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Technical note: Comparison of radiometric techniques for estimating recent organic carbon sequestration rates in inland wetland soils

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Abstract. For wetlands to serve as natural climate solutions, accurate estimates of organic carbon (OC) sequestration rates in wetland sediments are needed. Dating using cesium-137 (^{137}Cs) and lead-210 (^{210}Pb) radioisotopes is commonly used for measuring OC sequestration rates in wetland sediments. ^{137}Cs radioisotope dating is relatively simple, with calculations based on a single point representing the onset (1954) or peak (1963) of the ^{137}Cs fallout. ^{210}Pb radioisotope dating is more complex, as the calculations are based on multiple points. Here, we show that reliable dating of sediment cores collected from wetlands can be achieved using either ^{137}Cs or ^{210}Pb dating or their combination. However, ^{137}Cs and ^{210}Pb profiles along the depth of sediment cores need to be screened, analyzed, and interpreted carefully to estimate OC sequestration rates with high precision. To this end, we propose a decision framework for screening ^{137}Cs and ^{210}Pb profiles into high- and low-quality sediment profiles, and we compare dating using the 1954 and 1963 time markers, i.e., the rates of sedimentation and, consequently, OC sequestration over the past ~60 years. Our findings suggest that ^{137}Cs- and ^{210}Pb-based OC sequestration rates are comparable, especially when using the 1963 (vs. 1954) time marker.

1 Introduction

Wetlands in agricultural landscapes serve a crucial role in providing habitat for wildlife, regulating climate, improving water quality, and reducing floods. Moreover, these wetlands have the potential to sequester organic carbon (OC) (Bridgham et al., 2006; Nahlik and Fennessy, 2016; Bansal et al., 2023). Accounting for the balance between the sequestration and emission of carbon can help establish wetlands as essential candidates for natural climate solutions by offsetting carbon emissions (Hambäck et al., 2023). These wetlands embedded in agricultural landscapes are recognized as temperate inland wetland soils. The global carbon stock of temperate inland wetland soils is estimated to be 46 Pg C to 2 m depth, and Canada's temperate inland wetland soils are estimated to contain 4.6 Pg C (Bridgham et al., 2006). Compared to peatlands, the rapid rate of OC sequestration and the more considerable spatial extent of temperate inland wetland soils can help contribute significantly to regional or national carbon sequestration (Bridgham et al., 2006; Nahlik and Fennessy, 2016).

Canada encompasses around 25 % of the world's wetlands, with an area of approximately 1.29 million km^2, which accounts for 13 % of the country's terrestrial area (Environment and Climate Change Canada, 2016), highlighting the global importance of these wetlands. Unfortunately, there are minimal data on the OC sequestration rates in these wetlands. To estimate the OC sequestration potential of these

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Table 1. Characteristics of ^{137}Cs and unsupported ^{210}Pb (^{210}Pb$_{\text{ex}}$) dating to estimate sedimentation rates in wetlands.

Method of radiometric dating	^{137}Cs	^{210}Pb$_{\text{ex}}$
Type of radioisotope	Artificial (atmospheric deposition 1954–1963)	Natural
Half-life	30.17 years	22.3 years
Time marker	1954 (onset) and 1963 (peak)	Recent (10–20 years) to a maximum of 50–150 years
Radiometric technique	Gamma spectrometry (non-destructive)	Gamma (non-destructive) and alpha (destructive) spectrometry
Pre-processing	Drying, weighing, disaggregating, homogenizing, and sieving	For gamma analysis, drying, weighing, disaggregating, homogenizing, and sieving prior to analysis on a gamma counter. For alpha analysis, leaching with hydrochloric and nitric acid and electroplating of ^{210}Po, which constitutes allowing the digested and therefore extracted ^{210}Po isotope solution to settle on silver coins overnight before measuring the ^{210}Po (known tracer) and ^{210}Po activity (sample) the next morning through the alpha counter/ensemble.
Sample size	Minimum 1 g (larger sample size has higher certainty)	1 to 5 g for gamma spectrometry, 0.2 to 0.5 g for alpha spectrometry
Time requirement for radiometric dating	48 h for each sample for gamma spectrometry.	48 h for each sample for gamma spectrometry. 48 to 72 h for multiple samples plus sample preparation time for each sample for alpha spectrometry.
Output	A single average sedimentation rate	Variable sedimentation rate
Estimation approach	Onset of ^{137}Cs activity represents 1954, and highest peak of ^{137}Cs activity represents 1963, observed at 661.6 keV.	Activity of ^{210}Pb is observed at 46.5 keV. Excess ^{210}Pb is used to determine the vertical accretion.
Complexity in estimation	Simple; estimated by using time marker of onset or peak ^{137}Cs activity and associated sediment accumulation.	More complex; estimated by one of several models to estimate sedimentation rate. Most common models are (1) constant flux–constant sedimentation model, (2) constant rate of supply model, and (3) constant initial concentration model (Appleby and Oldfield, 1978).

wetlands, it is essential to establish precise measurements to quantify wetland OC sequestration, develop strategies to promote conservation and restoration efforts, incorporate carbon credits in the carbon markets, and validate the wetland-based ecosystem services.

There are several ways to estimate the potential of wetlands to store OC (Bansal et al., 2023). One of these methods is radiometric dating, which can calculate the OC storage rates of wetlands over periods of 10 to ≥ 1000 years. Frequently used radioisotopes for radiometric dating are cesium-137 (^{137}Cs) and lead-210 (^{210}Pb), which can be used to estimate relatively recent (up to the last 100 years) OC sequestration rates (Villa and Bernal, 2018). Estimating OC sequestration rates involves building an age–depth profile or model of ^{137}Cs and ^{210}Pb from sediment cores that demonstrates the relationship between the depth of sediment layers and their corresponding age. Since the inorganic radioisotopes (^{137}Cs and ^{210}Pb) strongly bind with the soil particles once in contact, the radioisotopes can act as an efficient tracer for investigating OC sequestration rates (Ritchie and McHenry, 1990; Craft and Casey, 2000). These characteristics allow the accurate tracking of carbon movement within ecosystems, thereby enabling the extraction of detailed information about carbon sequestration dynamics in wetlands.

The characteristics of ^{137}Cs and ^{210}Pb to estimate wetland OC sequestration rates are presented in Table 1. ^{137}Cs is an artificial radioisotope that was produced during thermonuclear bomb testing in the 1950s and 1960s, with the onset of atmospheric deposition in 1954 and a peak in 1963 (Ritchie and McHenry, 1990). The testing caused radioactive uranium to decay, and, as a result, the ^{137}Cs isotope was released into the atmosphere and was then deposited around the globe.

Although there may be challenges in applying our study to some parts of the world, the information is generally applicable and valuable for consideration in all regions. We encourage others to customize this approach further for use in other regions where ^{137}Cs deposition histories vary.

^{137}Cs has a half-life of 30.17 years, which can be used to estimate the last ~ 50–70 years of OC sequestration rates in wetlands (e.g., Bernal and Mitsch, 2012). ^{137}Cs dating assumes constant sedimentation rates measured since 1954 or 1963. In using the two time markers for ^{137}Cs, we do not expect the sedimentation rates to be equal, but we do expect them to be similar. The onset and the peak of ^{137}Cs activity at 661.6 keV can be used to mark 1954 and 1963, respectively. These time markers (1954 and 1963) can date sediment layers (Pennington et al., 1973; Ritchie and McHenry, 1990; DeLaune et al., 2003) and consequently the OC sequestration rates. ^{137}Cs has an additional time marker for Europe in 1986 due to the Chernobyl nuclear accident and for Japan in 2011 due to the Fukushima Daiichi nuclear accident (Foucher et al., 2021), indicating that OC sequestration estimates can be derived for different timescales. In the Americas, we do not see evidence of the 1986 or 2011 ^{137}Cs peaks, which are observed in Europe and Japan, respectively, so we did not need to use other radioisotope techniques (e.g., $^{239+240}$Pu) to distinguish the 1986 or 2011 ^{137}Cs peak from the 1963 ^{137}Cs peak. ^{137}Cs dating requires a gamma spectrometer to estimate OC sequestration rates. Sample preparation for gamma analysis involves drying, weighing, disaggregating, homogenizing, and sieving (Bansal et al., 2023). Samples vary from 1 to 1500 g, with smaller samples associated with higher uncertainties and therefore requiring longer times to analyze. Gamma analysis counting times range from 4 to 48 h for each sample (e.g., 4 to 12 h in Li et al., 2007; 12 to 24 h in Zarrinabadi et al., 2023; and 24 to 48 h in Kamula et al., 2017). ^{137}Cs dating provides a simple result (an average sedimentation rate), while ^{210}Pb dating provides a more complex result (using a supply rate model to reveal trends in sedimentation rates). Plutonium (Pu) may replace ^{137}Cs in the future due to concerns of half-life and persistence as a dating tool. In essence, $^{239+240}$Pu has the same source and deposition mechanism as ^{137}Cs. Its longer half-life will make its peak measurable when ^{137}Cs is no longer measurable.

Unlike ^{137}Cs, ^{210}Pb is a naturally occurring radionuclide derived from uranium-238 (^{238}U) and deposits atmospherically from the decay of radium-226 (^{226}Ra) (Walling and He, 1999). ^{210}Pb has a half-life of 22.3 years and is used to estimate the last 10–150 years of OC sequestration rates in wetlands (Craft and Richardson, 1998; Craft and Casey, 2000; Craft et al., 2018; Creed et al., 2022). ^{210}Pb activity can be measured using gamma (observed at 46.5 keV) and alpha (destructive) spectrometry (Walling and He, 1999; Bellucci et al., 2007). Traditional alpha analysis requires 0.2–0.5 g of sample and additional sample preparation involving leaching with hydrochloric and nitric acid and electroplating (up to 24 h for sample preparation) (Bansal et al., 2023). Alpha analysis can be considered an indirect method for ^{210}Pb dating where polonium-210 (^{210}Po) activity is measured, assuming both ^{210}Pb and ^{210}Po are in a secular equilibrium. ^{210}Pb activity is calculated by comparing ^{210}Po activity against the known activity of ^{209}Po (isotope tracer). In alpha analysis, the additional time required for sample preparation is compensated by running multiple samples simultaneously (Bansal et al., 2023). Gamma and alpha spectrometry of ^{210}Pb provides the total ^{210}Pb activity, which incorporates unsupported (or excess) ^{210}Pb (^{210}Pb$_{\mathrm{ex}}$) and supported ^{210}Pb. ^{210}Pb$_{\mathrm{ex}}$ is used to determine the mass or sediment accumulation rate. Supported ^{210}Pb is derived from the natural decay of radium-226 (^{226}Ra) in the sediment, while unsupported ^{210}Pb comes from the decay of atmospheric radon-222 (^{222}Rn), which deposits ^{210}Pb onto the sediment surface from the air. Unsupported ^{210}Pb activity decreases over time due to radioactive decay, unlike supported ^{210}Pb (Appleby and Oldfield, 1983). The choice of model used in ^{210}Pb dating can reflect constant and variable sedimentation rates (Sanchez-Cabeza and Ruiz-Fernandez, 2012) and, consequently, OC sequestration rates in wetlands. Some models used for ^{210}Pb dating are (1) the constant flux–constant sedimentation (CFCS) model, (2) the constant rate of supply (CRS) model, and (3) the constant initial concentration (CIC) model (Appleby and Oldfield, 1978). Both ^{137}Cs and ^{210}Pb provide suitable time markers and a longer time horizon compared to direct measurements using the time marker of horizons (2–10 years) to study sediment accretion and, subsequently, OC sequestration rates in wetlands (Bernal and Mitsch, 2013; Villa and Bernal, 2018). In this study, we compared the average OC sequestration rate derived from ^{137}Cs time markers with the progressive OC sequestration rates derived using a constant rate of supply model applied to ^{210}Pb.

The combined use of ^{137}Cs and ^{210}Pb may improve the accuracy of the dating estimation (Drexler et al., 2018; Creed et al., 2022). The more detailed assessment accrues a higher cost and time requirement, and the need for specialized equipment and technical expertise to conduct laboratory and data analyses may constrain the research efforts (Bansal et al., 2023). Furthermore, factors such as timescales, analytical complexity in interpreting radioisotope profiles (e.g., ^{137}Cs peak clarity), variability in atmospheric deposition, and mobilization of radioisotopes can contribute to uncertainty (Drexler et al., 2018; Loder and Finkelstein, 2020; Zhang et al., 2021; Bansal et al., 2023) and limit the applicability of one radioisotope over the other. Therefore, it is essential to consider the advantages and potential challenges of using radioisotopes before designing research studies.

The main objective of this research paper is to explore the use of ^{137}Cs and ^{210}Pb to estimate recent OC sequestration rates in undisturbed (i.e., not directly impaired by human activities) temperate inland wetland soils located on agricultural landscapes. Here, we aim to (1) categorize ^{137}Cs or ^{210}Pb profiles into high- and low-quality via a decision

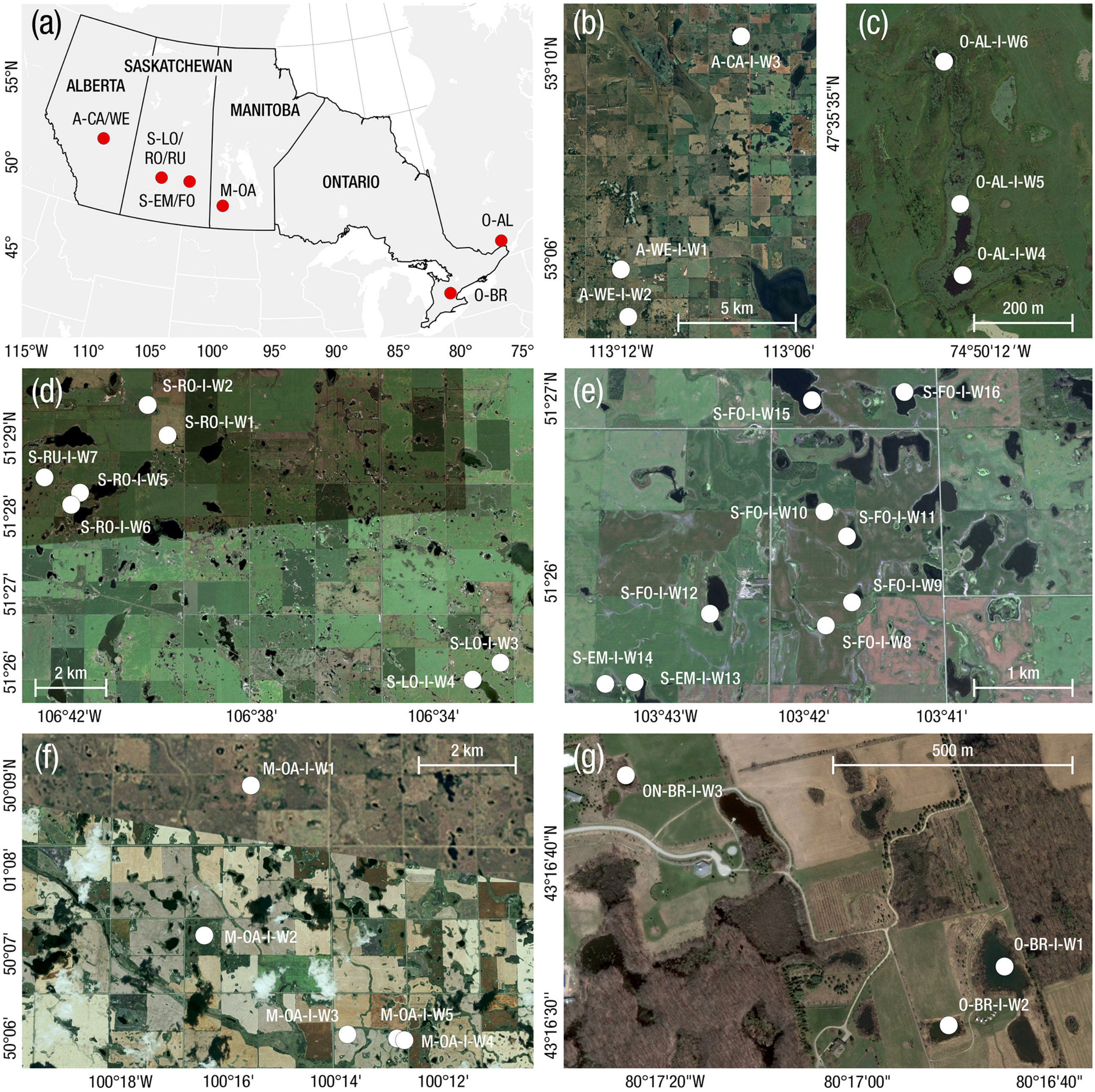

Figure 1. (a) Study area situated in four provinces of Canada, **(b)** three wetland sites in Alberta (AB), **(c)** three wetland sites in Ontario (ON), **(d)** seven wetland sites in Saskatchewan (SK), **(e)** nine wetland sites in Saskatchewan (SK), **(f)** five wetland sites in Manitoba (MB), and **(g)** three wetland sites in Ontario (ON). Panels **(b–g)** are based on the sampling locations of wetlands used in this study reproduced using Google Earth Images: **(b)** and **(c)** ©2024 Airbus; **(d)**, **(e)**, and **(g)** ©2024 Maxar Technologies; **(f)** ©2024 Airbus and Maxar Technologies.

framework, (2) apply the decision framework to estimate OC sequestration rates, (3) use 1963 and 1954 time markers to compare the ^{137}Cs- and ^{210}Pb-based OC sequestration rates to get a better understanding of the sediment history, and (4) select the best approach for ^{137}Cs and ^{210}Pb to estimate the OC sequestration rates with highest precision. This study helps reduce uncertainty in studies that rely on ^{137}Cs or ^{210}Pb radioisotope dating.

2 Methods

2.1 Sediment core collection

Triplicate sediment cores were collected from 30 undisturbed temperate inland wetland soils in agricultural landscapes across southern Canada (Fig. 1). A summary of the physical characteristics of these wetlands can be found in Table S1 in the Supplement. These wetlands were undisturbed, with no known history of cultivation. The sediment cores were extracted from the center of the wetland, constituting the open-water area. A Watermark Universal Core (inner diameter of 6.8 cm) or Vibecore Mini with Polycore tubes (inner diameter of 7.6 cm) was used to collect most of the sediment cores. A JMP Backsaver Soil Sampler (inner diameter of 3.2 cm) was used for compacted sediment cores. Shallow (16 to 90 cm) sediment cores were sectioned into 1 or 2 cm increments. Deeper (> 90 cm) sediment cores were sectioned into 5 cm increments. The sediment cores were stored at −5 °C for further processing at the laboratory.

2.2 Generation of ^{137}Cs and ^{210}Pb profiles

Sediment core increments were weighed (wet mass), dried, weighed again (dry mass), disaggregated, homogenized, and sieved. The increments were sieved to remove gravel (> 2 mm): radioisotopes do not bind on the gravel, and gravel does not contain OC; therefore, eliminating gravel improves the estimate of radioisotopes and OC. The increments were counted at 661.6 keV for ^{137}Cs activity and 46.5 keV for ^{210}Pb activity. ^{137}Cs analysis was performed using a gamma spectrometer, and ^{210}Pb analysis was performed using both gamma and alpha spectrometers to increase throughput rates. The gamma analysis was conducted using the high-purity germanium detectors; e.g., Broad Energy Germanium detectors (BE6530) and Small Anode Germanium well Detectors (GSW275L) (Mirion Technologies, Inc., Atlanta, GA, USA). The alpha analysis was conducted using an ORTEC® alpha spectrometer (AMETEK® Advanced Measurement Technology, TN, USA). Both radioisotope analyses were performed at the Landscape Dynamics Laboratory, University of Manitoba, Winnipeg, Canada. Although the underlying principles of gamma and alpha analysis differ, each analysis focuses on quantifying the decay of ^{210}Pb, generating comparable results (Zaborska et al., 2007). Measurement accuracy of gamma detectors is ensured by assessing the counting errors with reference materials within the same geometry as the sample (e.g., Petri dish). The detection error was < 10 % with a counting time of up to 24 h. Furthermore, Landscape Dynamics Laboratory undergoes regular proficiency testing through the International Atomic Reference Material Agency (IARMA), and previously through the International Atomic Energy Agency (IAEA), to ensure acceptable accuracy and precision of analytical results using gamma spectroscopy.

2.3 Screening of ^{137}Cs and ^{210}Pb profiles

Sediment cores were screened to remove profiles with evidence of vertical mixing, then the remaining profiles were used to estimate OC sequestration rates using ^{137}Cs or ^{210}Pb radioisotope dating. The actual ^{137}Cs peak can vary from the expected peak, increasing uncertainty in ^{137}Cs dating (Drexler et al., 2018). ^{137}Cs peaks can be "noisy" or "disturbed" (i.e., flattened, broadened, truncated, mixed, fluctuating; Drexler et al., 2018) or one-sided, where the ^{137}Cs peaks appear at the surface of the sediment core (indicating no or little sedimentation since 1963). The magnitude and shape of the ^{137}Cs peaks observed in the sediments can be affected by the atmospheric deposition rate of ^{137}Cs, which is obviously affected by the number and magnitude of emission events and the weather conditions following these events (UNSCEAR, 2000). The magnitude and shape of these peaks are also impacted by the movement of water and sediment within each wetland's catchment during the peaks' development (Milan et al., 1995; Zarrinabadi et al., 2023). Here, changes in the shape of the peaks are caused by the upward and downward movement of the sediment within the sediment profile (the movement of ^{137}Cs through diffusion (Klaminder et al., 2012) is presumed negligible). Bioturbation can cause an upward and downward mixing of the ^{137}Cs in the profile, resulting in peak attenuation (Robbins et al., 1977). Even wave action during the period of atmospheric deposition will have a similar attenuation effect (Andersen et al., 2000; Zarrinabadi et al., 2023). Following peak atmospheric deposition, soil erosion and the accumulation of sediment will deliver sediments to the top of the profile, and those sediments may be higher or lower in concentration depending on the degree of preferential sediment transport and the associated enrichment or depletion of ^{137}Cs in the added sediment (Zarrinabadi et al., 2023). Such noise in ^{137}Cs peaks needs careful interpretation to avoid over- or underestimating the OC sequestration rates.

Selecting suitable cores: Of the 90 sediment cores (30 wetlands × 3 replicates = 90), 79 were suitable (complete and datable) for ^{137}Cs dating and 47 were suitable for ^{210}Pb dating. Only some replicates from the same wetland were ideal for interpretation or further screening. The suitability of ^{137}Cs profiles for dating was assessed by zero activity before the onset and peak of ^{137}Cs activity. The suitability of ^{210}Pb profiles for dating was evaluated by determining the exponential decline in ^{210}Pb activity with depth until background levels were reached.

Classification of the selected ^{137}Cs *profiles*: The 79 suitable ^{137}Cs profiles were then classified into high- and low-quality using the following steps (Fig. 2a):

1. The ^{137}C depth profile and the shape of the peak were assessed. A clear and distinct peak associated with several points on both sides of the peak verified the ^{137}Cs depth profile as high-quality (e.g., Fig. 3a).

2. When analyzing sediment samples, a clear peak in ^{137}Cs activity did not always exist (e.g., Fig. 4a). If a peak was absent, which could have resulted from sediment influxes with very high or very low ^{137}Cs activity levels, the total ^{137}Cs activity of the entire profile was examined. If the cumulative ^{137}Cs inventory value for the entire profile was $\geq 500\,\mathrm{Bq\,m^{-2}}$, then the ^{137}Cs

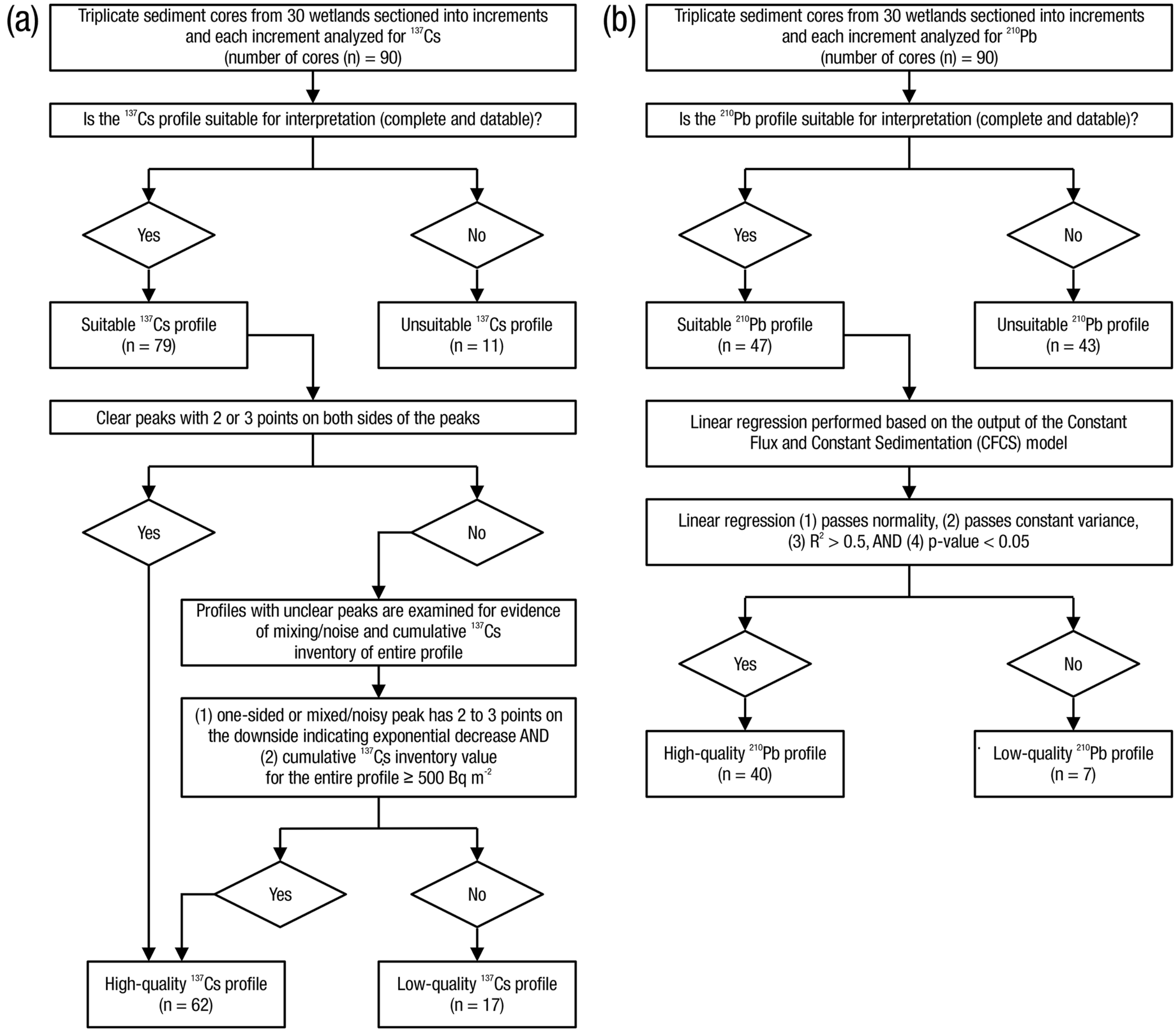

Figure 2. Classification of high- and low-quality ^{137}Cs and ^{210}Pb profiles outlining the decision frameworks for screening **(a)** ^{137}Cs and **(b)** ^{210}Pb profiles.

profile was considered high-quality. Conversely, if the cumulative ^{137}Cs inventory value for the entire profile was $< 500\,\mathrm{Bq\,m^{-2}}$, the ^{137}Cs profile was considered low-quality. The cut-off cumulative ^{137}Cs inventory value of $500\,\mathrm{Bq\,m^{-2}}$ was established by assessing the ^{137}Cs reference inventory value, the value of ^{137}Cs present in a non-eroded system with an undisturbed profile. The ^{137}Cs reference inventory value differs from region to region (Owens and Walling, 1996), and the most proximal regional value was used to select the cut-off ^{137}Cs inventory value (Sutherland, 1991; Kachanoski and Von Bertoldi, 1996; Zarrinabadi et al., 2023). The ^{137}Cs reference inventory is a catchment-wide reference value and not specific to the wetland center; thus, the cumulative ^{137}Cs inventory value of $500\,\mathrm{Bq\,m^{-2}}$ was viewed as a conservative indicator of the suitability of the ^{137}Cs profiles. Ideally, reference sites are large, open, level, non-eroded areas, usually in forage or grassland since the 1950s and within 10 km of the site of interest. In this study, it was impossible to identify a suitable reference site near every wetland; it is usually difficult to find reference sites in agricultural landscapes. However, we could locate reference sites used in other studies within 50 km except from nine wetlands in SK (51° N, 104° W), which were ~ 150 km from the reference site. Although this was not considered ideal, it was considered acceptable.

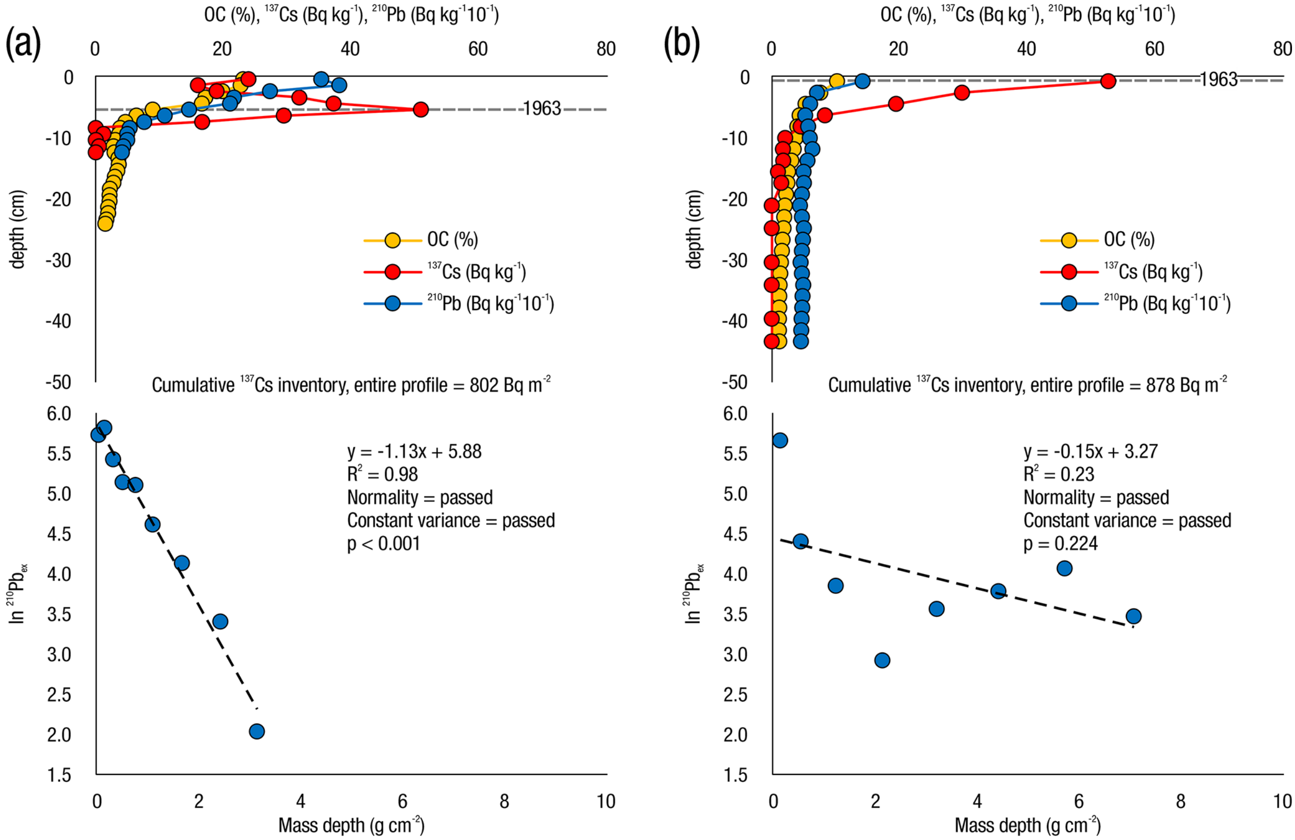

Figure 3. Examples of ^{137}Cs and ^{210}Pb classifications showing OC (%), ^{137}Cs (Bq kg^{-1}), and ^{210}Pb (Bq kg^{-1}) depth profiles and plots of log-transformed ^{210}Pb$_{\mathrm{ex}}$ against mass depth (g cm^{-2}): **(a)** high-quality ^{137}Cs and high-quality ^{210}Pb (S-LO-I-W3-T1-CW-R1) and **(b)** high-quality ^{137}Cs and low-quality ^{210}Pb (M-OA-I-W4-T3-CW-R3).

Classification of the selected ^{210}Pb *profiles*: The 47 suitable ^{210}Pb profiles were classified into high- and low-quality profiles based on the following steps (Fig. 2b):

1. ^{210}Pb activity was plotted with a log-transformed ^{210}Pb$_{\mathrm{ex}}$ against mass depth (g cm^{-2}).

2. A linear regression analysis was performed (where the slope is used to derive the mass or sediment accumulation rate in g cm^{-2} yr^{-1}).

3. If the linear regression passed both normality and constant variance tests and had an $R^2 > 0.5$ and a p value < 0.05, then the ^{210}Pb profile was classified as high-quality (e.g., Fig. 3a).

4. If either normality or constant variance tests were not passed, with an $R^2 \leq 0.5$ or a p value ≥ 0.05, then the ^{210}Pb profile was considered low-quality (e.g., Fig. 3b).

The ^{210}Pb profiles were also classified using a two-step piecewise linear regression model to capture recent shifts in OC sequestration rates. However, no significant improvement was observed. Consequently, ^{210}Pb-based OC sequestration rates were derived from the linear regression line. An $R^2 > 0.5$ was selected as the cut-off for selecting high-quality over low-quality profiles. Increasing the cut-off R^2 value may produce better profiles to be chosen for the study. Still, it can reduce the number of available sediment cores and potentially ignore the natural variability and significant events occurring in the real environment.

2.4 Organic carbon stocks and sequestration rates

Radioisotope activity measurements were utilized to assign two time markers, one for 1954 and the other for 1963, in the sediment cores. Sediment radioisotope dating was used to calculate the rates of sediment or mass accumulation and OC sequestration.

For ^{137}Cs dating, sediment accumulation and OC sequestration rates (Mg ha^{-1} yr^{-1}) were estimated using the cumulative sum of sediment or OC (Mg ha^{-1}) from the surface to the depth corresponding to the time markers of ^{137}Cs of each core and dividing by the number of years from the time marker to the years the samples were collected. Unit conversion is applied to report the OC sequestration rate estimates in Mg ha^{-1} yr^{-1} from g cm^{-2} yr^{-1} for easy standardization and comparability with other studies. For ^{137}Cs profiles with

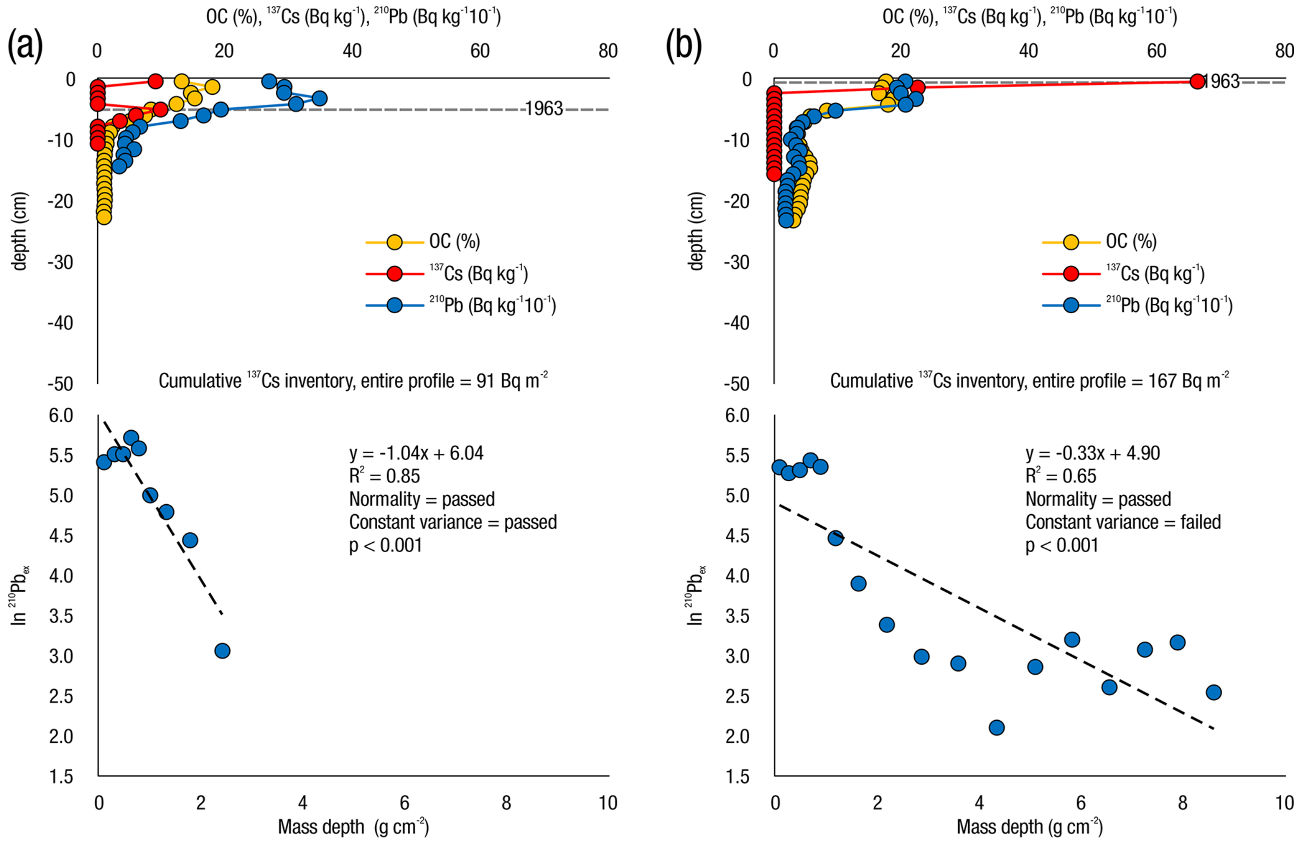

Figure 4. Examples of ^{137}Cs and ^{210}Pb classifications showing OC (%), ^{137}Cs (Bq kg^{-1}), and ^{210}Pb (Bq kg^{-1}) depth profiles and plots of log-transformed ^{210}Pb$_{ex}$ against mass depth (g cm^{-2}): **(a)** low-quality ^{137}Cs and high-quality ^{210}Pb (S-RO-I-W1-T3-CW-R3) and **(b)** low-quality ^{137}Cs and low-quality ^{210}Pb (S-RO-I-W6-T1-CW-R1).

noisy peaks and comparatively larger cumulative ^{137}Cs inventory values, the first elongated peak with a sharp rise after the onset of the ^{137}Cs peak was considered the 1963 peak instead of the peak with the highest activity in the profile.

For ^{210}Pb dating, mass or sediment accumulation and OC sequestration rates were estimated using the constant flux–constant sedimentation (CFCS) model (Sanchez-Cabeza and Ruiz-Fernandez, 2012; Kamula et al., 2017). Here, ^{210}Pb$_{ex}$ was estimated by subtracting ^{226}Ra activity (186 keV) from the total ^{210}Pb activity. The CFCS model uses the log-linear relationship of ^{210}Pb$_{ex}$ with mass depth and converts ^{210}Pb$_{ex}$ to the mass or sediment accumulation rate and, consequently, the OC sequestration rate. The OC stock was estimated by taking the cumulative sum of OC (Mg ha^{-1}) from the surface of each sediment core to the depth increments represented by the time marker (e.g., 1963).

OC stocks for the 1954 and 1963 time markers were calculated by multiplying the OC content per unit mass of soil (g). Here, OC content was calculated from OC concentration (%) measured by the loss-on-ignition (LOI) method (Kolthoff and Sandell, 1952; Dean, 1974) by the mass of sediment for each section interval and specific depth interval per unit area (g cm^{-2}) down the profile to the respective time marker. OC (%) was calculated by multiplying organic matter (%) by LOI with 0.58, assuming 58 % of the organic matter is carbon. Despite the broad applicability, simplicity in measurement techniques, and cost-effectiveness, the LOI approach is associated with some limitations, such as the ignition of non-organic particles at high temperatures or the use of a conventional conversion factor (Pribyl, 2010; Hoogsteen et al., 2015), which can result in overestimation of OC content.

2.5 Statistical analysis

Statistical analyses used sediment cores with ^{137}Cs- and ^{210}Pb-based OC sequestration rates available (number of sediment cores ($n = 44$). The ^{137}Cs- and ^{210}Pb-based estimates of OC sequestration rates were compared using a quantile–quantile (Q–Q) plot. Firstly, the comparison was done via assessment of the Q–Q plots. Four sample datasets were used to construct Q–Q plots to compare the distribution of ^{137}Cs- and ^{210}Pb-based OC sequestration against the 1 : 1 line.

The sample datasets included the following:

- D1, all suitable ^{137}Cs and ^{210}Pb profiles with OC sequestration rates estimated since 1954 ($n = 44$),

- D2, all suitable ^{137}Cs and ^{210}Pb profiles with OC sequestration rates estimated since 1963 ($n = 44$),
- D3, high-quality ^{137}Cs and ^{210}Pb profiles with OC sequestration rates estimated since 1954 ($n = 30$),
- D4, high-quality ^{137}Cs and ^{210}Pb profiles with OC sequestration rates estimated since 1963 ($n = 30$).

A Q–Q plot was calculated for each dataset. The x and y coordinates of a point in a Q–Q plot corresponded to the path percentiles of the two OC sequestration rate estimates being compared in the plot. Here, $p = (k - 0.5)/n$, where n is the sample size and $k = 1, \ldots, n$ (Jain et al., 2007). The distribution of the points on the Q–Q plot was compared against the $y = x$ (1 : 1) line to assess whether the two OC sequestration rate estimates are similar. If the points were distributed in a straight line and close to a 1 : 1 line, then it suggested that the two estimates came from the same distribution. In contrast, if the points were not distributed in a straight line or deviated from the 1 : 1 line, then it suggested that the two estimates did not come from the same distribution. The Q–Q plots were generated in Microsoft Excel (Microsoft 365, Version 2402, Redmond, WA).

Since interpreting the Q–Q plot through a visual inspection can be subjective to human perception, we compared the ^{137}Cs- and ^{210}Pb-based OC sequestration rate estimates using a distance sampling model. A distance sampling model captures how the detectability of objects from the observer (walking along a straight line) decreases with the increase in the object-to-observer distance. If the objects are closely distributed along the observer's path (i.e., if points of the Q–Q plot were closely distributed along the 1 : 1 line), then the distribution of the distances is expected to be a half-normal distribution. The Cramer–von Mises test was used to estimate whether the distances ($q1, q2, \ldots, qn$) from the points to the 1 : 1 line were from a half-normal distribution. Given a set of distance samples ($q1, q2, \ldots, qn$) and a detection function, the Cramer–von Mises test builds a model that fits the distance sampling data to the detection function (for details on modeling, see Miller et al., 2019). A half-normal key is commonly used as a detection function, corresponding to a half-normal distribution's shape.

The Cramer–von Mises test produced a p value and Akaike's information criterion (AIC) as its test statistics. A p value larger than the significant level ($p = 0.05$) indicated that the likelihood of points being observed closer to the 1 : 1 line is high and that the probability decreases as the distances increase. This provided evidence of the points being closely distributed along the 1 : 1 line. The AIC was used to rank the distance sampling models, which are built by the Cramer–von Mises tests, from best to worst (e.g., Burnham and Anderson, 2003): a small AIC value indicates a good fit to the half-normal key and thus provides evidence that the points are close to the 1 : 1 line (Miller et al., 2019). The distance-sampling Cramer–von Mises test was computed using the "distance" package in R version 4.0.3 (Miller and Clark-Wolfe, 2023; R Core Team, 2023).

3 Results

3.1 High- and low-quality ^{137}Cs and ^{210}Pb profiles

Of the 79 suitable ^{137}Cs profiles, 62 (78 %) were classified as high-quality. Of the 62 high-quality ^{137}Cs profiles, 61 % had clear and distinct peaks, with a smooth rise and decline. In contrast, the remaining 39 % had noise – either one-sided peaks or disturbed peaks (e.g., Fig. 4). Of the 62 high-quality ^{137}Cs profiles, 4 (6.5 %) were repositioned to capture the ^{137}Cs-enriched sediments post-1963 (e.g., ^{137}Cs profile of S-LO-I-W4-T2-CW-R2 in Fig. S2a in the Supplement). In these profiles, which had a cumulative ^{137}Cs inventory value $> 1200\,\mathrm{Bq\,m^{-2}}$, the depth that corresponded to the ^{137}Cs cumulative inventory value of $\sim 500\,\mathrm{Bq\,m^{-2}}$ was regarded as the 1963 time marker. The high total quantities of ^{137}Cs profile inventories can be attributed to receiving ^{137}Cs-enriched sediments from the surrounding landscape. Sediments that have undergone substantial preferential detachment and entrainment on their pathway into a wetland can have very high concentrations of ^{137}Cs and, when interlayered with sediments that are not so enriched, can generate multiple ^{137}Cs peaks in the sediment profile peaks after 1963. These observed multiple peaks are local and not regional, ruling out the association with the Chernobyl and Fukushima events. Two ^{137}Cs profiles were considered high-quality despite a cumulative ^{137}Cs inventory value $< 500\,\mathrm{Bq\,m^{-2}}$ because the 1963 peak was clear, distinct, and elongated with two to three points on both sides of the peak (e.g., ^{137}Cs profile of M-OA-I-W4-T2-CW-R2 in Fig. S7b in the Supplement). One ^{137}Cs profile was considered high-quality despite showing marginal quality to the set criteria in the decision framework, where the peak profile had good shape with several points on the downside of the peak and one point on the other side and had a cumulative ^{137}Cs inventory value of $499\,\mathrm{Bq\,m^{-2}}$. One ^{137}Cs profile was classified as low-quality despite a cumulative ^{137}Cs inventory value $> 500\,\mathrm{Bq\,m^{-2}}$ because the peak was highly fluctuating and not discernible (e.g., ^{137}Cs profile of O-AL-I-W6-T1-CW-R1 in Fig. S12b in the Supplement).

Of the 47 ^{210}Pb profiles, 40 (85 %) were classified as high-quality.

There were 44 sediment cores with both ^{137}Cs- and ^{210}Pb-suitable profiles available. Of these, 30 were categorized as high-quality ^{137}Cs and high-quality ^{210}Pb (Fig. 3a), 6 were categorized as high-quality ^{137}Cs and low-quality ^{210}Pb (Fig. 3b), 7 were classified as low-quality ^{137}Cs and high-quality ^{210}Pb (Fig. 4a), and 1 was categorized as low-quality ^{137}Cs and low-quality ^{210}Pb (Fig. 4b); see Figs. S1 to S12 in the Supplement for ^{137}Cs and ^{210}Pb profiles in all study wetlands.

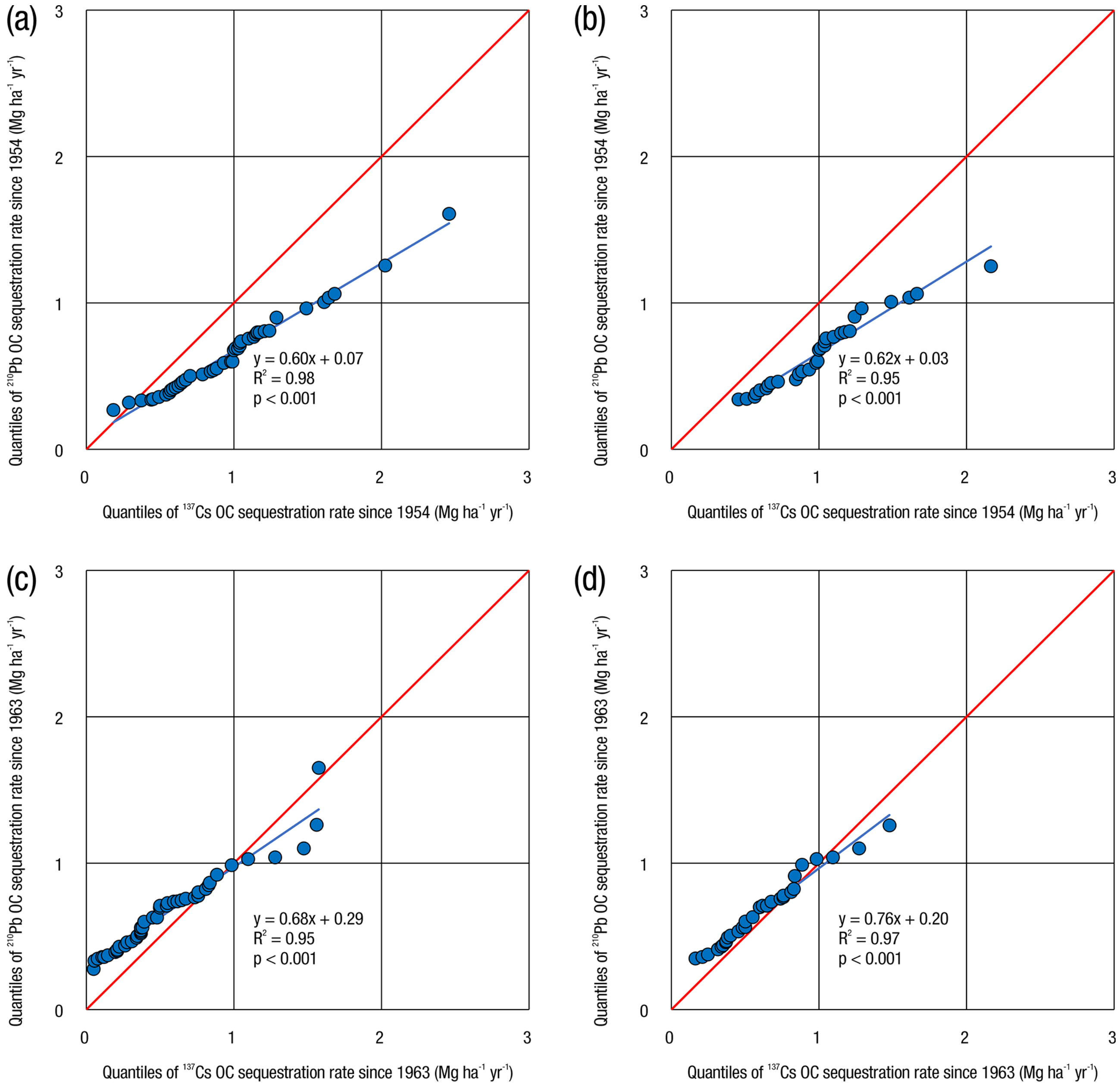

Figure 5. Q–Q plot of ^{137}Cs- vs. ^{210}Pb-based organic carbon (OC) sequestration rates using **(a)** all suitable ^{137}Cs and ^{210}Pb profiles estimated since 1954 (D1), **(b)** high-quality ^{137}Cs and ^{210}Pb profiles estimated since 1954 (D3), **(c)** all suitable ^{137}Cs and ^{210}Pb profiles estimated since 1963 (D2), and **(d)** high-quality ^{137}Cs and high-quality ^{210}Pb profiles estimated since 1963 (D4).

3.2 ^{137}Cs- vs. ^{210}Pb-derived organic carbon sequestration rates

For each of the four datasets (D1–D4), the points on the Q–Q plot were distributed in a straight line, showing a linear relationship between the two estimates being compared ($R^2 > 0.95$, p value < 0.001) (Fig. 5).

Visual inspection of the Q–Q plots showed that the points for D2 (i.e., all suitable ^{137}Cs and ^{210}Pb profiles using the 1963 time marker; Fig. 5c) and D4 (i.e., high-quality ^{137}Cs and ^{210}Pb profiles using the 1963 time marker; Fig. 5d) were distributed more closely along the 1 : 1 line compared to those of D1 (i.e., all suitable ^{137}Cs and ^{210}Pb profiles using the 1954 time marker; Fig. 5a) and D3 (i.e., high-quality ^{137}Cs and ^{210}Pb profiles using the 1954 time marker; Fig. 5b).

An intercept closer to 0 and a slope closer to 1 indicated good alignment of the regression line to the 1 : 1 line. The slope (s) and intercept (i) of the regression lines were: $s = 0.60$, $i = 0.07$ for D1 (Fig. 5a); $s = 0.62$, $i = 0.03$ for D3

Table 2. Sedimentation accumulation, organic carbon (OC) sequestration rates, and OC stocks of undisturbed wetlands estimated using high-quality ^{137}Cs and high-quality ^{210}Pb profiles.

Type of radiometric dating	^{137}Cs		^{210}Pb	
Time marker	1954	1963	1954	1963
Range of accumulated sediment (Mg ha^{-1})	214–1727	56–1272	111–1014	95–874
Mean (standard deviation) stock of OC (Mg ha^{-1})	66 (29)	35 (19)	43 (18)	38 (15)
Mean (standard deviation) rate of OC sequestration (Mg ha^{-1} yr^{-1})	1.02 (0.44)	0.63 (0.34)	0.67 (0.27)	0.68 (0.26)

(Fig. 5b); $s = 0.68$, $i = 0.29$ for D2 (Fig. 5c); and $s = 0.76$, $i = 0.20$ for D4 (Fig. 5d). D2 and D4 had regression lines and slopes closer to the 1 : 1 line but intercepts further from the origin than D1 and D3.

The Cramer–von Mises test was used to build distance-sampling models using the point-to-1 : 1-line distances computed from the Q–Q plots. Models built with the D4 dataset produced the best-fit model (i.e., p value > 0.05, AIC $= -114$). Models built with the D1, D2, and D3 datasets had weaker p values (p value < 0.05) and can be ranked based on lower AIC scores (AIC $= -116$ for D2, AIC $= -54$ for D1, and AIC $= -34$ for D3).

3.3 Sediment accumulation, organic carbon sequestration rates, and organic carbon stocks

The 30 sediment cores (68 % of all the suitable ^{137}Cs and ^{210}Pb profiles) with high-quality ^{137}Cs and ^{210}Pb profiles were used to calculate mass or sediment accumulation rates, OC sequestration rates, and OC stocks (Table 2). OC sequestration rates based on ^{137}Cs and ^{210}Pb dating estimated since 1954 and 1963 of 44 suitable sediment cores (where both ^{137}Cs and ^{210}Pb profiles were available) are presented in Table S2 in the Supplement.

Based on the 1954 time marker, the total sediment accumulation ranged from 214–1727 Mg ha^{-1} using ^{137}Cs dating and 111–1014 Mg ha^{-1} using ^{210}Pb dating. In contrast, the total sediment accumulation based on the 1963 time marker was lower, ranging from 56–1272 Mg ha^{-1} using ^{137}Cs dating and 95–874 Mg ha^{-1} using ^{210}Pb dating.

The ^{137}Cs-derived mean OC sequestration rate was almost 2 times larger, at 1.02 Mg ha^{-1} yr^{-1} using the 1954 time marker compared to 0.63 Mg ha^{-1} yr^{-1} using the 1963 time marker. The corresponding ^{137}Cs-based mean OC stocks were 66 Mg ha^{-1} for 1954 and 35 Mg ha^{-1} for 1963 (Table 2).

The ^{210}Pb-derived mean OC sequestration rate was similar at 0.67 Mg ha^{-1} yr^{-1} using the 1954 time marker compared to 0.68 Mg ha^{-1} yr^{-1} using the 1963 time marker. ^{210}Pb-based OC sequestration rates show minimal variation, since they were calculated using the same sedimentation rate. The corresponding mean OC stocks were 43 Mg ha^{-1} for the 1954 time marker and 38 Mg ha^{-1} for the 1963 time marker, with a variable depth.

Figure 3 and Figs. S1 to S12 in the Supplement present the depth distributions of ^{137}Cs and ^{210}Pb activity (along with the linear plots of log-transformed ^{210}Pb$_{ex}$ against mass depth in g cm^{-2}) of all suitable profiles ($n = 44$) where both radioisotope profiles are available.

4 Discussion

This study compared ^{137}Cs and ^{210}Pb dating for OC estimates in wetlands that were undisturbed (i.e., without direct impact human activities), since the dating of both radioisotopes is known to provide reliable forecasts for recent OC sequestration rates (i.e., post-1954, which coincides with the onset of ^{137}Cs atmospheric deposition) (Drexler et al., 2018; Creed et al., 2022).

This study highlights some advantages and disadvantages of using ^{137}Cs vs. ^{210}Pb dating. For example, the smaller number of suitable ^{210}Pb profiles (47/90 = 52 %) due to the lack of a complete decay profile (following the CFCS model as described in Sanchez-Cabeza and Ruiz-Fernandez, 2012) indicates that ^{210}Pb dating is more prone to disturbance than ^{137}Cs dating (79/90 = 88 %). For ^{137}Cs, even if the sediment core is disturbed, estimation of OC sequestration rates may be possible with careful interpretation (e.g., see Fig. 2). The larger number of sediment cores using ^{137}Cs dating can be beneficial in accurately representing the heterogeneity of OC sequestration rates, as it provides a larger dataset (a 36 % gain compared to ^{210}Pb).

Other advantages and disadvantages of ^{137}Cs vs. ^{210}Pb radioisotope dating are presented in Table 3. ^{137}Cs deposition was a pulse that occurred in 1954 and 1963. At the 1963 peak, the activity declined with time because of two factors: (1) peak natural radioactive decay, with the ^{137}Cs 30-year half-life reducing the peak size over time, and (2) peak attenuation due to physical, chemical, or biological reasons (Drexler et al., 2018). The declining ^{137}Cs activity limits its applicability as a radioisotope dating tool; however, recent studies have reported adequate ^{137}Cs reference inventories for Canadian landscapes (Sutherland, 1991; Kachanoski and Von Bertoldi, 1996; Li et al., 2008; Mabit et al., 2014; Zarrinabadi et al., 2023). In addition, the use of the ^{137}Cs inventory for dating to complement the peak has addressed the potential inadequacies that could be attributed to declining peak resolution with time. ^{137}Cs dating is advantageous for

Table 3. The advantages and disadvantages of using ^{137}Cs and unsupported ^{210}Pb (^{210}Pb$_{ex}$) to estimate wetland organic carbon (OC) sequestration rates.

Method of radiometric dating	^{137}Cs	^{210}Pb$_{ex}$
Advantages	– Calculations are based on single points representing the peak (1963) and onset (1954) of the fallout. There are additional time markers for Europe (1986 due to the Chernobyl nuclear accident) and Japan (2011 due to the Fukushima Daiichi nuclear accident). – The sedimentation peak may still be evident, allowing the estimation of the OC sequestration rate even if parts of the sediment core are disturbed. – The sedimentation rate can be estimated using gamma detection, which is non-destructive, so the sample can be re-analyzed or used for other analyses. – Less sample preparation time for gamma analysis. – After the ^{137}Cs activity is measured, post-processing of data is less challenging.	– Calculations are based on multiple points, as there is continuous atmospheric deposition. – The sedimentation rate can be estimated using two reliable methods, i.e., both alpha and gamma detection. – Less sample preparation time for gamma analysis compared to alpha. – Gamma analysis is non-destructive, so samples can be re-analyzed for other analyses compared to alpha. – Can run multiple samples at a time on a single detector in alpha method.
Disadvantages	– Risk of mixing of restored and drained states when estimating OC sequestration rates due to specificity of the 1954 and 1963 time markers (e.g., if drained and restored after 1963). – Declining atmospheric deposition and declining inventory due to radioactive decay (i.e., with no more nuclear testing, atmospheric deposition only comes from recent accidental releases from Chernobyl and Fukushima). – Sometimes the peak is not distinct. – Can be estimated using only one reliable method, i.e., gamma detection. – Can run only one sample at a time on a single detector. – Sensitive to vertical mobilization of sediments. – Sensitive to declining ^{137}Cs inventory due to radioactive decay. – Sensitive to changes in redox potential. – More sensitive to biological and chemical activity compared to ^{210}Pb (e.g., ^{137}Cs can be taken up by plants instead of sodium or potassium, and ^{137}Cs is soluble and therefore subject to mobility into solution then moving up and down the core).	– Requires full profile of ^{210}Pb to do the calculations; if the sediment core is disturbed, then it cannot be used to estimate OC sequestration rates. – Sensitive to vertical mobilization of sediments, but not as much as ^{137}Cs. – The alpha method is destructive; therefore the sample is not available for re-use or re-analysis. – The alpha method requires extra precaution using hydrochloric acid for digesting, heating, and spiking with a ^{209}Po tracer (i.e., analysts come into direct contact with the radioactive material ^{209}Po and hot acid). – The alpha method takes more time per sample (i.e., overnight digest followed by at least 48 h on the alpha counter) and is more labor-intensive (i.e., digesting, engraving coins, plating, transferring into ensemble). – The alpha method requires more technical expertise for post-processing of the data. – Uncertainty of ^{210}Pb$_{ex}$ results derived from gamma analysis can be higher than from alpha.

its simplicity in pre- and post-processing of samples and the presence of additional time markers in other regions (Breithaupt et al., 2018; Foucher et al., 2021). For example, additional time markers correspond to the 1986 Chernobyl nuclear plant accident and the 2011 Fukushima accident. However, their effect has yet to be recorded in North America due to the substantial distance from the source. Recognizing that there may be regional or local variation in peaks, we used non-eroded ^{137}Cs reference sites to deal with regional variation in deposition. We also used multiple sampling sites within wetlands to assess local variation in deposition. Furthermore, we looked for evidence from Chernobyl and Fukushima nuclear events in our data but found none (data not shown).

Furthermore, we looked for evidence from the Chernobyl and Fukushima nuclear events in our data but found none (data not shown). ^{137}Cs dating is best suited for where the total OC is sequestered since a fixed time marker (1954 onset or 1963 peak) or where the average OC sequestration rate is needed. In contrast, the atmospheric deposition of ^{210}Pb is continuous and therefore not limited in its applicability as a radioisotope dating tool. ^{210}Pb dating is best suited for where variable OC sequestration rates are needed over a more extended period (earlier than 1954). ^{210}Pb dating is advantageous because its calculations are based on multiple points associated with progressive OC sequestration rates derived using a constant rate of supply model – including the 1954 onset and 1963 peak of ^{137}Cs activity – improving the precision of the OC sequestration rates. This precision enables the estimation of OC sequestration rates when wetlands are not undisturbed (history of drainage or at different ages since restoration) and when they are undisturbed (no history of drainage).

4.1 Challenges in interpreting the ^{137}Cs peak

A potential weakness of ^{137}Cs radioisotope dating arises from the challenges in interpreting the disturbed 1963 peak. The noise in the 1963 peak in wetlands on agricultural landscapes can be due to the redistribution of sediments, since wetlands are susceptible to receiving a large mass of sediments resulting from various erosional processes due to their positioning within the landscape (Lobb, 2011; Zarrinabadi et al., 2023). Soil erosion resulting from wind, water, and tillage can lead to higher or lower ^{137}Cs levels (Li et al., 2010; Foucher et al., 2021; Zarrinabadi et al., 2023) in wetlands in agricultural landscapes. If ^{137}Cs-enriched soil from the surrounding landscape gets deposited on top of the wetland's original soil layer, it can increase the ^{137}Cs inventory value (Walling and Quine, 1991; Li et al., 2010). The magnitude of ^{137}Cs enrichment depends on whether sediment comes from surface or sub-surface layers (Li et al., 2010; Lal, 2020). For example, if the wetland receives ^{137}Cs-enriched topsoil post-1963, the ^{137}Cs inventory would be higher than the ^{137}Cs-depleted subsoil.

The screening of ^{137}Cs profiles (Fig. 2a) considered the redistribution of sediments within the landscape. It demonstrated that the difficulty in disturbed ^{137}C profile interpretation can be reduced by investigating the cumulative ^{137}C inventory value. A cut-off cumulative ^{137}Cs inventory value can help exclude questionable profiles. The range of ^{137}Cs reference inventory values from previous erosion studies within the study area (e.g., Sutherland, 1991; Kachanoski and Von Bertoldi, 1996; Zarrinabadi et al., 2023) can help in establishing and setting the cut-off cumulative ^{137}Cs inventory value. The mean ^{137}Cs reference inventory values in the four provinces of Canada where our wetland sites are located were utilized in this instance. The mean ^{137}Cs reference inventory value is estimated to be 1684 Bq m^{-2} (coefficient of variation (CV) = 49 %) for three AB wetland sites (53° N, 113° W) (Zarrinabadi et al., 2023), 989 Bq m^{-2} (CV = 20 %) for seven SK wetland sites (51° N, 107° W) (Sutherland, 1991), 1008 Bq m^{-2} (CV = 17.9 %) for nine SK wetland sites (51° N, 104° W) (Sutherland, 1991), 1430 Bq m^{-2} (CV = 8.6 %) for five MB wetland sites (50° N, 100° W) (Zarrinabadi et al., 2023), 1447 Bq m^{-2} (CV = 8.8 %) for three ON wetland sites (43.3° N, 80.3° W) (Kachanoski and Von Bertoldi, 1996), and 1534 Bq m^{-2} (CV = 1.7 %) for three ON wetland sites (45.6° N, 74.8° W) (Kachanoski and Von Bertoldi, 1996). The ^{137}Cs reference inventory values were decay-corrected to 2021 for comparability. The cut-off cumulative ^{137}Cs inventory value for this study was selected by checking the minimum ^{137}Cs reference inventory value of the local region, i.e., 546 Bq m^{-2} (using values reported in Sutherland, 1991; Kachanoski and Von Bertoldi, 1996; and Zarrinabadi et al., 2023). Hence, any ^{137}Cs inventory value less than 500 Bq m^{-2} was considered questionable and low-quality. Additionally, > 75 % of ^{137}C profiles had a cumulative ^{137}Cs inventory value of > 500 Bq m^{-2}, indicating that the ^{137}Cs reference inventory value of our wetland sites is most likely around 500 Bq m^{-2}.

Variations in the ^{137}Cs peak types (e.g., distinct, broadened, fluctuating) and in ^{137}Cs inventory values in this study suggested that the ^{137}Cs profiles were impacted by various regional erosional processes in the surrounding agricultural landscape. Recent evidence suggests that there may be an outward movement of sediment and ^{137}Cs from the center of the wetlands to the riparian area (Zarrinabadi et al., 2023), suggesting that the base ^{137}Cs inventory value observed in the center of wetlands from atmospheric deposition in the 1950s–1960s could be less than that of the non-eroded reference ^{137}Cs values from the surrounding catchment. A ^{137}Cs inventory of a sediment core can further help assign the ^{137}Cs peak. For example, the ^{137}Cs peak was repositioned in disturbed sediment cores with a higher ^{137}Cs inventory, where the first discernable peak after the sharp rise from the onset of ^{137}Cs activity and exceeding or around the reference value was assumed to be the original ^{137}Cs peak. $^{239+240}$Pu isotopes, like ^{137}Cs, are a product of nuclear testing and can be used to identify the peak of ^{137}Cs. Future research will use $^{239+240}$Pu to replace ^{137}Cs as ^{137}Cs levels diminish.

4.2 Challenges in interpreting ^{137}Cs and ^{210}Pb profiles

Mobilization of ^{137}Cs and ^{210}Pb in the sediment often occurs in wetlands. Vertical mixing of ^{137}Cs within sediments can be caused by remobilization and redistribution by wind and water, ice movement and inversion, disturbance by animals, and disturbance by humans who ditch and drain the wetlands when they are dry and let cattle access them for water, which causes disturbances to the bottom sediments (Robbins et al., 1977; Milan et al., 1995; Takahashi et al., 2015). Vertical mixing affects the profile by attenuating the peak upward and downward (which we addressed using the ^{137}Cs inventory value and not just the peak when assessing the profile). Horizontal mixing of ^{137}Cs dating within sediment occurs by physical movement of sediments into or out of the wetland, causing uneven distribution of the OC content, where accumulation may be high at the edges of open water of the wetland (Lobb, 2011; Zarrinabadi et al., 2023). This heterogeneity can be caused by the horizontal focusing of sediments in sub-basins within a wetland, i.e., multiple center points. Sampling multiple sediment cores from individual wetlands can help capture the heterogeneity within the wetland. Suppose the ^{137}Cs activity of most of the sediment cores from a particular wetland is noisy with a higher inventory value (e.g., ^{137}Cs profile of S-LO-I-W4-T2-CW-R2 in Fig. S2a in the Supplement). In that case, the impact by erosional processes can be deduced with higher certainty. The higher observed inventory value could result from the movement of enriched material via erosion/lateral flow to the center of the wetland, increasing the amount of ^{137}Cs. In this study, the assumption of no substantial downward mixing of ^{137}Cs was supported by (1) sampling three cores from each wetland, (2) assessing the sharpness of the rise of the peaks (a sharp rise means negligible mixing), (3) examining the cumulative ^{137}Cs inventory value and validating against the known reference level, (4) classifying ^{137}Cs profiles, and (5) corroborating with ^{210}Pb dating.

4.3 ^{137}Cs- vs. ^{210}Pb-derived organic carbon sequestration rates and stocks

^{137}Cs radioisotope dating using the 1954 or 1963 time markers gives reasonable estimates of OC sequestration rates as compared to ^{210}Pb radioisotope dating. The ^{137}Cs–^{210}Pb Q–Q plot of the 1963 OC sequestration rates is closer to the 1:1-line, suggesting compatibility between ^{137}Cs- and ^{210}Pb-based estimates (Fig. 5c and d). Conversely, the ^{137}Cs–^{210}Pb Q–Q plot of the 1954 OC sequestration rates showed more deviation from the 1 : 1 line; ^{137}Cs-based OC sequestration rates were more dispersed and were higher than the ^{210}Pb-based OC sequestration rates (Fig. 5a and b). The mean OC sequestration rates in Table 2 further verify the comparability of OC sequestration rates using the 1963 time marker (the mean ^{137}Cs OC sequestration rate is 0.63 $\mathrm{Mg\,ha^{-1}\,yr^{-1}}$, while the mean ^{210}Pb OC sequestration rate is 0.68 $\mathrm{Mg\,ha^{-1}\,yr^{-1}}$). For the dispersion using the 1954 time marker, the mean ^{137}Cs OC sequestration rate is 1.02 $\mathrm{Mg\,ha^{-1}\,yr^{-1}}$, while the mean ^{210}Pb OC sequestration rate is 0.67 $\mathrm{Mg\,ha^{-1}\,yr^{-1}}$. Providing better sequestration rate estimates has consequences for estimating OC stocks with an improved degree of accuracy, which may provide policymakers with better tools to make informed carbon management decisions supported with data.

To put our findings into practice and into the broader OC sequestration perspective, we consider a scenario where two independent studies were performed using ^{137}Cs and ^{210}Pb (with the CFCS model) at the exact locations. If the cores were not selected based on the criteria we used to choose high-quality profiles, then these two studies' OC sequestration rate estimates are likely to disagree. However, we know and have demonstrated through our findings that they are linearly dependent and that the equation of our linear regression lines may be used to transform one estimate to the other. Consequently, if the cores were selected based on our selection criteria, then one can expect the OC sequestration rate estimates to have similar values, which alleviates the interpretation challenges of having two different estimates from two independent studies. This observation may help with consistency when disagreements in estimates are observed. Another practical application of our findings may be in data augmentation. For example, if we have ^{210}Pb data for a set of locations and ^{137}Cs data for other locations, the linear regression equation could transform ^{210}Pb data to augment ^{137}Cs data and vice versa. This can help data-driven modeling approaches, whereas larger datasets help achieve robust modeling tools. Similarly, because OC stocks can be derived from sequestration rates for specific years, estimates derived using one radioisotope can be used to estimate OC from a dataset derived from another estimate, further contributing to the augmentation of the corresponding OC stock data.

Based on the results of this study, we recommend (1) using high-quality ^{137}Cs and ^{210}Pb profiles to estimate OC sequestration rates, (2) carefully interpreting ^{137}Cs profiles from agricultural landscapes from the perspective of redistribution of sediments, and (3) using both ^{137}Cs and ^{210}Pb to compare and validate estimates if logistically possible. However, in cases where one has to choose between ^{137}Cs and ^{210}Pb, we recommend the following: (1) for ^{137}Cs, use 1963 time markers to estimate OC sequestration rates (compared to 1954), since it is found to be most comparable with ^{210}Pb dating techniques (CFCS model); (2) for ^{210}Pb (CFCS model), OC sequestration rates from 1963 to present can be estimated with the highest precision, since we corroborated the estimates with ^{137}Cs. However, we cannot comment on the precision of ^{210}Pb-based OC sequestration rate estimation before 1963 based on the scope of this study.

5 Conclusions

Information regarding OC sequestration rates within temperate inland wetland soils is crucial for evaluating the potential of these ecosystems to serve as natural climate solutions. Radiometric dating using ^{137}Cs and ^{210}Pb presents a valuable tool for estimating the recent OC sequestration potential of wetlands. Notably, a robust 1 : 1 linear correlation has been observed between ^{137}Cs- and ^{210}Pb-based OC sequestration rates in high-quality sediment profiles.

While estimations based on the onset of ^{137}Cs in 1954 or its peak in 1963 were reasonable, estimates anchored to the 1963 peak of ^{137}Cs exhibited closer alignment with those derived from ^{210}Pb data (using the CFCS model). These findings suggest that estimates derived from ^{137}Cs and ^{210}Pb radioisotope dating methods are interchangeable and reasonably comparable when utilizing the 1963 ^{137}Cs time marker.

Combining ^{137}Cs and ^{210}Pb tracers provides a comprehensive assessment of sedimentation rates. While one tracer offers an average sedimentation rate over 60 years, the other provides a temporal trend over the same period. This interchangeability enables more thorough evaluations of the average sedimentation rate in wetlands, which is crucial for leveraging them as natural climate solutions.

Code and data availability. The R code for the distance sampling modeling, along with the data to run the code, is available at https://doi.org/10.5281/zenodo.10951658 (Mistry et al., 2024). The organic carbon (OC) sequestration rate data used to check the comparability of the radioisotope profiles can be found in the Supplement. These sequestration rate data and the geographical locations, years of sampling, and additional information about the sediment cores are available at https://doi.org/10.5281/zenodo.13696300 (Creed et al., 2024). The radioisotope profiles used for screening are in the paper and Supplement. The paper and Supplement present other relevant data to support our conclusion.

Supplement. The supplement related to this article is available online at: https://doi.org/10.5194/bg-21-4699-2024-supplement.

Author contributions. The authors' contributions are as follows. PM: methodology, field and lab analysis, statistical analysis and modeling, and writing. IFC: conceptualization, methodology, field and lab analysis, editing, and supervision. CGT: conceptualization, editing, and supervision. EE: methodology, field and lab analysis, and editing. DAL: methodology, field and lab analysis, and editing.

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