detection limit; otherwise, methylation rate was considered as 0%/h. The precision of the method was 32% CV (n = 19) based on duplicate analysis.

The determination methods of dissolved iron, alkalinity, and dissolved sulfate are described by Gieskes et al. [21]. Dissolved sulfide was determined by a modification of the spectrophotometric method described in Strickland and Parsons [22].
bottles under anaerobic conditions, and Na₂SO₃ was injected. Serum bottles were incubated 2 to 3 d at room temperature. Then a test tube containing 2.5 cm³ of 10% (w/v) anoxic zinc acetate solution was placed inside the serum bottle as a sulfide trapping solution. The serum bottles were injected with anoxic 1 M Cr(II)-HCl (8 cm³) and 12 N HCl (4 cm³) solution and gently agitated for 2 d. The liberated H₂S precipitates as ZnS in the trapping solution. After removal of this solution from the serum bottles, the sulfide produced was quantified by scintillation counting [23].

Sulfate-reducing bacterial counts were carried out in duplicate for sites A, C, and S2 using the most-probable-number assay in 48-well microtiter plates. A 0.9-cm³ volume of brackish seawater medium, containing one of carbon sources (20 mM), formate, acetate, or lactate, and sulfate as electron acceptor, was added to each well under anoxic conditions. Then the inoculum (0.1 cm³), prepared by adding 0.2 g of sediment to 2 cm³ of degassed medium, was added to the first well from which the serial dilution begins. The plates were incubated at room temperature (21°C) under anoxic conditions to allow bacterial growth. After two weeks of incubation, 0.2 cm³ of 2 M FeSO₄ solution was added to each well. Wells with positive SRB growth were identified by the formation of a black FeS precipitate. Black FeS formation as a function of dilution was analyzed with a statistical table for most-probable-number count [24].

Statistical analyses
Analysis of variance (single-factor ANOVA) and post hoc multiple comparison of means (Fisher’s protected least squares difference) were used to test the significance of differences between two data sets considering the arcsin(√x)-transformed data set to comply with heteroscedasticity and normalization of percentages data [25]. Correlations between factors were fitted with linear model established and tested for 95% significance using SigmaPlot 8.0 (Systat Software, San Jose, CA, USA). Unless otherwise stated, values represent means with one standard deviation.

RESULTS AND DISCUSSION
Geochemical parameters in pore waters
The vertical profiles of the geochemical parameters for the pore waters of the sampling sites are shown in Figure 2. Large increases in alkalinity with depth were observed at sites C and S0 from 4.3 to 36 mM and from 23 to 38 mM, respectively, which reached a maximum value of 80 mM at 120 cm (data not shown). The alkalinity profiles of sites C and S0 agree with a large increase in sulfide and a decrease in sulfate, suggesting that a principal factor affecting alkalinity in pore water is bacterial sulfate reduction [26]. Such high alkalinity conditions of sites C and S0 have been observed in organic-matter-rich sediments in various settings, such as in the anoxic fjord Saanich Inlet in British Columbia, Canada [27], as well as in Po Delta sediments in Italy [28]. Alkalinity in pore waters of S1, S2, and B showed relatively small increases at 50- to 120-cm depth, which agrees with small increases in sulfide at the same depth range (data not shown).

Dissolved Fe peaks were observed in sites B, S1, and S2 and the surface of C, indicating that Fe oxide reduction is a process that contributes to the diagenesis of organic matter immediately below the sediment–water interface. In site C, a decrease in dissolved Fe is followed by an increase in dissolved sulfide, suggesting formation of solid FeS. The surface of site S0 might have had similar dissolved Fe profile to site C, yet it was not possible to observe the dissolved Fe peak because of the loss of the surface sediments during the collection process. Sites C and S0 probably rapidly accumulate organic rich materials because of their distinctiveness as nearshore and channel sites, which causes high sulfide production close to the sediment–water interface. Indeed, sulfate was depleted within the upper 30 cm in sites C and S0, while other sites demonstrated a moderate decrease in sulfate from 30 to 15 mM in the depth range of 0 to 120 cm (data not shown). Increases in alkalinity below the depth of disappearance of sulfate as a result of incipient processes of methanogenesis were shown in sites C and S0 (data not shown) [27,28].

Total Hg in sediments
The maximum amount of total Hg in surface sediments (0–2.5 cm) was found in site C with a concentration of 1,144 ng/g, while the other sites ranged from 209 to 691 ng/g (Fig. 2). Previous studies have shown that the Porto Marghera area is the most Hg-contaminated zone in the Venice Lagoon, together with the canals in the city of Venice [16,17]. The total Hg concentrations determined in the surface layers of nearshore and open water sites (B, S0, S1, and S2) ranged between 482 and 691 ng/g, which agreed with literature data (641 ± 260 ng/g [16]). This range corresponds to the upper range of total Hg concentrations found in several urbanized estuaries on the East Coast of the United States (200–700 ng/g [29]). Surface sediments of site A showed the lowest total Hg concentration of 209 ng/g. However, this value is still almost twice as much as the background Hg level of 118 ng/g found in surface sediment of the Adriatic side of Lido Inlet [17]. It has been estimated that approximately 1,100 kg of Hg were carried to the Adriatic Sea each year on suspended matter because of the natural and anthropogenic resuspension of surface sediments [16].

We generally found that Hg concentrations in the upper 2.5 cm of the surface sediments decreased with increasing distance from the Porto Marghera, following the trend of organic carbon content in the surface sediments (Fig. 3). The total Hg concentration in estuarine sediment often has a positive correlation with the organic carbon content because of the strong association between sedimentary organic carbon and Hg [12,29]. The correlation coefficient increases when the data point of S2 is not included in the linear regression (n = 5, r² = 0.95, p < 0.05), showing that the surface sediments of S2 have low total Hg content in spite of high organic carbon concentration. This may be the result of inflow from the southern basin, which is hydrologically isolated from the northern basin and flushed with cleaner water from the Adriatic Sea. Indeed, approximately fivefold less Hg was observed in the southern basin than the northern basin in the water column [16]. In addition, site S2 is farther away from the main source at Porto Marghera.

Vertical profiles of Hg in sediment cores were relatively homogeneous between 0- and 20-cm in depth with the exception of the nearshore sites C and S0 (Fig. 2). Historically, the largest source of Hg in this area has been a chloro-alkali plant that was run from 1951 to 1988 in the Marghera area without pollution control [16]. The vertical profiles of total Hg in C and S0 sites showed that Hg increases with depth from the surface to 7 cm. Sedimentation rates were not determined in this study, but the sedimentation rate found by Frignani et al.
MMHg in sediments

The MMHg concentrations in surface sediments ranged from 0.31 to 1.7 ng/g (Fig. 4), which are typical concentrations for urbanized lake and estuarine sediments [19,29]. The fraction of Hg as MMHg averaged 0.10 ± 0.02% (n = 6) at 2.5 cm/year offshore Porto Marghera, demonstrating that the decreased inputs of sedimentary Hg started in the late 1980s. Similar depth profiles of other heavy metals, increasing with depth because of the reduced input in recent years, have been reported for Zn and Cu [14].
to 5 cm and $0.07 \pm 0.05\% \ (n = 23)$ at 5 to 20 cm, while significantly different (ANOVA, $p < 0.0001$) and higher percentages of MMHg were found in the upper 2.5-cm sediments ($0.17 \pm 0.08\%, \ n = 6$), demonstrating that Hg methylation occurs predominantly in the surface oxic/anoxic transition zone. This result is consistent with the increase of Hg methylation rate in the upper 2.5 cm (Fig. 2).

Similar vertical profiles noted between total Hg and MMHg in most of the sampling sites, except increases in MMHg in surface layers, suggest that Hg availability is one of the important factors for determining sedimentary MMHg concentrations (Fig. 2). The correlation between MMHg and total Hg was linear for the upper 0 to 2.5 cm except for sites S2 and C ($n = 4, r^2 = 1.0, p < 0.05$) at 2.5 to 5 cm ($n = 6, r^2 = 0.96, p < 0.05$) (Fig. 4). Considerably lower concentrations of MMHg were observed in surface layers of C and S2 compared to what would be expected on the basis of the linear regression models, suggesting that total Hg concentration was not a limiting factor for the production of MMHg in surface of C and S2. In deeper sediments, the different biogeochemical settings (e.g., dissolved sulfide, dissolved iron, and microbial community composition) may lead to large spatial variations in Hg methylation conditions, resulting in a poor correlation between total Hg and MMHg (5–10 cm, $n = 12, r^2 = 0.43, p > 0.05$; 10–15 cm, $n = 6, r^2 = 0.07, p > 0.05$; 15–20 cm, $n = 5, r^2 = 0.17, p > 0.05$; Fig. 4).

**Total Hg and MMHg in pore waters**

The total Hg concentrations in pore waters for all the sampling sites at all depths (Fig. 2) ranged from 19 to 148 pM ($\bar{x} = 71 \pm 33$ pM, $n = 16$), which is approximately one order of magnitude higher than the average dissolved Hg concentrations found in overlying waters of the northern lagoon (7.2 ± 0.6 pM [16]). The values of log $K_d$ ($K_d = C_s$ [mole/kg]/$C_w$ [mole/dm$^3$], 1 dm$^3$ = 1.02 kg) determined at all the sampling sites and depths were within a relatively narrow range (log $K_d$ = 4.80 ± 0.11, $n = 14$), which suggests that a common factor that is relatively constant at various sites and depths controls the sediment–water partitioning of total Hg. The range of $K_d$ found in the present study (4.56–5.01) is consistent with the range found in other estuarine sediments (3.18–4.92 [12]; 4.89 ± 0.43 [30]).

When all the sampling sites are considered together, a negative exponential relationship was observed between dissolved Fe and dissolved Hg in pore waters, suggesting that processes associated with Fe oxide reduction lead to decreases in Hg solubility (Fig. 5). Taking into account that Fe oxide reduction releases dissolved organic matter [31] and that dissolved organic matter provides strong binding sites for Hg [32], a neg-
Mercury methylation in the Venice Lagoon sediments

Mercury methylation rates in surface sediments ranked B, S0 > S1, S2 > A > C, which generally agrees with the ranking of % MMHg (S1, B, S0 > A > C, S2). The second lowest Hg methylation rate at site A is in agreement with the least amount of organic carbon at that site, emphasizing the role of sulfate-reducing bacteria. The lowest Hg methylation rate was found at site C, which supports the low % MMHg previously reported in the same area of the lagoon [16]. This may be due to the lowest sulfate reduction rate, as shown in Figure 2. However, the low % MMHg and Hg methylation rates of surface S2 with a relatively high sulfate reduction rate suggest that microbial factors involving organic carbon availability and sulfate reduction rate cannot provide a sufficient explanation for the spatial distribution of MMHg production.

The effect of the Hg solubility on the rate of Hg methylation was assessed in Figure 6. The comparison in Figure 6A is similar to that used by Hammerschmidt and Fitzgerald [12] for Long Island Sound sediments. In their study, the Hg methylation rate showed significant correlation with the log $K_d$ of Hg, while the correlation in Figure 6A is not clear. However, the log $K_d$ of Hg in Long Island Sound sediments demonstrated a larger range (3.2–5.0) compared to the range of the log $K_d$ in Venice Lagoon sediments (4.6–5.0). In our sites, not only the narrow range of $K_d$ but also the large variation in the sulfate reduction rate may cause a nonsignificant correlation between the solubility of Hg and the Hg methylation rate. In order to eliminate biological activities involved in each site and depth, the log $K_d$ was shown as a function of the Hg methylation rate divided by the sulfate reduction rate (Fig. 6B). The resulting correlation factor was improved, suggesting that availability of dissolved Hg is one of the critical factors for MMHg production rate. The low significant correlation factor ($n = 11$, $r^2 = 0.64, p < 0.05$) may represent a relatively high CV (32%) in the analysis of the Hg methylation rate and/or heterogeneity between two cores collected for the measurement of sedimentary Hg and pore-water Hg. In addition, the chemical speciation change of dissolved Hg [39], especially in high-sulfide cores C and S0, may contribute to this low correlation factor.

CONCLUSION

Venice Lagoon sediments are extensively contaminated with Hg as a result of resuspension and tidal flushing of contaminated sediments originating from chloro-alkali discharge and other sources. Sedimentary Hg concentration in the surface layer ranged from 209 to 1,144 ng/g, with the lowest value
near the Lido Inlet (site A) and the highest value in the Porto Marghera area (site C). The distribution of total Hg in the surface sediments was affected by the organic carbon content in the sediment as well as the distance from the Porto Marghera. High sulfate and other nutrient concentrations in pore water caused active microbial production of MMHg, averaging 1.0 ± 0.98 %/d (6.5 ± 7.0 ng/g/d) for 0 to 20 cm and 1.7 ± 1.3 %/d (10 ± 7.6 ng/g/d) for 0 to 5 cm. These ranges of Hg methylation rates are lower than those of low-sulfide coastal sediments (1–10%/d in 0–4 cm [12]) but higher than those of high-sulfide freshwater sediments (1–10 ng/g/d in 0–4 cm [38]).

The constant fractions of Hg as MMHg were observed in the upper 2.5-cm sediments with the exception of the high-organic sites, C and S2, suggesting that Hg availability is the key limiting factor for MMHg production in the surface sediments of the Venice Lagoon. The discrepancy in high-organic sites was explained by the activity of sulfate-reducing bacteria and Hg solubility. The former factor was limited in the highest Hg site, C, resulting in a lower Hg methylation rate compared to other sites. The importance of the latter factor was demonstrated by the fact that the Hg methylation rates normalized to the sulfate reduction rates showed a negative linear correlation with the logarithm of the particle–water partition coefficient of Hg. Indeed, the highest \( K_d \) (Hg) of surface layers was shown in site S2. The role of sedimentary organic carbon and acid-volatile sulfide has been emphasized, as they are related to the \( K_d \) of Hg in several estuarine sediments [10, 40]. However, organic carbon concentrations were relatively constant in various sites and depths of Venice Lagoon sediments, and thus these concentrations did not show a significant correlation with the \( K_d \) of Hg. Instead of organic carbon, solid FeS seems to be important for controlling the solubility of Hg in Venice Lagoon sediments, where sulfate and iron reductions are the dominant electron-accepting processes.

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Mercury methylation in the Venice Lagoon sediments

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