Although additional fieldwork is needed to narrow the uncertainty, this work indicates that termites have the potential to significantly influence the atmospheric budgets of some important trace gases.

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13. *Reticulitermes tibialis* Banks individuals were collected in Colorado near the National Center for Atmospheric Research by placing 15-cm square pieces of corrugated cardboard under rocks where termites had been observed. Within 1 month each piece of cardboard typically contained 119 to 2000 individuals. *Gnathamitermes perplexus* Banks specimens were collected in Arizona pastures in cow dung and shipped to our laboratory. Because each termite sampling site could have represented a discrete colony, the individuals from each site were kept in a separate culture.

    Each culture container was prepared by placing a 4-cm layer of pea gravel in the bottom of a 1-liter glass canning jar. The gravel was followed by a thin layer of glass wool and an 8-cm layer of sand. Each jar was moistened with 150 ml of distilled water. About 0.3 liter of head space remained in each jar. The jars, gravel, and sand were washed with distilled water and baked at 200°C for 15 hours before use. The lids of the jars were modified with 1/8-inch (outer diameter) copper inlet and outlet tubes. The colony of *Reticulitermes* that was analyzed for sulfur emissions was kept in an all-Teflon container prepared as described above and equipped with Teflon inlet and outlet lines. A flow of clean air (produced by an Aadco model 737 pure air generator with a methanizer option) at 10 ml/min was passed through each culture. The jars were placed in a thermostatically controlled water bath maintained at 30°C. The number of termites placed in each jar varied from 119 to 2000. All termites were counted at the beginning and at the end of the experiment. *Reticulitermes* cultures were fed a diet of sweet gum (*Liquidambar styraciflua*) pretreated with the fungus *Gloephyllum trabeum*. The *Gnathamitermes* were fed a diet of cow manure.

    Blanks were prepared and treated in an identical manner except that no termites were added. Some blanks included cow manure, others included fungus-treated wood. When the termites were removed from some jars that had been occupied for periods of a few weeks to 6 months, methane emissions were reduced to zero. Conversely, no methane uptake could be detected from any of the blanks. This indicates that methane-oxidizing bacteria were not present or that their activity was too low to affect methane concentrations during this experiment.

    Termite food consumption was calculated from the weight loss of the food placed in each culture. Food was dried at 70°C for approximately 12 hours before it was weighed. Termites consume their dead; therefore, the estimated weight of the dead termites was added to the weight of the food consumed to calculate emission yields. We used 0.45 as the ratio of the carbon in the food to the food dry weight (*32*).
14. The field enclosure method used was similar to that described for vegetation by P. R. Zimmerman, in *Final Report EPA-450/4-79-004* (Environmental Protection Agency, Office of Air Quality Planning and Standards, Research Triangle Park, N.C., March 1979).
15. Measurements were collected from three *Reticulitermes* and two *Gnathamitermes* colonies. Some gas samples were removed with a 10-ml gas-tight syringe (Precision Sampling Co.). For others, the exhaust line of the jar was connected directly to the sampling loop of a six-port sampling valve (Carle, model MK-2). The samples were analyzed chromatographically for CH_4, CO, and CO_2 by the method described by Rasmussen and Khalil (*9*); for C_2 to C_{10} hydrocarbons by the method of P. R. Zimmerman [*EPA 904/9-77-028, appendix C* (Environmental Protection Agency Region IV Air and Hazardous Materials Division, Atlanta, 1979)] and for reduced sulfur compounds by the method of S. O. Farwell *et al.* [*Anal. Chem.* **51**, 609 (1976)].
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19. An average emission yield was calculated for each of the five colonies. The standard deviation reported here represents the dispersion among these five data points. The DMDS conversion efficiency reported is the average of two samples collected from one colony; therefore, no standard deviation was calculated.
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33. Termite densities are from T. G. Woods and W. A. Sands [in (*18*), pp. 245–292]. Woods and Sands estimate O_2 consumption for termites in various ecosystems from termite densities and body sizes and from this calculate energy flows and biomass consumption. Our global estimates of biomass consumption, termite densities, and CO_2 evolution result in a calculated O_2 consumption per termite that is in good agreement with their experimental data.
34. We thank F. M. Weesner, who developed methods for collecting and handling the termites and provided us with some of the *R. tibialis* and with valuable advice. We thank R. B. Zimmerman for collecting the *G. perplexus*. L. E. Heidt, B. Bonsang, and J. Shetter performed some of the analyses and provided technical assistance. We appreciate R. Cicerone's assistance and advice. The National Center for Atmospheric Research is sponsored by the National Science Foundation.

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Tin and Methyltin Species in Seawater: Concentrations and Fluxes

Abstract. *The concentrations of tin and methyltin species in rivers, an estuary, and the surface and deep ocean generally are less than 50 picomoles of tin per liter. Estuarine profiles and river concentrations suggest that the dissolved riverine input of tin is only a minor source of this element to the oceans. Oceanic concentrations of inorganic tin decrease both with distance from land and with increasing depth from the surface, an indication of atmospheric transport to the surface ocean. Most of the contemporaneous eolian influx of tin to the oceans is anthropogenic. The vertical structure of tin concentrations in the northwestern Atlantic can be explained in terms of a model based on eolian input, advective processes, and removal of tin by particulate scavenging.*

Although a considerable body of knowledge exists on the geochemistry of tin in rocks and ores (*1*), the cycle of this element through weathering, transport from the continents to the sea by rivers and the atmosphere, oceanic circulation, and removal from seawater to marine sediments remains unknown, largely because of analytical problems. Meaningful determinations of tin concentrations in natural waters have been obtained only recently (*2–4*) after development of sufficiently sensitive analytical techniques and successful prevention of sample contamination. Three geochemical characteristics make tin an element of unique interest. (i) Its mobilization by man, as evidenced by an annual production of about 240 × 10^9 g, exceeds tenfold the natural rate of mobilization of tin by erosion. (ii) Tin is one of the three most highly enriched metals (after lead and tellurium) in atmospheric particulate matter as compared to the earth's crust (*5*). (iii) Tin can be biomethylated in the environment to organometallic species comparable in their toxicity to methylmercury (*2–4, 6*).

We present here data on the distributions of inorganic and methylated species of tin in rivers, an estuary, and the open ocean. These data provide evidence that most of the transport of tin from the continents to the open oceans is via the atmosphere, that this flux is primarily anthropogenic, and that the distribution of tin in the oceans may be used as a tracer for water mass transport and particulate scavenging removal processes. Methylated forms of tin are shown to be almost ubiquitous in natural waters.

Samples were collected from rivers in the southeastern United States and Europe and from Ochlockonee Bay in northwestern Florida. All riverine and estuarine samples were collected directly into acid-cleaned, low-density polyethylene Cubitainers. We sampled surface ocean waters with a sampler constructed of polycarbonate and polyethylene or from a rubber boat upwind of the ship. Subsurface samples from the Atlantic were obtained with a polypropylene sampler ("vane") designed and built by Boyle (*7*) to eliminate hydrowire contamination. Below a depth of 1500 m some contamination with water from the near-surface may have occurred, so that these values should be considered upper limits. Ocean samples were stored in Cubitainers or the polypropylene sample container. We analyzed all samples without pretreatment, usually within 24 hours of collection.

We analyzed samples by hydride generation coupled with cryogenic trapping on a chromatographic packing (*4*). The various tin species are detected by slow, controlled warming of the trap, and the hydrides are eluted into the flame photometric detector. With a 100-ml sample volume, detection limits are 1.7 p*M* (1 p*M* = 10^{-12} mole per liter) for inorganic tin and 1.3 p*M* for the methyltins. The precision of the technique is ± 5 percent.

The concentrations of dissolved tin species in the riverine samples are shown in Table 1. The average concentration of dissolved inorganic tin is 51 p*M* (87 p*M* if the Mississippi River value of 1036 p*M* is included). These values are in good agreement with Braman and Tompkins' (*3*) value of 101 p*M* for south Florida rivers. Our data for the methyltins differ from those of Braman and Tompkins (*3*). Where they found 54 percent of the total riverine tin to be in methylated forms, we found (with the exception of the highly polluted Rhine) only 5 percent of the total tin to be methylated.

We are studying the behavior of tin in the estuary of the Ochlockonee River, a small, relatively unpolluted coastal plain river entering the Gulf of Mexico southwest of Tallahassee, Florida. Our preliminary analysis of the estuarine behavior of tin shows that the riverine flux is an upper limit to the flux of dissolved tin into the ocean, as scavenging is documented during some time of the year, and a release of dissolved tin from particulates in the estuary is not supported by our data (Fig. 1a).

The behavior of the methyltin compounds in this estuary also showed some variation between sampling periods. Generally, there is a significant production of dimethyltin in the saline end of the estuary (Fig. 1b). Whether this reflects a production in the water column or input from the sediments is not clear. The low-salinity part of the estuary usu-

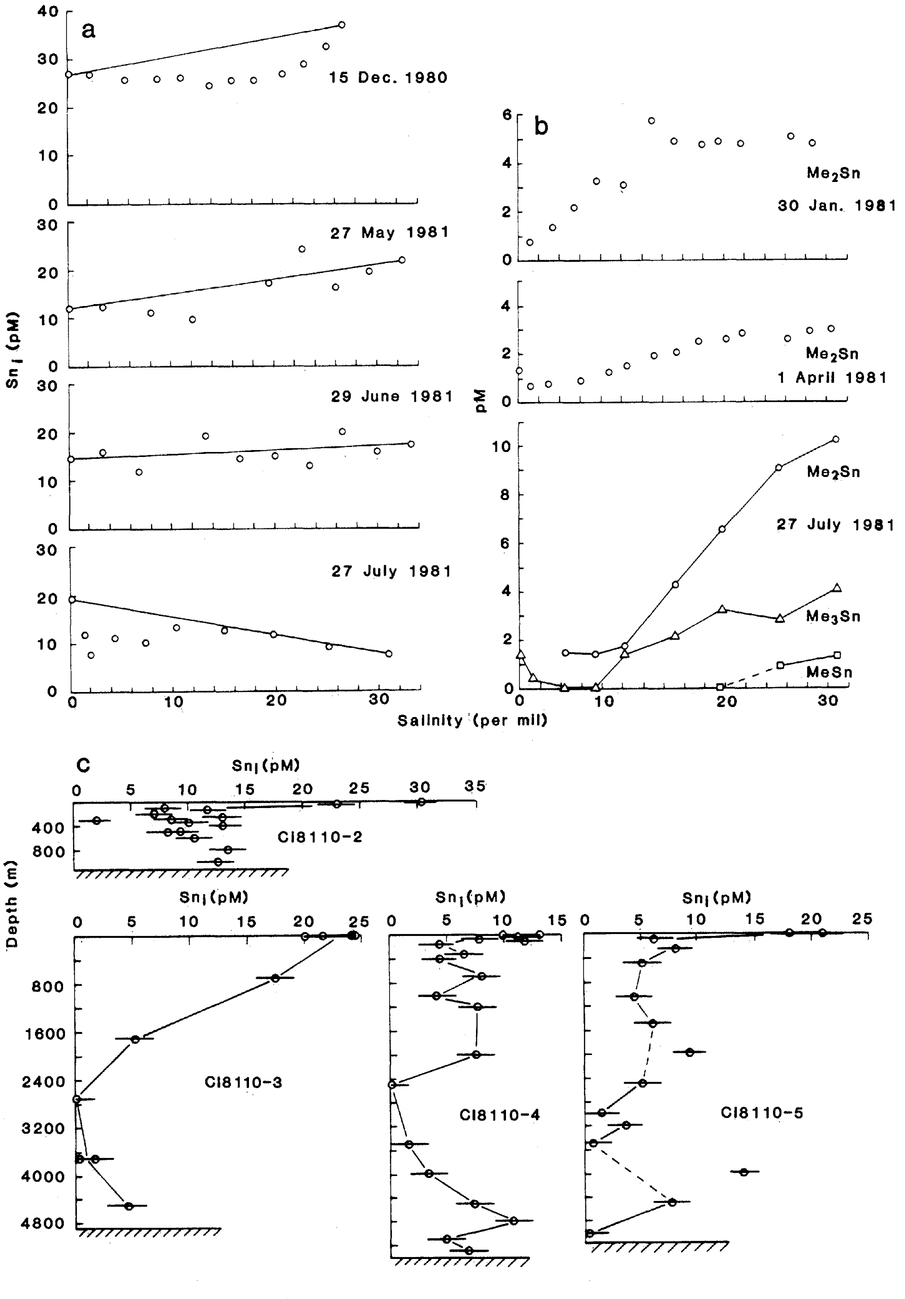

Fig. 1. (a) Profiles of inorganic tin concentrations versus salinity from the Ochlockonee River estuary. Straight lines connecting the fresh and saline end-members represent the conservative mixing model. Within experimental error (± 10 percent relative standard deviation), the data points fall on or below this line. This is evidence that tin is at some times removed by estuarine processes, while at other times no changes other than mixing of the end-members occur. At no time is there evidence for an estuarine source of tin, for example, by benthic inputs. (b) Profiles of methyltin species in the Ochlockonee River estuary. (c) Vertical profiles of the distribution of inorganic tin at four stations in the Gulf Stream (CI8110-2: 27°45′N, 77°58′W) and the Sargasso Sea (CI8110-3: 28°50′N, 75°08′W; CI8110-4: 30°05′N, 71°06′W; CI8110-5: 30°23′N, 69°00′W). Horizontal bars represent an estimate of analytical uncertainty based upon 1 standard deviation of the replicate determinations made on surface water samples.

ally shows evidence of some consumption of dimethyltin. The concentrations of monomethyl- and trimethyltin usually are significantly below those of dimethyltin and do not display a discernible trend within the estuary. An exception is the data from 27 July 1981, when trimethyltin reached relatively high concentrations in the saline end of the estuary and there was evidence of consumption in the low-salinity part of the estuary. The sum of the concentrations of the methyltin species in the saline end-member on this date exceeds the concentration of inorganic tin by a factor of 2. In contrast to other transects, this section was collected at low tide. The increase in the methylated tin concentration with the surface-to-volume ratio of the estuary would suggest a benthic origin for at least part of the methyltins. This is analogous to the conversion of inorganic mercury to methylmercury in sediments (*8*).

The vertical distributions of inorganic tin at four stations across the Gulf Stream into the Sargasso Sea are shown in Fig. 1c. Three features are prominent in these profiles: (i) a strong surface maximum, often with a steep downward decreasing gradient in the upper few hundred meters; (ii) a minimum at about 3000 m, where tin concentrations are at or below the detection limit; and (iii) an increase in the deepest part of the profiles, with an indication of a maximum a few hundred meters above the bottom.

The surface water concentration of inorganic tin decreases with distance from the continent, from a value of 39 p*M* nearshore to 10 p*M* 1100 km from shore, with the exception of one surface sample at the station farthest from land (20 p*M* 1300 km from shore). This result suggests that the continent is the source of the surface water maximum observed in the profiles. The transport into the open ocean could be either by lateral advection of tin originating from rivers or diffusing out of shelf sediments or by atmospheric transport. There is, however, no evidence for a release of significant amounts of tin from particulate matter or from the sediments in our estuarine data, so that a source from the shelf sediments does not appear likely. Only when the water column overlying the sediments becomes anoxic is there evidence for a benthic input of tin (*4*).

The behavior of tin in the deep Atlantic is somewhat more difficult to interpret because this oceanic region is dominated by advective behavior (*9*) and because we have no information on the concentrations of tin in the source regions of the intermediate and deepwater masses. Thus, a minimum at 3000 m could be due to the fact that the anthropogenic signal has not yet propagated to this depth. The lead profiles of Schaule and Patterson (*10*) in the Sargasso Sea parallel our tin profiles, displaying high surface concentrations that decrease rapidly below 1000 m. The similarities between the upper water profiles for these two elements argue strongly for an anthropogenic input of tin from the atmosphere comparable to that demonstrated for lead (*10*).

We suggest that the vertical distribution of tin below the minimum near 3000 m is controlled by advective processes. Recent work on the distribution of Freon 11 and Freon 12 in the northwestern Atlantic (*11*) has shown (at latitudes comparable to our stations) a minimum at about 3000 m, reflecting high water mass age. A maximum is found near 4000 m, caused by rapid advection of recently downwelled sub-Arctic surface water to this level (North Atlantic deep water). A near-bottom minimum with no detectable fluorocarbons is caused by the influence of the Antarctic bottom water. This distribution of the anthropogenic tracer Freon is closely reflected by tin, for which we also suggest a largely anthropogenic source. Both the existence of low tin concentrations just above the bottom and the absence of evidence for benthic inputs for inorganic tin in estuarine environments argue against the possibility that elevated deepwater concentrations of tin result from tin input from the sediment surface, in contrast to the behavior of copper (*12*).

On the basis of thermodynamic calculations, the dominant tin species expected in seawater is $\mathrm{Sn(OH)_4^0}$(aq) (*13*). Assuming that tin behaves like a geochemically closely related element, thorium,

Table 1. Concentrations (in picomoles per liter) of tin species in some U.S. and European rivers: $\mathrm{Sn_i}$, inorganic tin; MeSn, monomethyltin; $\mathrm{Me_2Sn}$, dimethyltin; $\mathrm{Me_3Sn}$, trimethyltin; ΣSn, total tin; N.D., not determined.

Date	River	Location	$\mathrm{Sn_i}$	MeSn	$\mathrm{Me_2Sn}$	$\mathrm{Me_3Sn}$	ΣSn
13 March 1981	Trinity	Liberty, Texas	11.0	N.D.	1.7	1.7	14.4
14 March 1981	Sabine	Deweyville, Texas	9.3	Trace	0.8	N.D.	10.1
14 March 1981	Neches	Beaumont, Texas	23.6	1.7	0.8	N.D.	26.1
13 March 1981	Colorado	Wharton, Texas	21.1	Trace	1.7	1.7	24.5
14 March 1981	Pascagoula	Benndale, Mississippi	7.6	N.D.	Trace	N.D.	7.6
15 March 1981	Escambia	Century, Florida	13.5	0.8	0.8	1.7	16.8
14 March 1981	Pearl	Highway 59, Louisiana	20.2	0.8	0.8	N.D.	21.8
14 March 1981	Mississippi	Port Allen, Louisiana	1036	11.8	26.1	0.8	1075
14 March 1981	Atchafalaya	Krotz Springs, Louisiana	57.3	1.7	3.4	N.D.	62.4
7 September 1980	Chipola	Highway 20, Florida	110.4	0.8	0.8	N.D.	112.0
7 September 1980	Apalachicola	Highway 20, Florida	47.2	0.8	0.8	1.7	50.5
23 September 1980	Flint	Albany, Georgia	79.2	N.D.	Trace	N.D.	79.2
24 September 1980	St. Marys	State Line, Florida	65.7	1.7	N.D.	N.D.	67.4
24 September 1980	Suwannee	Fargo, Georgia	75.8	4.2	N.D.	N.D.	80.0
24 September 1980	Suwannee	Route 6, Georgia	75.8	1.7	Trace	N.D.	77.5
24 September 1980	Suwannee	Ellaville, Florida	219.1	Trace	N.D.	N.D.	219.1
7 September 1980	Ochlockonee	Highway 20, Florida	35.4	1.7	0.8	0.8	38.7
26 October 1980	Ochlockonee	Highway 20, Florida	48.0	N.D.	N.D.	Trace	48.0
26 October 1980	Ochlockonee	Downstream of Highway 20, Florida	24.4	N.D.	N.D.	Trace	24.4
26 October 1980	Ochlockonee	Downstream of Highway 20, Florida	28.6	N.D.	N.D.	Trace	28.6
26 October 1980	Ochlockonee	Downstream of Highway 20, Florida	19.4	N.D.	N.D.	N.D.	19.4
26 October 1980	Ochlockonee	Downstream of Highway 20, Florida	27.0	N.D.	N.D.	N.D.	27.0
8 August 1980	Ochlockonee	Highway 90, Florida	107.8	N.D.	3.4	14.3	125.5
18 August 1980	Tuckaseegee	Bryson City, North Carolina	6.7	N.D.	N.D.	N.D.	6.7
18 August 1980	Nantahala	Nantahala, North Carolina	N.D.	N.D.	N.D.	N.D.	N.D.
19 August 1980	Hiwassee	Murphy, North Carolina	1.7	N.D.	N.D.	N.D.	1.7
12 April 1981	Rhine	Oppenheim, Germany	155.0	592.3	1769	13.5	2530
13 April 1981	Main	Frankfurt, Germany	101.1	N.D.	3.4	2.5	107.0

Table 2. Estimated fluxes of tin into the atmosphere from various sources.

Source	Global production or consumption (10^{12} g/year)	Emission factor (g/g source)	Tin flux (10^9 g/year)
	Anthropogenic input		
Coal burning	3245	0.2×10^{-6} (19)	0.65
Coal burning	3245	0.17×10^{-6} (16)	0.55
Oil burning	1600	0.01×10^{-6} (19)	0.002
Wood and agricultural burning	1320	0.75×10^{-6}* (20)	0.17
Waste incineration	540	8.6×10^{-5} (21)	47
Iron and steel production	1220	0.1×10^{-6}† (1)	0.12
Nonferrous metal production	18	3.5×10^{-4}† (1)	6.3
Tin production	0.24	0.005‡ (22)	1.2
Organotins	0.008	0.05	0.4
Estimated range			10 to 60
	Natural input		
Sea spray	1000	2.8×10^{-10}	0.0003
Soil dust	800	0.15×10^{-5} (14)	1.2
Volcanoes	25	0.96×10^{-5} (15)	0.24
Forest fires	320	0.75×10^{-6} (19)	0.24
Biomethylation			6?
Estimated range			2 to 10
Total flux			12 to 70

*Volatilization efficiency assumed to be 10 percent. †Emission efficiency assumed to be 10 percent. ‡Volatilization efficiency assumed to be 50 percent.

and is removed by particulate scavenging with a half-life of ~ 3.5 years in the upper 350 m (*14*), we estimate a removal flux from the surface ocean (with an average concentration of 10 p*M*) of 22×10^9 g/year. This removal rate must be balanced by riverine and eolian inputs. We have no data yet on the concentration of tin in the suspended load of rivers, but a lower limit for the concentration of tin in suspended matter is likely to be that in shale, 2 parts per million.

The absence of a clear connection between the general state of pollution of the rivers investigated and the concentrations of dissolved tin observed (with the exception of the Mississippi for inorganic tin and the Rhine for dimethyltin) suggests that most of the pollutant tin in rivers becomes quickly attached to suspended particulates. Assuming (somewhat arbitrarily) that about 20 percent of the world consumption of tin, 236×10^9 g/year, enters rivers and becomes attached to particulates, we estimate a total particulate flux of 83×10^9 g/year, compared with a dissolved flux of 0.22×10^9 g/year. Our estuarine data suggest that the particulate fraction of the riverine flux is not redissolved in the estuarine and marine environment. Essentially all the particulate flux is deposited on the continental shelves and does not further contribute to the marine geochemical cycle of tin. Thus only the dissolved tin in rivers contributes significantly to the oceanic mass balance of this element. This flux is two orders of magnitude lower than the scavenging removal flux estimated above. We propose that this difference between riverine input and scavenging output is due to a large eolian flux.

Tin is one of the ten most highly enriched elements in atmospheric particulates (*5*). On the basis of a compilation of data from various sources and recent careful measurements from Barrow, Alaska (*5*), an enrichment factor (relative to the crustal average) of about 1000 has been calculated. The concentrations of particulate tin in air are some tens of nanograms per cubic meter of air in polluted environments, about 1 ng/m^3 in Northern Hemisphere marine air (*5*), and about one to two orders of magnitude lower in Southern Hemisphere marine air (this estimate must be considered very crude). Using these data, we derive a deposition estimate of 55×10^9 g/year based on the deposition velocities for wet and dry deposition of 0.6 cm/sec (*15*) and an estimate for the mean concentration of tin in atmospheric particulates of 1 ng/m^3 for the Northern Hemisphere (*5*) and 0.1 ng/m^3 for the Southern Hemisphere. In order to verify this estimate and to identify important sources, we have compiled source fluxes of tin to the atmosphere, using consumption values given by Walsh *et al.* (*16*) in their discussion of the arsenic cycle together with tin emission factors (Table 2). Because of the scarcity of data for tin as compared to those for arsenic, these emission factors are often very rough estimates. However, we believe that for most values the error should be not more than an order of magnitude. The anthropogenic emissions appear to be dominated by waste incineration and nonferrous metal production (in high-temperature processes tin is very volatile). The amount accounted for by waste incineration is about 20 percent of the tin manufactured. As this percentage is higher than the fraction of waste being incinerated rather than ending up in landfill, it is probable that this value is an overestimate; the real value may be lower by up to a factor of 10. In view of the uncertainties in the fluxes discussed above, we suggest a range of 10×10^9 to 60×10^9 g/year for the anthropogenic input of tin.

The natural inputs of tin into the atmosphere are also very difficult to estimate because of the scarcity of data. However, all natural sources except biomethylation are relatively unimportant (Table 2) when compared to the anthropogenic flux. The biomethylation source is based on the observation of Braman and Tompkins (*3*), who have found rainwater in a Florida coastal location to contain significant amounts of tin in methylated forms: their average tin concentration was 14 ng/liter in the form of mono-, di-, and trimethyltin. Because there are no obvious anthropogenic sources for this material, it is likely that biomethylation of tin in soils is the source of these methyltin compounds.

Assuming the concentration given above to be globally representative, we estimate a wet deposition tin flux of 6×10^9 g/year in the form of methyltins. This estimate is very crude since it ignores geographic variations in source strength and alternate removal modes, but it points out the possibility that a natural flux through biomethylation may be an important, if not the dominant, natural source of tin to the atmosphere.

The total tin flux is thus in the range of 12×10^9 to 70×10^9 g/year, which is in good agreement with the estimate of Lantzy and Mackenzie (*17*) of 43×10^9 g/year, derived on a somewhat different set of assumptions. All these atmospheric flux estimates are in general agreement with the particulate scavenging flux for ocean water of 22×10^9 g/year.

The recent addition of large amounts of tin to the nearshore sediments makes this element useful as a tracer of man's activities. Tin concentrations in the sediments of Lake Michigan and Narragansett Bay (*18*) document a large increase of tin fluxes as a consequence of increased industrial activity. These data are consistent with our conclusion that the present fluxes of tin through the atmosphere and surface ocean are dominated by anthropogenic effects.

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Bronchial Hyperreactivity Associated with Tracheal Gangliosides

Abstract. *Gangliosides, which are membrane constituents of animal cells, may be altered under various conditions that cause change in metabolism. In this study, gangliosides from tracheal and lung tissues were extracted and measured as a function of bronchial hyperreactivity in a guinea pig model of bronchial asthma. When plotted logarithmically, the data showed that tracheal gangliosides decreased with an increase in airway reactivity index, suggesting a linear relation between them. Differential analysis of tracheal gangliosides in the acute stage of bronchial hyperreactivity indicates accumulation of polysialogangliosides. These data support the hypothesis that tracheal gangliosides are intimately involved in the development of bronchial asthma.*

Bronchial asthma is characterized by increased responsiveness of the trachea and bronchi to various stimuli and is manifested by a narrowing of the airways. The severity of the disease may change either spontaneously or as a result of therapy (*1*). Nearly 7 percent of the population in the United States (*2*) and 1.5 percent in India have this disease. The etiology of asthma is unknown, but the lungs of asthmatic patients release histamine both in vitro (*3*) and in vivo (*4*) when challenged with an appropriate allergen, and the plasma histamine concentration is increased in spontaneously occurring asthmatic attacks (*5, 6*). Confirmation of the proposal by Ash and Schild (*7*) that at least two types of receptors (H1 and H2) are involved in the histamine response has caused a renewed interest in histamine in the pathogenesis of bronchial asthma (*8*). Because it is the airway muscle that is ultimately mobilized during anaphylactic bronchospasm, the present studies were designed to test the hypothesis that the defect in the guinea pig model of asthma may stem from a specific abnormality of a cell surface component of the airway smooth muscle.

The guinea pig model of allergic asthma was developed by Agrawal (*9*), who monitored airway reactivity to histamine. Agrawal recorded three parameters, that is, (i) airflow at nares, dV_2/dt; (ii) the difference between the change in chest cage volume and the air volume respired at the nares ($V_1 - V_2$); and (iii) the difference between the rate of change in chest cage volume and the airflow at the nares, $d(V_1 - V_2)/dt$ (Fig. 1). The airway reactivity index (ARI) is the ratio of the peak values of $d(V_1 - V_2)/dt$ after and before histamine inhalation. The peak values of $d(V_1 - V_2)/dV_2$ were calculated by dividing the magnitude of dV_2/dt waveform at the corresponding points.

Gangliosides, a subclass of glycosphingolipids, are constituents of the surface membrane of mammalian cells and may be altered under various conditions that cause changes in metabolism (*10, 11*). Moreover, gangliosides have receptors for tetanus toxin, interferon, Sendai virus, and cholera toxin (*12, 13*). They also appear to bind 5-hydroxytryptamine (*12, 14*).

In this study, total gangliosides from guinea pig tracheal tissue were extracted

Table 1. Glycosphingolipid composition (per gram of tissue, wet weight) of guinea pig trachea and lung.

Glycosphingolipids	Trachea		Lung	
	Normal	Hyperreactive	Normal	Hyperreactive
Gangliosides (nmole of sialic acid)	2422.62 ± 808.52 (8)	398.47 ± 38.41*(6)	71.91 ± 22.59 (6)	65.83 ± 17.15 (6)
Sphingomyelin (μg)	141.67 ± 29.15 (8)	115.03 ± 30.50 (6)	202.16 ± 17.84 (8)	210.17 ± 20.84 (7)
Cerebroside (μg)	268.40 ± 26.90 (8)	269.70 ± 40.10 (7)	21.46 ± 1.22 (7)	21.48 ± 4.33 (6)
Sulfatide (μg)	110.40 ± 25.50 (8)	132.40 ± 22.50 (7)	1.15 ± 0.09 (7)	1.42 ± 0.13 (6)

*$P < .001$. Animals with an ARI between 1.1 and 2.75 were grouped as normal and those between 3.0 and 5.38 were grouped as hyperreactive. Total lipids and then gangliosides were extracted from trachea and lung tissues as described in Fig. 1. After removal of the upper phase lipid for gangliosides, the lower phase was dried under a stream of N_2. Ten to fifteen milligrams of lower phase lipid was dissolved in 1 ml of 0.5*N* methanolic sodium hydroxide. The solution was allowed to stand for 4 hours at room temperature, then 1 ml of 1.5*N* methanolic hydrochloric acid was added and the solution was mixed thoroughly and left for 30 minutes. To this acidified solution, 3 ml of chloroform and 1 ml of distilled water were added and mixed. The upper aqueous phase was discarded, the lower phase was washed twice with methanol and water, 1:1 by volume (1 ml). Sphingomyelin was separated on silica gel H in chloroform, methanol, and water (60:25:4 by volume) and the lipid phosphorus was determined (*22*) [micrograms of phosphorous × 25 = micrograms of sphingomyelin (*23*)]. The cerebrosides and sulfatides were isolated by column chromatography with a combination of Florisil and DEAE-cellulose (acetate) (*24*). The results are expressed as means ± standard error. The significance of the results is analyzed by Student's *t*-test. The number of animals used in each experiment is given in parentheses.

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