

Measuring low concentrations of ^{234}Th in water and sediment

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Abstract

$^{234}\text{Th}/^{238}\text{U}$ disequilibria have been used extensively in studies of particle dynamics and the fate and transport of particle-reactive matter in marine environments. Similar work in low salinity, estuarine, and freshwater systems has not occurred primarily because the lower concentrations of both parent and daughter nuclides that are typical of these systems often render established methods for the analysis of ^{234}Th inadequate. The application of this radionuclide tracer technique to these systems, however, has great potential. To this end, we present a method for measuring low activities of ^{234}Th in relatively small samples (<200 l) using low background gas-flow proportional counters, a ^{229}Th yield monitor, and empirical corrections for the interferences from real and apparent betas that are emitted by other thorium isotopes and their progeny. For samples with low $^{234}\text{Th}/^{228}\text{Th}$ activity ratios, we improve upon current beta counting methodologies that rely on immediate sample counting, weak beta absorption, or multiple beta counts so that, using the analytical approach outlined here, it should be possible to measure ^{234}Th activities (i) as low as 1.5 dpm/total sample, (ii) up to 2 weeks after radiochemical purification of thorium, and (iii) with only one sample count for alpha and beta activity.

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1. Introduction

In marine environments, particle-reactive scavenging rates (e.g., Coale and Bruland, 1985; Gustafsson et al., 1997; Bacon and Anderson, 1982), sediment trap efficiencies (e.g., Buesseler et al., 1994), surface sediment mixing (e.g., Aller and Cochran, 1976; Carpenter et al., 1984), and vertical and horizontal particle transport rates (e.g., Kershaw and Young, 1988; McKee et al., 1984; Feng et al., 1999) have

all been calculated in part by measuring disequilibria between naturally occurring, particle-reactive ^{234}Th and its generally soluble parent, ^{238}U . In many brackish and freshwater systems, however, ^{238}U and ^{234}Th activities are an order of magnitude or more lower than those found in marine systems and difficulties arise in accurately assessing the activity of ^{234}Th because of limitations in the established methods used to measure its activity.

Established methods used to measure ^{234}Th differ not so much with respect to thorium concentration and isolation (see, e.g., Rutgers van der Loeff and Moore, 1999) but rather by mode of detection. ^{234}Th ($t_{1/2} = 24.1$ days) emits gamma (γ) radiation immediately

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following its decay to ^{234m}Pa by beta (β) emission. ^{234m}Pa ($t_{1/2} = 1.2$ min) subsequently decays by beta emission to ^{234}U . These emissions can be measured using gamma spectrometers and gas-flow proportional (beta) counters.

Each method of detection (γ and β) has its advantages and drawbacks. On one hand, sample preparation for gamma spectrometry is relatively simple and nondestructive. Gamma spectrometry can also identify and quantify individual radionuclides. However, low photon abundance combined with relatively low detector efficiencies typically result in only ~ 1.5 γ cpm per 100 dpm of ^{234}Th (Buesseler et al., 1992). Consequently, the measurement of ^{234}Th by gamma spectrometry has required a sample size of at least 500–5000 l of seawater. To extend this method to freshwater, where ^{234}Th activities can be over an order of magnitude lower, would require a proportionate increase in sample size. Handling samples of this size on the small research vessels or at remote land-based sites commonly encountered in sampling estuarine and freshwater systems would be, at best, cumbersome, if not very difficult.

Low background gas-flow proportional counters, on the other hand, are very efficient at detecting beta radiation with combined counting efficiencies near 100% for the $^{234}\text{Th} + ^{234m}\text{Pa}$ pair. High detector efficiencies and low instrument backgrounds permit very low detection limits for these beta-emitting nuclides so that current ^{234}Th beta counting methods require a sample size of only 2–20 l of seawater (Buesseler et al., 2001; Benitez-Nelson et al., 2001). Unlike gamma spectrometry, however, individual beta sources cannot be distinguished through proportional counting alone.

The major problem associated with beta counting ^{234}Th in natural samples is the unknown contribution of beta counts from (i) the ingrowth of beta producing progeny from ^{228}Th (and ^{229}Th , if added as an isotopic diluent); and (ii) internal conversion electrons from the alpha decay of ^{228}Th , ^{229}Th , ^{230}Th , and ^{232}Th . The contribution of extraneous (i.e., non- $^{234}\text{Th} + ^{234m}\text{Pa}$) and apparent (i.e., internal conversion electron) beta counts to the gross beta count will vary, of course, depending on the sample source and matrix. In the open ocean, where ^{234}Th is nearly in secular equilibrium with its parent, the activity ratio of ^{234}Th to other naturally occurring thorium isotopes is on the

order of 10^3 – 10^4 (Broecker and Peng, 1982). Under these conditions, interference from the naturally occurring, alpha-emitting thorium isotopes to the $^{234}\text{Th} + ^{234m}\text{Pa}$ beta count can be assumed to be negligible. However, at very low ^{234}Th activities found in certain estuarine and freshwater environments, the activity ratio of ^{234}Th to other naturally occurring thorium isotopes decreases substantially and extraneous and apparent beta counts originating from these other thorium isotopes must be taken into account.

Corrections for these non- $^{234}\text{Th} + ^{234m}\text{Pa}$ beta counts are typically made in one of three ways which we will define as (1) the immediate sample count method, (2) the weak beta absorber method, and (3) the multiple beta count method.

The “immediate sample count” method relies simply on counting the sample immediately after thorium separation and preparation of the beta source (planchet). While this minimizes the ingrowth of extraneous (non- $^{234}\text{Th} + ^{234m}\text{Pa}$) beta counts originating from ^{228}Th (and ^{229}Th) progeny, it does not take into account the contribution of apparent beta counts from internal conversion electrons.

The “weak beta absorber” method relies on counting only the high-energy betas of ^{234}Th 's daughter, ^{234m}Pa ($E_{\text{max}} = 2.29$ MeV), by selective removal of weak energy betas through the use of an absorber (e.g., a piece of aluminum foil) between the sample and gas-flow proportional counter. This absorber method, however, is not appropriate for samples containing any significant activity of ^{228}Th or ^{229}Th because both isotopes rapidly produce progeny that also give off high-energy betas (e.g., ^{212}Bi : $E_{\text{max}} = 2.25$ MeV; ^{213}Bi : $E_{\text{max}} = 1.42$ MeV) that cannot be selectively absorbed or distinguished in the presence of ^{234m}Pa (Baltakmens, 1977). Decreasing the $^{234}\text{Th} + ^{234m}\text{Pa}$ beta signal will also necessitate an increase in sample size at very low ^{234}Th concentrations.

The “multiple beta count” method requires counting the sample beta source repeatedly (e.g., two to six times) over a period of up to six half-lives of ^{234}Th (~ 145 days). By fitting an exponential curve with a decay rate of 24.1 days to the gross beta count (Buesseler et al., 2001), or alternatively, by plotting the gross beta count rate of the thorium sample as a function of $e^{-\lambda t}$, where λ is the decay constant of ^{234}Th and t is the time elapsed since separation of

^{234}Th from its parent ^{238}U in the sample (Aller and Cochran, 1976), the initial activity of ^{234}Th in the sample can be calculated from the slope of the decay line. However, for samples containing relatively high activities of ^{228}Th , there are constraints on the timing of the initial beta count. Because the ingrowth of beta producing nuclides from ^{228}Th (and to a lesser extent ^{229}Th) has a similar time scale as the rate of ^{234}Th decay, the true slope of ^{234}Th decay will be masked until the other beta producing nuclides reach transient (or secular) equilibrium. For ^{228}Th , this will take ~ 3 weeks. Determining the activity of a single sample using the multiple-count method, therefore, may take months. Furthermore, the initial activity of ^{234}Th in the sample must be high enough so that it can be counted at least 3 weeks after sample collection. If not, the sample size must increase.

In view of these limitations, we present here a new approach to measuring low concentrations of ^{234}Th in samples with low $^{234}\text{Th}/^{228}\text{Th}$ activity ratios by directly assessing beta interferences from each of the alpha-emitting thorium isotopes in the sample.

2. Experimental section

A flowchart showing our approach to measure both dissolved and particle-bound ^{234}Th fractions in a water (or sediment) sample is presented in Fig. 1.

2.1. Sample collection and preparation

An initial estimate of the amount of water and sediment needed to measure ^{234}Th may be determined by first measuring the dissolved ^{238}U activity in the sample matrix (Kaplan et al., 1994; Edgington et al., 1996). In Lake Michigan, where samples for this study were collected, ^{238}U activities in the water averaged 0.23 ± 0.02 dpm l^{-1} ($n=12$). Accordingly, we ascertained that 100–200 l of Lake Michigan water would be more than sufficient to measure both dissolved and particle-bound ^{234}Th fractions.

Water samples (200 l) were collected using a submersible pump and stored briefly in 50-l plastic containers until separation of the phases by filtration. The particle-bound thorium fraction on suspended sediment was separated from the water sample by filtration through pre-weighed nitrocellulose filters

(0.45 μm , 293 mm, Millipore)—usually within 2 h of collection. The time of filtration (t_{1a}) was noted and the filter was stored at 4 °C until it could be dried at 60 °C and weighed to determine the mass of total suspended solids.

One hundred liters of the filtrate was placed in a 200-l plastic conical container for the recovery of the dissolved thorium fraction (operationally defined by the filter pore size). The thorium was co-precipitated onto newly formed iron hydroxide and the precipitate immediately separated by filtration. In detail, the initial filtrate was acidified with 500 ml of concentrated HCl (12 M), and the requisite aliquot portion of the isotopic diluent, ^{229}Th (e.g., 0.4 dpm), was added and stirred vigorously for 15 min. Next, 1 mg l^{-1} ferrous iron was added as ferrous sulfate (i.e., 0.5 g) and the solution was again well stirred for a few minutes. Ferric hydroxide was precipitated by adding 500 ml of concentrated NH_4OH (14 M) and this precipitate was immediately collected onto a nitrocellulose filter (0.45 μm , 293 mm, Millipore). The time of filtration (t_{1b}) was recorded and the filter was stored at 4 °C. The formation of ferric hydroxide by the oxidation of Fe^{2+} produces a more easily filterable precipitate than one produced directly from the addition of Fe^{3+} , which tends to clog the filters.

2.2. Chemical separation of thorium

Filters carrying the small amounts of either the particulate or precipitated thorium fractions were moistened with deionized water and combusted in a muffle furnace at 520 °C for 8 h.

2.2.1. Dissolved fraction

The ashed ferric hydroxide precipitate was dissolved in 20 ml of 6 M HCl in a 250-ml glass beaker and gently evaporated to a moist residue. The residue was then taken up in 20 ml of 8 M HNO_3 .

2.2.2. Particle-bound fraction

The ashed sediment fraction was reweighed to determine sample loss on ignition and transferred to a 600-ml glass beaker. The requisite isotopic diluents (e.g., 0.4 dpm of ^{229}Th and 0.2 dpm of ^{236}U) and 50 ml of 6 M HCl were then added to the sediment sample. The beaker was covered with a watch glass

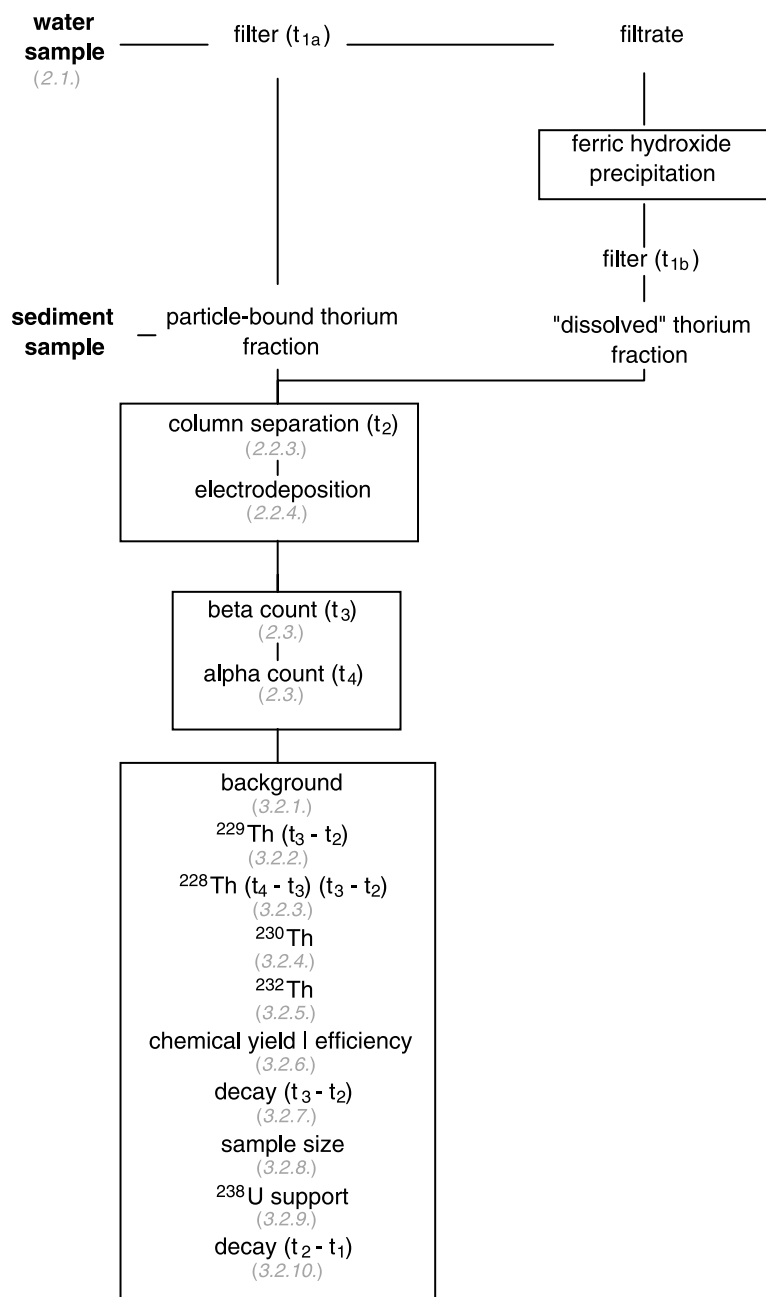


Fig. 1. A procedure for measuring low concentrations of ^{234}Th in water and sediment. Section numbers for each step of the procedure in the text are shown in brackets.

and heated to just below boiling point for 8 h. The leachate was then centrifuged to remove any insoluble residue. Another 50-ml aliquot of 6 M HCl was added to the once-leached sediment and the leaching process

was repeated. The combined ~ 100 ml of leachate was evaporated to a moist residue and then re-dissolved in 20 ml of 8 M HNO_3 prior to column separation.

2.2.3. Separation of thorium using ion-exchange

Thorium was isolated from uranium and all other elements by separation on an anion ion-exchange column (Edgington et al., 1996; Faris and Buchanan, 1964). Disposable small columns were prepared using glass Pasteur pipettes (8.5×0.6 cm, Fisher Scientific) and filled with anion exchange resin (Bio-Rad, AG1-X8, 100–200 mesh) to give a 7-cm column (approximately 0.8 g of resin). The anion exchange resin column was first converted to the nitrate form using a 5-ml aliquot portion of 8 M HNO_3 . The entire sample was then transferred to the top of the column and allowed to elute before washing with an additional 20 ml of 8 M HNO_3 in four 5-ml aliquot portions. The eluate was saved for later analysis of the small amount of ^{238}U carried along with the thorium in the precipitation (dissolved fraction) or leaching (particle-bound fraction) stage.

Only thorium and plutonium were retained on the resin at this point. Thorium was separated from plutonium and eluted from the column by washing with 20 ml of concentrated HCl (12 M) in four 5-ml aliquot portions. The time of elution (t_2) was noted, since ingrowth of thorium progeny began at this point.

2.2.4. Electrodeposition

The separated thorium isotopes were finally transferred to a stainless steel counting planchet by electro-deposition to provide an essentially weightless source. In detail, the concentrated HCl solution containing the thorium fraction was evaporated to near dryness, redissolved in 16 ml of a 1 M ammonium chloride–0.01 M oxalic acid plating solution, transferred to a plating cell and electroplated (20 V, 0.85 A, 60 min) onto a stainless steel planchet using the technique of Puphal and Olsen (1972). Typically, using all steps in this method, from the collection of water to electrodeposition, the efficiency of recovery, based on separate experiments with ^{229}Th , is better than 95%.

2.3. Sample counting

The sample planchet was first counted using a low background (~ 0.7 cpm) gas-flow proportional counter with 2 1/4-in.-diameter detectors and anti-coincidence circuitry (G542 System, Gamma Products) for four 500-min periods to obtain total (gross) beta counts. The planchet was then counted using a silicon

surface barrier detector to determine the activities of ^{228}Th , ^{229}Th , ^{230}Th , and ^{232}Th by α -spectrometry. Total alpha count times were activity-dependent, but typically ranged between 7000 and 14,000 min.

2.3.1. Corrections for beta nuclide ingrowth and other interferences

Beta counts originating from the ingrowing progeny of ^{228}Th and ^{229}Th and apparent beta contributions from internal conversion electrons provide the sources of a positive bias to the $^{234}\text{Th} + ^{234\text{m}}\text{Pa}$ beta count. To correct this bias, standard solutions of ^{228}Th , ^{229}Th , ^{230}Th , and ^{232}Th (with some ^{228}Th present) were first purified using anion exchange (as outlined in Section 2.2.3) and electrodeposited onto separate stainless steel planchets. Each planchet containing a freshly isolated thorium isotope was then immediately placed into a gas-flow proportional counter and counted, repeatedly, for 500-min periods over an interval of 14 days (or longer). The planchets were finally counted using α -spectrometry in detectors of known efficiency to determine specific plate activities of each thorium isotope. These counts were then used to prepare correction algorithms for the beta count contribution from each α -active isotope to the gross beta count.

2.3.2. Detector efficiencies

Detector efficiencies were evaluated for both gas-flow proportional (β) and silicon surface barrier (α) detectors. To accomplish this, a secondary standard of isotopically pure ^{238}U was first isolated from its ^{234}Th progeny on an anion exchange column in 9 M HCl. The pure ^{238}U was then electrodeposited onto a stainless steel planchet and counted in a NIST calibrated (SRM 4904L-F-72) 2π alpha counter (windowless type). The calibrated ^{238}U planchet was then counted in each of the α -detectors to determine specific detector efficiencies. Next, the calibrated ^{238}U planchet (with ingrowing $^{234}\text{Th} + ^{234\text{m}}\text{Pa}$) was counted repeatedly for 10 or more 500-min cycles in each beta detector. The gross beta count rate for each counting cycle was plotted against $1 - e^{-\lambda t}$ where λ equals the decay constant for ^{234}Th and t is equal to the elapsed time (in days) between the column separation of uranium and the beta count. The effective efficiency of the beta detector (cpm/dpm) was then calculated as the slope of the ingrowth line (which equals the beta count rate (cpm) of ^{234}Th in secular equilibrium with

its ^{238}U parent on the planchet) divided by the known activity of ^{238}U (dpm) on the planchet.

3. Results and discussion

3.1. Sample preparation and chemical separations

There are a number of methods that can be used to concentrate and isolate thorium from the sample matrix (see, e.g., Rutgers van der Loeff and Moore, 1999; Buesseler et al., 2001). Many of these methods, however, are truly applicable only in the open ocean. While the approach we outline here may seem labor-intensive, there are few, if any, shortcuts that can be taken with the analysis of samples containing low concentrations of ^{234}Th . In particular, we suggest that the co-precipitation of thorium in the dissolved phase and isotope dilution with ^{229}Th followed by radiochemical separation of thorium by ion-exchange chromatography is the simplest and most accurate approach to measuring ^{234}Th in low salinity and freshwater systems.

While it has been shown that manganese dioxide-impregnated nylon filters are effective in extracting thorium from seawater (Buesseler et al., 1992), this technique is not totally applicable for measurements of ^{234}Th in freshwater systems. Freshwater systems are more diverse and geochemically heterogeneous than oceanic waters, and the calibration of MnO_2 filters would be complicated by: (1) the variable composition in terms of DOM and alkalinity which strongly affect the formation of both Th and U complexes and, thus, adsorption to the filter; (2) the effect of the presence of anoxic water—which is common in small and large lakes—on adsorption; and (3) the absence of a reliable water depth where ^{234}Th may be assumed to be in secular equilibrium with ^{238}U for standardization.

The dissolved ^{234}Th fraction can be scavenged by precipitation of either $\text{Fe}(\text{OH})_3$ or MnO_2 . Experiments in our laboratories have shown that using 1 mg l^{-1} of Fe^{3+} results in the scavenging of $17.6 \pm 2.6\%$ ($n = 12$) of the ^{238}U in solution in Lake Michigan. This translates to an uncertainty of $\sim \pm 0.18 \text{ dpm m}^{-3} \text{ day}^{-1}$ in supported ^{234}Th ingrowth between the time of filtration of the ferric hydroxide precipitate (t_{1b}) and the time of separation of thorium by ion-exchange

chromatography (t_2), which in most instances is insignificant. In lakes or rivers with higher or lower alkalinity, a lesser or greater fraction of uranium would be carried, respectively. If the amount of uranium carried was problematic, it is possible to reduce the uranium fraction scavenged by adding ammonium bicarbonate to the water prior to acidification. While MnO_2 precipitation has the advantage over $\text{Fe}(\text{OH})_3$ in that it scavenges very little uranium, thereby, eliminating the requirement for a correction for supported thorium, it is a relatively new technique (Rutgers van der Loeff and Moore, 1999) and its effectiveness in natural freshwater systems has not yet been tested.

Both precipitates (i.e., MnO_2 and $\text{Fe}(\text{OH})_3$) scavenge high and reproducible yields of thorium (>95%) and theoretically do not require the addition of a yield monitor. However, because we separate thorium from all other nuclides on an ion-exchange resin column and electroplate the purified thorium onto a stainless steel planchet, a yield monitor is required. Measuring low concentrations of ^{234}Th in samples from low salinity and freshwater systems requires thorium isotope isolation because of the relatively high concentrations of other naturally occurring particle-reactive beta emitting nuclides (e.g., ^{210}Pb , ^{228}Ra). Furthermore, the entire sample must be reduced to a thin, nearly weightless source because it is necessary to measure the activity of alpha-emitting thorium isotopes as well.

^{230}Th and ^{229}Th are commonly used as yield monitors in the recovery of ^{234}Th . While ^{230}Th is often favored because it is a pure alpha emitter whereas ^{229}Th decays to both beta-emitting progeny that inflate the gross beta count and alpha-emitting progeny that can contaminate alpha detectors (Rutgers van der Loeff and Moore, 1999), the use of ^{229}Th here is preferable for several reasons. First, the activity of ^{230}Th can be quite high in freshwater environments. In Lake Michigan surface sediments, for example, the activity of ^{230}Th is approximately 0.7 dpm g^{-1} . This requires adding a substantial activity of ^{230}Th to dilute any small variation in the ambient signal. Even though ^{230}Th is a pure alpha emitter, it decays to the ground state of ^{226}Ra only 76.3% of the time (International Commission on Radiological Protection, 1983). Approximately 23% of all ^{230}Th disintegrations emit an internal conversion electron. At ~ 50 to 70 keV , these electrons have sufficient energy to pass through the ($< 2 \text{ mm}$) air path

between the sample and detector window in our beta counting system as well as the 0.08 mg cm^{-2} detector window itself (personal communication, Gamma Products). Once in the detector, they are counted as beta events (Friedlander et al., 1964). Therefore, unless suitable shielding is added to block these conversion electrons, adding substantial activities of ^{230}Th will necessitate a significant correction to the gross beta count. In addition, we are interested in measuring the ambient activities of all thorium isotopes. Using ^{230}Th as a yield monitor would therefore require us to collect and process an additional sample as well as take extra precautions to avoid cross-contamination in the laboratory. Finally, the problem of ^{229}Th progeny contaminating alpha detectors is not particularly severe. ^{225}Ac , the longest-lived, alpha-emitting daughter of ^{229}Th , has a half-life of only 10 days. Moreover, none of the alpha-emitting daughters of ^{229}Th have energies that interfere with the alpha spectra of any of the thorium or uranium nuclides.

3.2. Corrections to the gross beta count

After a sample planchet is counted, corrections have to be made to the total beta count in order to accurately calculate the original sample activity of ^{234}Th . These corrections are shown in Fig. 1 and are discussed below.

3.2.1. Instrument background correction

The first step was simply to subtract the gas-flow proportional counter background from the gross beta count.

3.2.2. ^{229}Th beta count correction

Next, corrections were made for the contribution of beta counts from ^{229}Th , ^{228}Th , ^{230}Th , and ^{232}Th decay. It was possible to define empirical equations that described this ingrowth of beta counts from separate counts of their respective isotopic sources (as outlined in Section 2.3.1).

Beta counts from ^{229}Th decay (β_{229} , cpm) grew in at an initial rate of:

$$\beta_{229} \text{ (cpm)} = A^{229}\text{Th (dpm)}[0.0297(t_3 - t_2) + 0.2741], \quad (1)$$

for $t_3 - t_2 \leq 14.6$ days (Fig. 2a), where $t_3 - t_2$ was equal to the time elapsed in days between column separation (t_2) and the beta count (t_3), and $A^{229}\text{Th}$ was equal to the activity of ^{229}Th (dpm) on the sample planchet as measured by α -spectrometry.

3.2.3. ^{228}Th beta count correction

Beta counts from ^{228}Th decay (β_{228} , cpm) grew in at an initial rate of:

$$\beta_{228} \text{ (cpm)} = A^{228}\text{Th (dpm)}[0.9688(1 - \exp \times (-0.1635(t_3 - t_2))) + 0.026], \quad (2)$$

where $t_3 - t_2 \leq 14.6$ days in units of days (Fig. 2b) and $A^{228}\text{Th}$ was equal to the activity of ^{228}Th (dpm) on the sample planchet at the time of the beta count (t_3). Because ^{228}Th decays at a rate of $\sim 3\%$ per month, corrections were also required for any decay of ^{228}Th that occurred between the beta (t_3) and alpha (t_4) counts (generally less than 2% for our samples). The experimentally observed β cpm/Th dpm relationships described by Eqs. (1) and (2) generally agreed with the predicted ingrowth of beta activity based on Bateman equations (Friedlander et al., 1964). However, the ingrowth equations provided here are purely empirical and will depend on, among other factors, sample configuration, planchet material, and detector window thickness.

3.2.4. ^{230}Th beta count correction

Though not a beta emitter per se, ^{230}Th did produce a beta signal (β_{230} , cpm):

$$\beta_{230} \text{ (cpm)} = A^{230}\text{Th (dpm)} \times [0.08], \quad (3)$$

where $A^{230}\text{Th}$ was equal to the activity of ^{230}Th (dpm) on the sample planchet as determined by α -spectrometry. The source of these beta counts is discussed in Sections 3.1 and 3.2.5.

3.2.5. ^{232}Th beta count correction

Determination of the beta contribution from ^{232}Th was more difficult because ^{232}Th cannot be easily separated from ^{228}Th . Therefore, the beta contribution from ^{232}Th was measured by first counting total beta emissions from a mixture of ^{232}Th and ^{228}Th , followed by counting the activity of each isotope on the

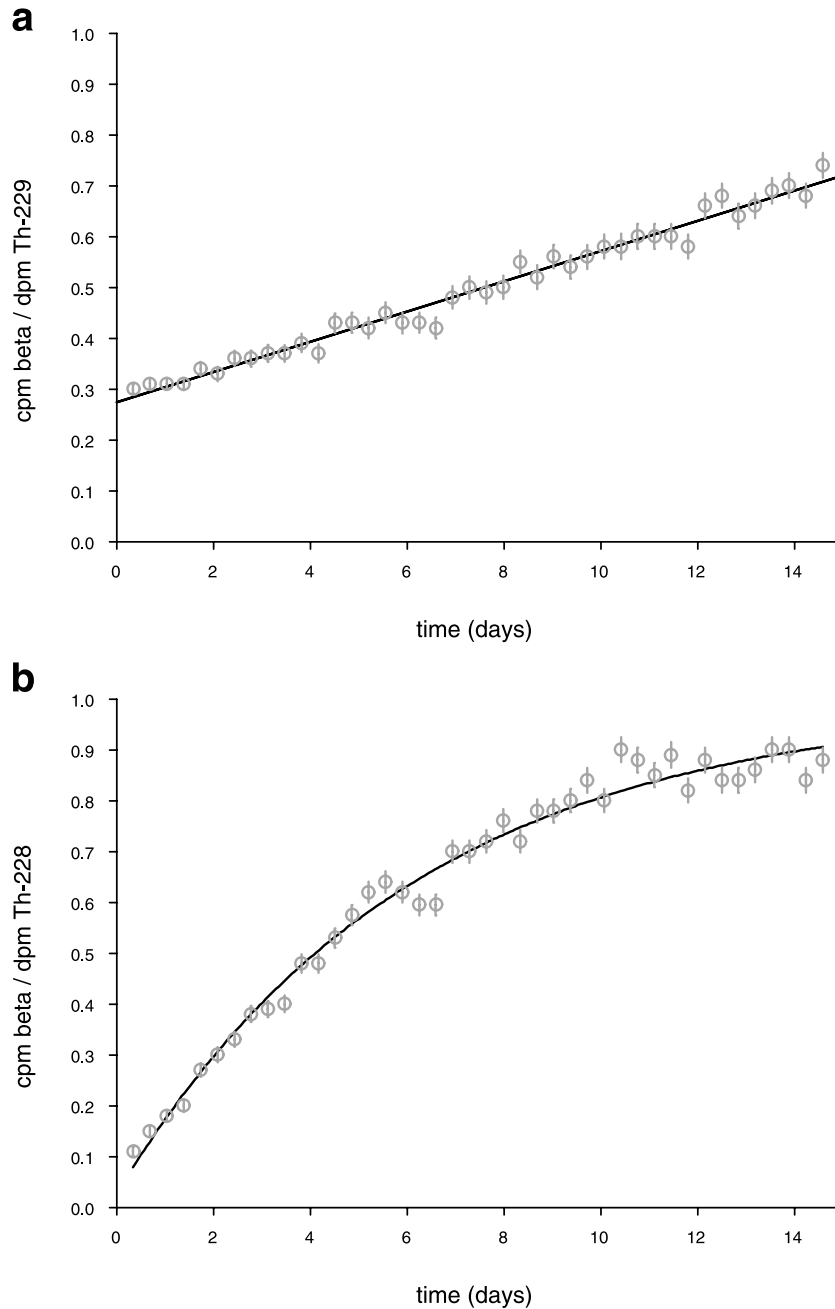


Fig. 2. Beta ingrowth from (a) ^{229}Th and (b) ^{228}Th over a period of 14.6 days after thorium separation on an anion exchange resin column. The circles show the measured beta count rate (cpm, $\pm 1s$ counting errors) as a function of thorium activity (dpm) measured later by α -spectrometry. Equations for the solid lines are given in the text as (a) Eq. (1) and (b) Eq. (2).

Table 1
Calculation of excess ^{234}Th activity on four replicate samples of homogenized Lake Michigan sediment (14–16 cm deep)

sample		^{229}Th dpm	^{228}Th dpm	^{232}Th dpm	^{230}Th dpm	^{229}Th recovery %				^{238}U dpm/g	
alpha counts	1	0.60 ± 0.01	1.53 ± 0.02	1.42 ± 0.02	1.15 ± 0.02	73 ± 2				0.80 ± 0.04	
	2	0.59 ± 0.01	1.66 ± 0.02	1.55 ± 0.02	1.34 ± 0.02	71 ± 2				1.03 ± 0.05	
	3	0.80 ± 0.02	1.87 ± 0.02	1.90 ± 0.02	1.56 ± 0.02	98 ± 2				0.94 ± 0.05	
	4	0.74 ± 0.02	1.86 ± 0.02	1.84 ± 0.02	1.50 ± 0.02	90 ± 2				0.91 ± 0.05	
sample		beta background	β^{229} ¹ cpm	β^{228} ² cpm	β^{232} ³ cpm	β^{230} ⁴ cpm	recovery ⁵ factor	decay factor	sample mass factor		
beta count correction	1	0.73 ± 0.02	0.24 ± 0.01	0.82 ± 0.02	0.07 ± 0.01	0.09 ± 0.01	1.20 ± 0.02	1.14	0.96		
	2	0.77 ± 0.02	0.24 ± 0.01	0.89 ± 0.02	0.08 ± 0.01	0.11 ± 0.01	1.22 ± 0.02	1.14	0.93		
	3	0.76 ± 0.02	0.33 ± 0.01	1.00 ± 0.02	0.09 ± 0.01	0.12 ± 0.01	0.89 ± 0.02	1.14	0.95		
	4	0.84 ± 0.02	0.30 ± 0.01	1.00 ± 0.02	0.09 ± 0.01	0.12 ± 0.01	0.97 ± 0.02	1.14	0.94		
sample	initial β count cpm	- bkgd cpm	$-\beta^{229}$ cpm	$-\beta^{228}$ cpm	$-\beta^{232}$ cpm	$-\beta^{230}$ cpm	^{234}Th at t_3 dpm/sample	^{234}Th at t_2 dpm/sample	^{234}Th dpm/g	excess ^{234}Th dpm/g	
beta count	1	2.47 ± 0.04	1.74 ± 0.04	1.49 ± 0.04	0.67 ± 0.05	0.60 ± 0.05	0.51 ± 0.05	0.61 ± 0.06	0.70 ± 0.07	0.67 ± 0.06	-0.13 ± 0.08
	2	3.07 ± 0.04	2.30 ± 0.04	2.06 ± 0.05	1.18 ± 0.05	1.10 ± 0.05	0.99 ± 0.05	1.21 ± 0.07	1.38 ± 0.08	1.29 ± 0.07	0.26 ± 0.09
	3	3.26 ± 0.04	2.49 ± 0.04	2.17 ± 0.05	1.16 ± 0.05	1.07 ± 0.05	0.94 ± 0.05	0.84 ± 0.05	0.96 ± 0.06	0.92 ± 0.05	-0.02 ± 0.07
	4	3.24 ± 0.04	2.39 ± 0.05	2.09 ± 0.05	1.09 ± 0.05	1.00 ± 0.05	0.88 ± 0.05	0.85 ± 0.05	0.97 ± 0.06	0.92 ± 0.06	0.01 ± 0.07

The initial (gross) beta count (lower left corner) is progressively corrected for beta interferences outlined in Section 3.2. Beta count corrections with superscripts 1–5 were calculated using the activities of the alpha-emitting thorium isotopes and Eqs. (1–5). Errors for each value reflect propagated counting errors ($\pm 1s$) only.

sample planchet by α -spectrometry. Beta counts attributable to ^{228}Th were then subtracted from the background corrected beta count of the ^{232}Th and ^{228}Th mixture using Eq. (2). The remaining beta counts, attributable to ^{232}Th (β_{232} , cpm), were equal to:

$$\beta_{232} \text{ (cpm)} = A^{232}\text{Th (dpm)} \times [0.05], \quad (4)$$

where $A^{232}\text{Th}$ was equal to the activity of ^{232}Th (dpm) on the sample planchet as determined by α -spectrometry.

The apparent beta signal from each of the alpha-emitting thorium isotopes (i.e., ^{229}Th and ^{228}Th (at $t_3 - t_2 = 0$), ^{230}Th , and ^{232}Th) was higher than the expected number of betas from alpha–beta cross-talk ($\sim 1.3\%$ with a ^{210}Po source). We attribute these higher beta counts to internal conversion electrons. A comparison between the number of beta counts measured per alpha decay and the number of internal conversion electron events ($E > 50$ keV) per alpha decay (International Commission on Radiological Protection, 1983) for all four thorium isotopes showed a linear relationship ($r^2 = 0.97$) and a conversion electron counting efficiency of 30%.

3.2.6. Chemical yield and detector efficiency correction

Next, corrections were made for the overall chemical yield (extraction, separation, and electrodeposition) and the efficiency of the gas-flow proportional counters. ^{234}Th activity on the planchet at the time of the beta count (^{234}Th at t_3 , dpm) was equal to:

$$A^{234}\text{Th at } t_3 \text{ (dpm)} = \beta^{234}\text{Th (cpm)} \times [1/(E \times ^{229}\text{Th yield})], \quad (5)$$

where $\beta^{234}\text{Th}$ (cpm) was equal to the remaining beta count after corrections for all positive bias in the total beta count, E was equal to the effective efficiency (cpm/dpm) of each of the gas-flow proportional counters ($E = 1.15$), and ^{229}Th yield was equal to the

ratio of ^{229}Th recovered on the sample planchet (as determined by α -spectrometry and the known efficiency of the α -detector itself) to that added to the initial sample matrix.

3.2.7. Decay ($t_3 - t_2$) correction

The next step was to correct for the decay of ^{234}Th on the planchet between the time of the beta count (t_3) and the time of column separation (t_2).

3.2.8. Sample size correction

The activity of ^{234}Th on the sample planchet was then normalized to the initial sample size to give ^{234}Th activity in dpm l^{-1} of water or dpm g^{-1} of particles at the time of column separation.

3.2.9. ^{238}U support correction

Next, a correction was made for ^{234}Th activity supported by uranium. For dissolved ^{234}Th samples, a small correction had to be made for uranium carried with the ferric hydroxide precipitate (see Section 3.1). While it was found that corrections for ^{234}Th supported by the decay of particle-bound ^{238}U were generally insignificant, supported ^{234}Th had to be taken into account for sediment samples with very low ^{234}Th activity. These corrections are made directly from an independent measurement of uranium in the sample as described by Edgington et al. (1996).

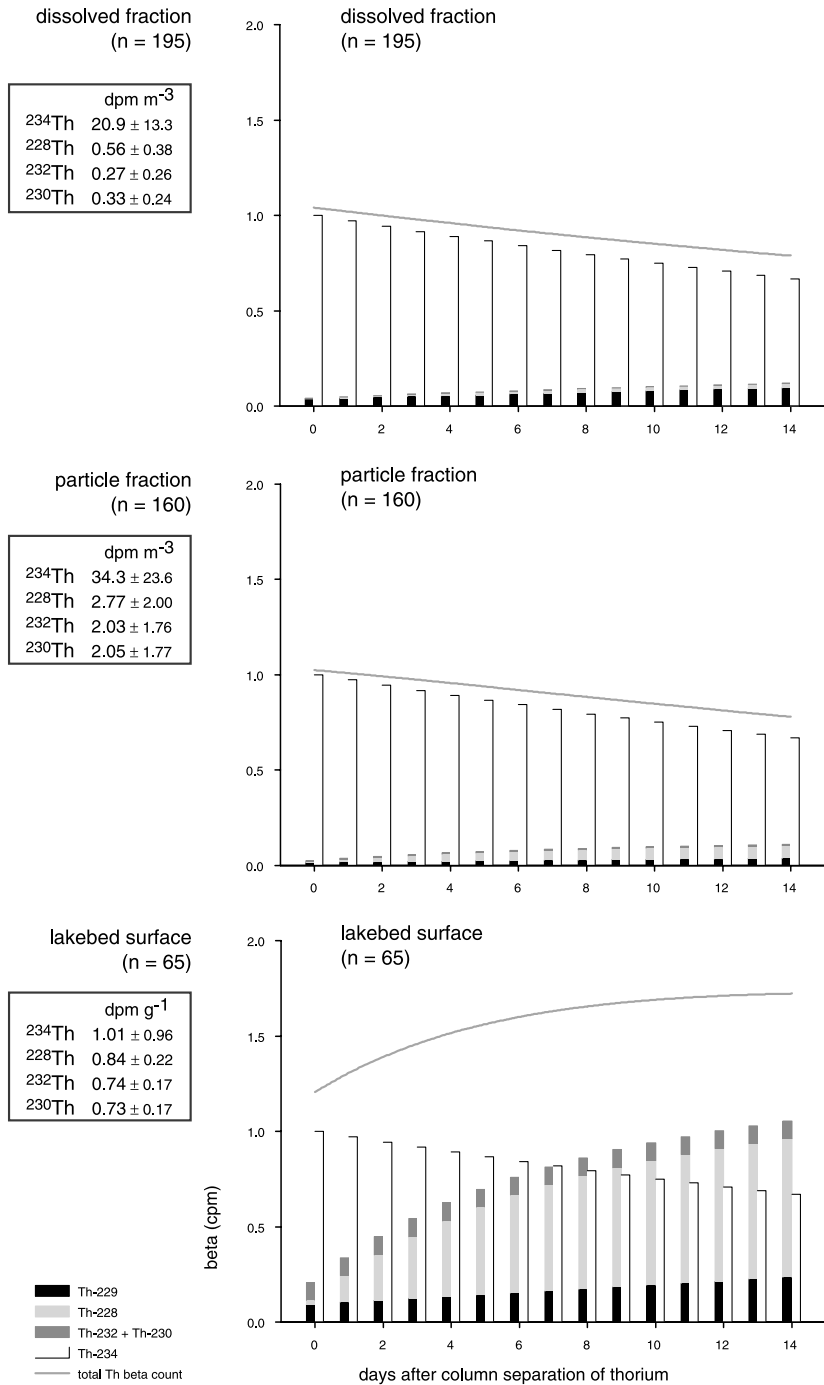
3.2.10. Decay ($t_2 - t_1$) correction

A final correction was made for the decay of unsupported (i.e., excess) ^{234}Th from the time of column separation (t_2) back to the time of sample collection and particle separation (t_{1a}). In instances where the dissolved ^{234}Th fraction could not be precipitated immediately, a straightforward correction was applied for the ingrowth of supported ^{234}Th (based on the dissolved ^{238}U activity of the sample) during the time elapsed between the initial particle separation at t_{1a} and the ferric hydroxide separation at t_{1b} .

Fig. 3. The contribution of beta counts from all thorium isotopes in Lake Michigan samples from the dissolved water fraction (top), the particle-bound water fraction (middle), and lakebed surface (bottom, 0–0.5 cm) sediment. Activities (mean \pm 1s) of the naturally occurring thorium isotopes in each of the three sample types are shown on the left. Subsequent beta count rates for all thorium isotopes (including the isotopic diluent ^{229}Th) normalized to 1 $\text{cpm } ^{234}\text{Th}$ at t_2 over a period of 14 days after thorium purification at t_2 are shown on the right.

in situ thorium activities

thorium beta counts over time



3.3. ^{234}Th blank analysis

Combusted blank nitrocellulose filters (0.45 μm , 293 mm, Millipore) yielded a negligible residue, both in terms of weight and isotopic contamination.

A blank analysis of the entire dissolved fraction procedure (i.e., Sections 2.1–2.3) was run on 46.5 l of deionized water that included the ferric hydroxide precipitation reaction with 55 mg Fe^{3+} , 500 ml HCl (12 M), and 500 ml NH_4OH (14 M) and 0.24 dpm of ^{229}Th as a yield monitor. Total sample beta activity, corrected for instrument background and ^{229}Th derived beta activity (see Section 3.2.2), was equal to 0.086 ± 0.064 dpm.

3.4. Verification of method and uncertainties

To test the accuracy of our approach to measuring ^{234}Th , we measured samples with ^{234}Th activities in secular equilibrium with ^{238}U . Because there is little chance of collecting a naturally occurring freshwater sample where ^{234}Th activities may be assumed to be in secular equilibrium with ^{238}U , we analyzed four ~ 1 g samples of sediment from a single sample of homogenized (14–16 cm deep) Lake Michigan sediment collected in 1991. In these samples, ^{234}Th bound to the sediment must be in secular equilibrium with its parent, ^{238}U . By measuring both nuclides, any biases introduced by this approach should show as a net difference in the measured activities of the parent and daughter.

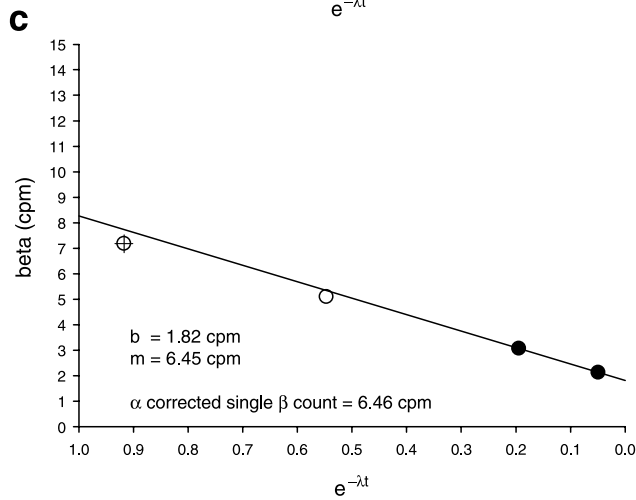
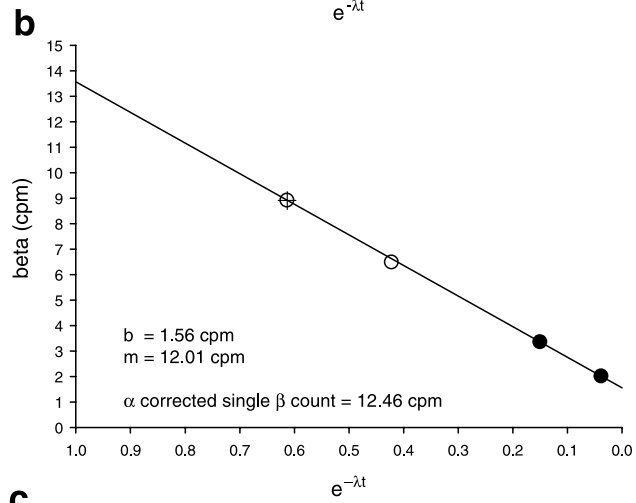
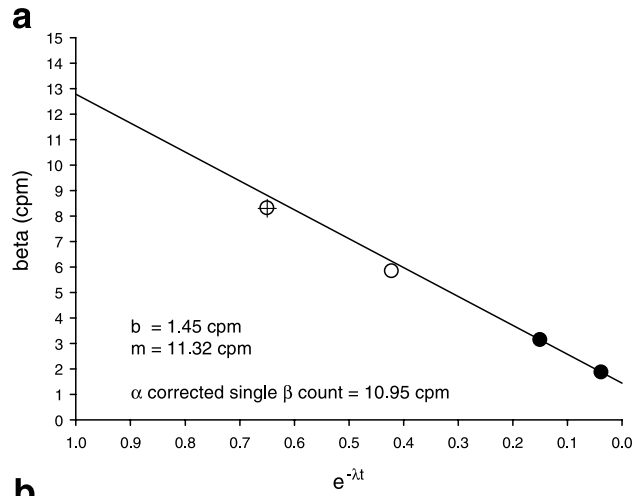
After adding 0.8 dpm ^{229}Th (NIST SRM-4328b) to each of the sediment samples, the samples were analyzed according to the methods described above and counted for gross beta activity (for four consecutive 500-min counts) approximately 4.5 days after column separation (t_2). The results obtained in this comparison are shown in Table 1.

The top section of the table details the results obtained for all of the α -active isotopes measured in

each sample, including the isotopic diluent (^{229}Th) and ^{238}U . The reported activities of the thorium isotopes in dpm reflect activities of each isotope on the sample planchet based on the known efficiencies of the alpha detectors. The total recovery of thorium based on the isotopic diluent varied between 71% and 98%. Variability in the uranium activity of the sediment samples was primarily due to variability in the HCl leaching process—which is difficult to control and not a problem as far as the recovery of surface-bound unsupported (excess) ^{234}Th is concerned.

The second (middle) section of the table details the corrections for the background counts from the beta counters (first column) and the contributions from all isotopes other than ^{234}Th that will be subtracted from the initial total beta counts (first column in the bottom section of the table). The correction for detector background accounted for an average of 26% of the original beta counts, and subsequent corrections for beta counts generated by ^{229}Th , ^{228}Th , ^{232}Th , and ^{230}Th further reduced the initial total beta count by an average of 9%, 31%, 4%, and 3%, respectively. These corrections are subtracted cumulatively going from left to right in the bottom section of the table. Corrections for sample yield based on total ^{229}Th recovery, gas-flow proportional counter efficiency, ^{234}Th decay between beta counting and column separation, and normalizing for sample size resulted eventually in a mean ^{234}Th activity of 0.95 ± 0.26 dpm g^{-1} ($1s$, $n=4$) (penultimate column). Direct measurement of ^{238}U removed during the initial leaching of the sediment sample yielded an average activity of 0.92 ± 0.09 dpm g^{-1} ($1s$, $n=4$). Subtracting the calculated ^{234}Th activities from the measured ^{238}U activities in each sample resulted in a mean excess ^{234}Th activity (final column) of 0.03 ± 0.16 dpm g^{-1} ($1s$, $n=4$). The one-sigma ($1s$) error based on replicate sample analyses was two times higher than

Fig. 4. ^{234}Th decay curves. In panels (a–c), total beta count rates (circles) from three Lake Michigan water column samples are plotted against $e^{-\lambda t}$, where λ equals the ^{234}Th decay constant ($\lambda = 0.02876 \text{ day}^{-1}$) and t equals the time in days since thorium separation on an anion exchange resin column. The slope (m) and y -intercept (b), obtained from two-point regressions (solid line) on the last two counts of each sample (solid circles), give the initial activity of ^{234}Th (in counts per minute) at column separation and activity generated by background and decay products of the other thorium isotopes (i.e. ^{228}Th , ^{229}Th , ^{230}Th , and ^{232}Th), respectively. The slight deviation of the first two counts (open circles) from the regression line is caused by the incomplete ingrowth of beta activity from ^{228}Th (and ^{229}Th) decay chains. The first count in each of the three samples (cross hair) was used to make a single time-point or “ α -corrected single β count” calculation of ^{234}Th activity at the time of column separation using the method we describe here.



the average 1 σ error based solely on propagated counting statistics because sampling errors and other processing uncertainties are not factored into simple propagated counting statistics (Buesseler et al., 2001). From these results, we conservatively estimate the detection limit of ^{234}Th for this method at 1.5 dpm/total sample (American Public Health Association, 1998; National Council on Radiation Protection and Measurements, 1985).

3.5. Comparison of computational methods

3.5.1. Immediate sample count method

The immediate sample count method relies on measuring the activity of ^{234}Th before significant ingrowth of extraneous beta activity occurs. While the rate of extraneous ingrowth of beta activity depends ultimately on the activities of ^{229}Th , ^{228}Th , ^{230}Th , and ^{232}Th deposited on the sample planchet, it also depends largely on the original isotopic composition of the sample. To determine the error associated with ignoring this fraction of the total beta count, we compare the results of ^{234}Th measurements obtained using our approach to those of the immediate sample count method for samples from Lake Michigan. These samples were collected over a period of 5 months in 2000, in water depths ranging between 5 and 40 m, and include (1) the dissolved ($<0.45\mu\text{m}$) fraction in the water ($n=195$); (2) the suspended particle fraction in the water ($n=160$); and (3) lakebed surface (0–0.5 cm) sediment ($n=65$) (Fig. 3, in situ thorium activities). For ready comparison, all thorium isotope activities were normalized to 1 cpm of ^{234}Th at the time of column separation (t_2). Estimates of beta generation from the normalized α -activities of ^{229}Th , ^{228}Th , ^{232}Th , and ^{230}Th were then calculated for a period of 14 days after column separation using Eqs. (1–5) (Fig. 3, thorium beta counts over time). At 1 day after column separation, extraneous and apparent (non- $^{234}\text{Th} + ^{234\text{m}}\text{Pa}$) beta counts accounted for an average of $\sim 4\%$ of the total beta count from the dissolved and particle-bound water fractions (Fig. 3, top and middle). This percentage rose to an average of 27% in surface sediment samples, (Fig. 3, bottom). Moreover, the error was compounded when correcting for ^{234}Th decay between column separation (t_2) and the beta count itself (t_3). Ignoring the non- $^{234}\text{Th} + ^{234\text{m}}\text{Pa}$ beta signal in water column samples led

to an average ^{234}Th overestimate of $\sim 5\%$ if the counts were completed 1 day after column separation. This increased to a 38% error for surface sediment samples.

3.5.2. Multiple beta count method

To compare our approach to measuring ^{234}Th with the multiple beta count method, we counted the beta activity of three suspended particle samples from Lake Michigan four times over a 114-day period after column separation of thorium (Fig. 4). For this method, estimates of the beta count rate (cpm) at column separation (t_2) due specifically to ^{234}Th were

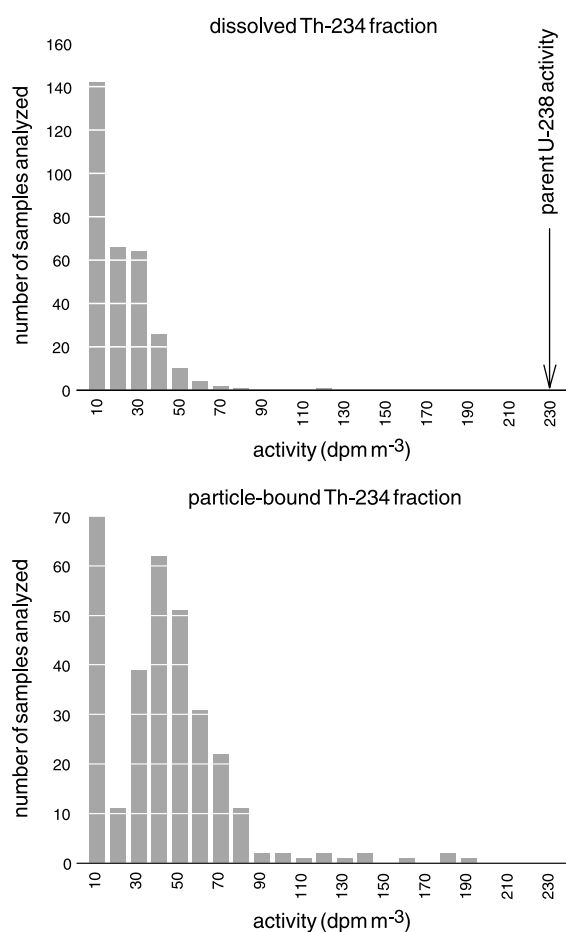


Fig. 5. Frequency distribution of Lake Michigan ^{234}Th activities in the dissolved fraction (top, $n=316$) and the particle-bound fraction (bottom, $n=311$) of water samples collected between 1998 and 2000 in water column depths ranging from 5 to 150 m.

calculated using the slope of a regression of the last two counts (solid circles). Estimates of the ^{234}Th beta count rate at column separation using our approach were calculated from the first count of each sample (crosshatched circles). In spite of the fact that our initial counts were made 15, 17, and 3 days after column separation (Fig. 4, panels a–c, respectively), agreement between the two counting methods was excellent. The mean difference in calculated initial activity (at t_2) for all three samples was 0.03 cpm with a standard error of the residual mean equal to ± 0.24 cpm. These results are important because they demonstrate that we are able to effectively account for the ingrowth of non- $^{234}\text{Th} + ^{234\text{m}}\text{Pa}$ beta activity over a period of at least 2 weeks after column separation of thorium. Moreover, because the initial beta count can be made immediately after thorium is purified, our method reduces by up to one half the minimum initial sample activity required by the method relying on multiple beta counts.

3.6. ^{234}Th activities in water samples collected from Lake Michigan

This technique has been used to measure ^{234}Th in over 300 water samples from Lake Michigan.

These samples were collected at different seasons between 1998 and 2000 from surface waters (~ 2 – 20 m) at stations in shallow nearshore (5 m) to deepwater (150 m) environments. The results of these analyses are presented in Fig. 5, which displays the distribution of activities measured on the dissolved and particle-bound fractions. From these data, it is readily seen that the amount of ^{234}Th in either the dissolved or particle-bound fraction is a very small fraction of the expected activity in secular equilibrium with ^{238}U . The preponderance of ^{234}Th activity in the 0 – 10 dpm m^{-3} range in the dissolved fraction reflects the strong affinity of thorium for particles as well as the generally high suspended particle load in these relatively shallow systems. A similar preponderance in the frequency of low ^{234}Th activities on particles reflects the rapid flux of particles from these surface water environments.

3.7. Method application to low salinity and fresh-water systems

The sample size of water or sediment needed to obtain a measurement of ^{234}Th depends ultimately on the source strength or activity of the parent, ^{238}U . In

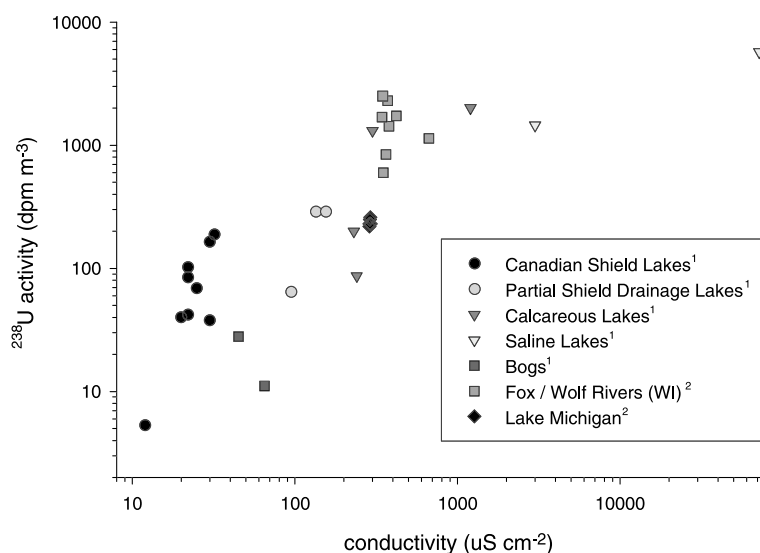


Fig. 6. Dissolved ^{238}U activity in selected North American lakes, bogs, and rivers as a function of conductivity; ¹Wahlgren and Orlandini (1982); ²this study.

marine systems, the activity of dissolved ^{238}U correlates strongly with salinity (Chen et al., 1986) and knowledge of the latter is often sufficient for estimating potential ^{234}Th activity. In freshwater systems, however, Wahlgren and Orlandini (1982) and the work we have done here show only a weak correlation between dissolved ^{238}U and conductivity in North American lakes, rivers, and bogs (Fig. 6). Regional variations in the uranium content of weatherable minerals as well as large variations in carbonate alkalinity and dissolved organic carbon (which affect the distribution coefficient of uranium between dissolved and particle-bound states) preclude an exact correlation. Nevertheless, the correlation is strong enough to show that disequilibria between ^{234}Th and ^{238}U can be measured in all but the most pure freshwater systems.

4. Conclusions

Measuring $^{234}\text{Th}/^{238}\text{U}$ disequilibria in aquatic systems can provide answers to a host of rate-related questions regarding the behavior of particles and particle-reactive matter. Increasing the accuracy and lowering the detection limit of these measurements decreases both the uncertainties of our results and the geographical and temporal limits on where and when this isotope chronometer can be used. In this study, we presented a new approach to measuring ^{234}Th in low salinity and a freshwater systems that improves upon currently used beta counting methodologies in significant ways. By correcting for apparent beta counts (i.e., conversion electrons), our method eliminates the bias inherent in the method relying on immediate sample counting. By correcting for extraneous betas produced by the progeny of other thorium isotopes in a sample (e.g., the ingrowth of ^{212}Bi from ^{228}Th), our method eliminates the bias inherent in the method relying on weak beta absorbers, which, for ^{212}Bi at least, cannot be differentiated from or selectively absorbed in the presence of $^{234\text{m}}\text{Pa}$. Finally, because we count the sample for betas only once—and can do so before the complete ingrowth of ^{228}Th progeny—our method reduces by up to one half the minimum initial sample activity required by the method relying on multiple beta counts.

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References

- Aller, R.C., Cochran, J.K., 1976. $^{234}\text{Th}/^{238}\text{U}$ disequilibrium in near-shore sediment: particle reworking and diagenetic time scales. *Earth Planet. Sci. Lett.* 29, 37–50.
- American Public Health Association, 1998. *Standard Methods for the Examination of Water and Wastewater*, 20th ed. American Public Health Association, Washington, DC.
- Bacon, M.P., Anderson, R.F., 1982. Distribution of thorium isotopes between dissolved and particulate forms in the deep sea. *J. Geophys. Res.* 87, 2045–2056.
- Baltakmens, T., 1977. Accuracy of absorption methods in the identification of beta emitters. *Nucl. Instrum. Methods* 142, 535–538.
- Benitez-Nelson, C.R., Buesseler, K.O., Rutgers van der Loeff, M., Andrews, J., Ball, L., Crossin, G., Charette, M.A., 2001. Testing a new small-volume technique for determining ^{234}Th in seawater. *J. Radioanal. Nucl. Chem.* 248, 795–799.
- Broecker, W.S., Peng, T.H., 1982. *Tracers in the Sea*. Eldigio Press, Columbia University, 690 pp.
- Buesseler, K.O., Cochran, J.K., Bacon, M.P., Livingston, H.D., Casso, S.A., Hirschberg, D., Hartman, M.C., Fler, A.P., 1992. Determination of thorium isotopes in seawater by non-destructive and radiochemical procedures. *Deep-Sea Res.* 39, 1103–1114.
- Buesseler, K.O., Michaels, A.F., Siegel, D.A., Knap, A.H., 1994. A three dimensional time-dependent approach to calibrating sediment trap fluxes. *Glob. Biogeochem. Cycles* 8, 179–193.
- Buesseler, K.O., Benitez-Nelson, C.R., Rutgers van der Loeff, M., Andrews, J., Ball, L., Crossin, G., Charette, M.A., 2001. An intercomparison of small- and large-volume techniques for ^{234}Th in seawater. *Mar. Chem.* 74, 15–28.
- Carpenter, R., Peterson, M.L., Bennett, J.T., Somayajulu, B.L.K., 1984. Mixing and cycling of uranium, thorium and ^{210}Pb in Puget sound sediments. *Geochim. Cosmochim. Acta* 48, 1949–1963.
- Chen, J.H., Edwards, R.L., Wasserburg, G.J., 1986. ^{238}U , ^{234}U and ^{232}Th in seawater. *Earth Planet. Sci. Lett.* 80, 241–251.
- Coale, K.H., Bruland, K.W., 1985. $^{234}\text{Th}/^{238}\text{U}$ disequilibria within the California Current. *Limnol. Oceanogr.* 30, 22–33.
- Edgington, D.N., Robbins, J.A., Colman, S.C., Orlandini, K.A., Gustin, M.P., 1996. Uranium-series disequilibrium, sedimentation, diatom frustules, and paleoclimate change in Lake Baikal. *Earth Planet. Sci. Lett.* 142, 29–42.

- Faris, J.P., Buchanan, R.F., 1964. Anion exchange characteristics of elements in nitric acid medium. *Anal. Chem.* 36, 1157–1158.
- Feng, H., Cochran, J.K., Hirschberg, D.J., 1999. ^{234}Th and ^7Be as tracers for the transport and dynamics of suspended particles in a partially mixed estuary. *Geochim. Cosmochim. Acta* 63, 2487–2505.
- Friedlander, G., Kennedy, J.W., Miller, J.M., 1964. *Nuclear and Radiochemistry*, 2nd ed. Wiley, New York, NY.
- Gustafsson, O., Gschwend, P.M., Buesseler, K.O., 1997. Settling removal rates of PCBs into the Northwest Atlantic derived from ^{238}U – ^{234}Th disequilibria. *Environ. Sci. Technol.* 31, 3544–3550.
- International Commission on Radiological Protection, 1983. *Radionuclide Transformations: Energy and Intensity of Emissions*. ICRP Pub., vol. 38. Pergamon, Oxford.
- Kaplan, D.I., Bertsch, P.M., Adriano, D.C., Orlandini, K.A., 1994. Actinide association with ground water colloids in a coastal plain aquifer. *Radiochim. Acta* 66–67, 181–187.
- Kershaw, P., Young, A., 1988. Scavenging of ^{234}Th in the Eastern Irish Sea. *J. Environ. Radioact.* 6, 1–23.
- McKee, B.A., DeMaster, D.J., Nittrouer, C.N., 1984. The use of $^{234}\text{Th}/^{238}\text{U}$ disequilibrium to examine the fate particle-reactive species on the Yangtze continental shelf. *Earth Planet. Sci. Lett.* 68, 431–442.
- National Council on Radiation Protection and Measurements, 1985. *A Handbook of Radioactivity Measurements Procedures*. NCRP Rep. No. 58. Bethesda, MD.
- Puphal, K.W., Olsen, D.R., 1972. Electrodeposition of alpha-emitting nuclides from a mixed oxalate–chloride electrolyte. *Anal. Chem.* 44, 284–289.
- Rutgers van der Loeff, M.M., Moore, W.S., 1999. Determination of natural radioactive tracers. In: Grasshoff, K., Kremling, K., Ehrhardt, M. (Eds.), *Methods of Seawater Analysis*. Wiley–VCH, Weinheim, pp. 365–397. Chapter 13.
- Wahlgren, M.A., Orlandini, K.A., 1982. Comparison of the geochemical behavior of plutonium, thorium and uranium in selected North American lakes. *Environmental Migration of Long-lived Radionuclides*. IAEA, Vienna, pp. 757–774.