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# How to Estimate Carbon Dioxide Emissions

## INTRODUCTION

Understanding how blue carbon ecosystems contribute to climate change mitigation requires an understanding of ecosystem extent (Chapter 2), the amount of carbon present in each pool (Chapters 3 and 4), and the how much carbon is emitted or sequestered over time (presented here). The amount of carbon emitted to the atmosphere or sequestered can be determined directly using methods that detect gas exchange, or more commonly using carbon stock change as a proxy for gas exchange. However, other greenhouse gases (GHGs) such as methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) can be measured only by using gas exchange methods. Assessing changes in blue carbon stocks and GHG emissions can be used to establish baselines for tracking sequestration rates and emission trends, identifying activities that increase emissions, assessing progress of restoration efforts, communicating with stakeholders, and more. Together, this information enhances countries' and institutions' ability to meet current and future reporting needs, participate in carbon-based markets, and inform mitigation policy and management practices at local to national scales.

## APPROACHES TO MEASURING EMISSIONS

To determining changes in carbon stocks (as a proxy for CO<sub>2</sub> emissions), two approaches can be used:

- 1) Stock-difference method: This method estimates the difference in carbon stocks measured at two points in time and results in IPCC Tier 3 estimates (Kauffman *et al.* 2014).
- 2) Gain-loss method: This method estimates the difference in carbon stocks based on emissions factors for specific activities (e.g., plantings, drainage, rewetting, deforestation) derived from the scientific literature and country activity data and results in Tier 1 and 2 estimates (GOFC-GOLD 2009; IPCC 2013).

In addition, a third method exists, for the direct measurement of GHG emissions:

- 3) Flux method: This method estimates the GHG flux between the soil and vegetation and the atmosphere/water column through direct measurements or by modeling results in Tier 2 and 3 estimates.

Often, a mix of methods is used, but ultimately the method selected will be based on the needs of the project, the resources available, and the level of accuracy required.

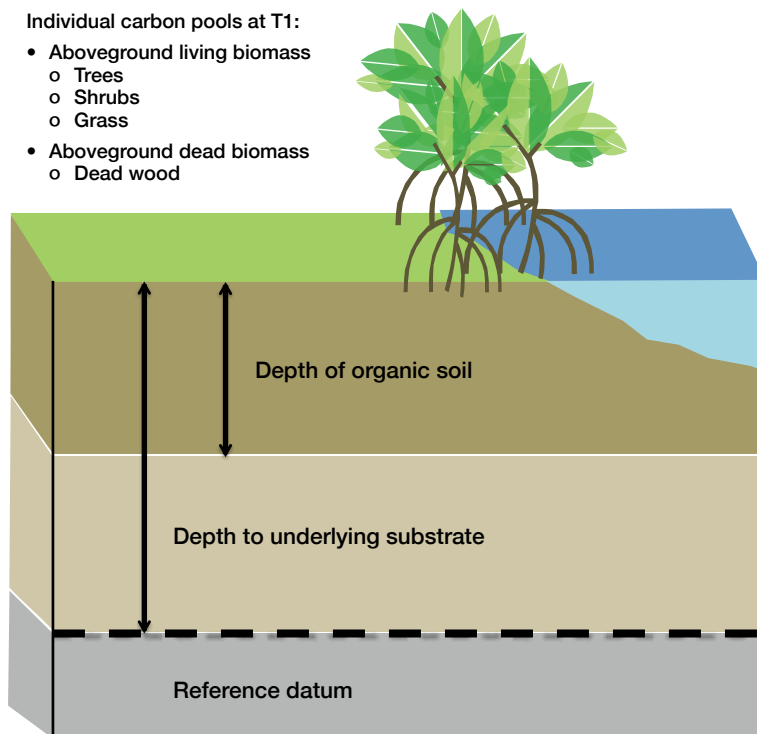
### Stock-difference Method

The stock-difference method requires carbon measurements for each significant carbon pool at least two points in time (T1 = initial assessment and T2 = subsequent assessment). Results from each inventory are then compared and analyzed for changes. This approach not only tracks changes to carbon stocks as a whole, but also changes in the individual carbon pools. For example, if the living biomass pool decreases but the dead biomass pool increases, it can be assumed that the ecosystem was degraded between the two time points. However, this technique alone cannot determine the cause of the degradation (natural or human caused).

## INITIAL ASSESSMENT (T1)

The most accurate method is to establish permanent plots at the time of the first inventory so that subsequent assessments are done at the exact same location, for the same size plot, with the same protocols. However, if establishing permanent plots is not feasible, future measurements can be made from a similar location with higher uncertainty. If not using permanent plots, it is important to ensure that the area of land being assessed at T1 and T2 are identical to avoid confounding stock change estimates with area changes (IPCC 2013).

Carbon pools are measured according to the methods laid out in Chapters 2–4. The initial assessment will be used as a reference point (datum) for future change. This is particularly important for soils in blue carbon ecosystems because much of the stock change occurs when the soil accretes (gains volume, such as through elevation gain) or erodes. Examples of possible reference points for a change in soil volume include depth to the organic soil base or depth to an underlying substrate (i.e., bedrock) (**Fig. 5.1**). A constant soil volume reference datum, such as the underlying substrate, is useful because any change could only have come from soil accretion or erosion since the substrate is constant (baseline scenario). Other reference points, such as carbon pool measurements, will be based on the results of the initial assessment (project scenario). A separate datum must be established for each parameter being tracked.



**Figure 5.1** Establishing a soil volume reference datum

## TIME BETWEEN ASSESSMENTS

The time recommended between measurements depends on the project site, intensity of activities, and the time needed to see a measurable change. If the site is pristine, and no land use change has been observed, the timing between measurements should relate to the time required to see natural changes in the carbon stocks. For example, seasonal changes

in aboveground biomass associated with loss of leaves in the cooler months is balanced out by regrowth in the spring, so the size of that biomass carbon pool is relatively stable. Thus, annual measurements are usually not needed; instead, measurements every 5–10 years might be more valuable. Similarly, soils accrete very slowly (rate of millimeters per year), and the carbon found in the deeper sediments (beneath the top meter) is relatively stable and not likely to change if the ecosystem is not altered. Therefore, adequate soil carbon measurements may also be taken at 5–10 years or longer (10–20 years). However, if the project area has undergone a land use change, or if land-use changes nearby are impacting the site, measurements should be taken more regularly to record any significant impacts.

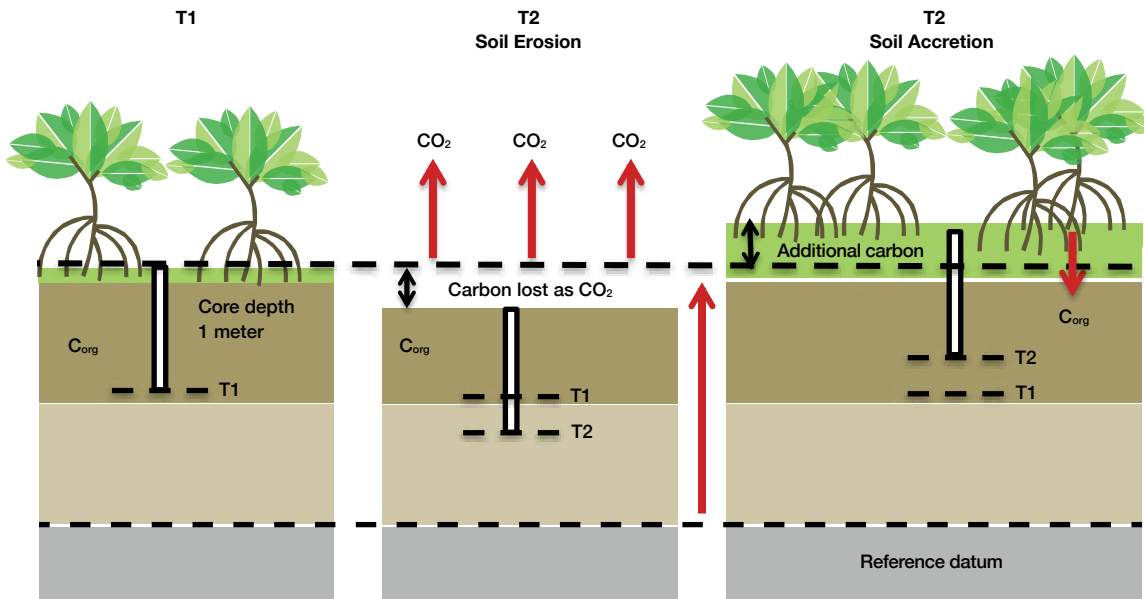
## SUBSEQUENT ASSESSMENTS (T2)

Return to the same location analyzed at T1, or an alternative location that is representative of the size and current species density of the original location, and repeat the blue carbon inventory. Vegetative carbon pools can be measured according to the steps outlined in Chapter 4. The carbon inventory from T1 can be used as a reference to determine changes in the vegetative carbon pools.

Measuring the soil carbon pool is more challenging due to soil accretion/erosion that impact soil volume or carbon density. For example, if significant soil accretion/erosion has occurred at the study site, a core taken from 0–100 cm below the soil surface at T1 will not cover the same absolute depth range as a core taken at 0–100 cm from the soil surface at T2 (**Fig 5.2**). In order to correct for this, the change in soil volume between T1 and T2 must be determined by establishing a reference datum as described above. The concept is that a reference datum is a horizontal line below the influence of plant activity (i.e., all carbon added by root growth or surface deposition occurs above the reference). One approach for establishing a reference datum is to locate a feature in the soil profile that can be clearly identified over time, such as bedrock or a sharp transition from organic-rich to organic-poor soil (**Fig 5.1**). For example, Kauffman *et al.* 2014 used the boundary between organic soils and marine sands as the reference datum in Honduras. Some commonly used methods for establishing a reference datum do not capture the full soil profile, leaving open the possibility that soil volume below the feature changed. For example, the position of the soil surface in 1963 can be established by locating a peak in cesium-137. Soil Elevation Tables (SETs) are an expensive but accurate method for assessing change soil volume change across the full soil profile in cases where there is no clear boundary.

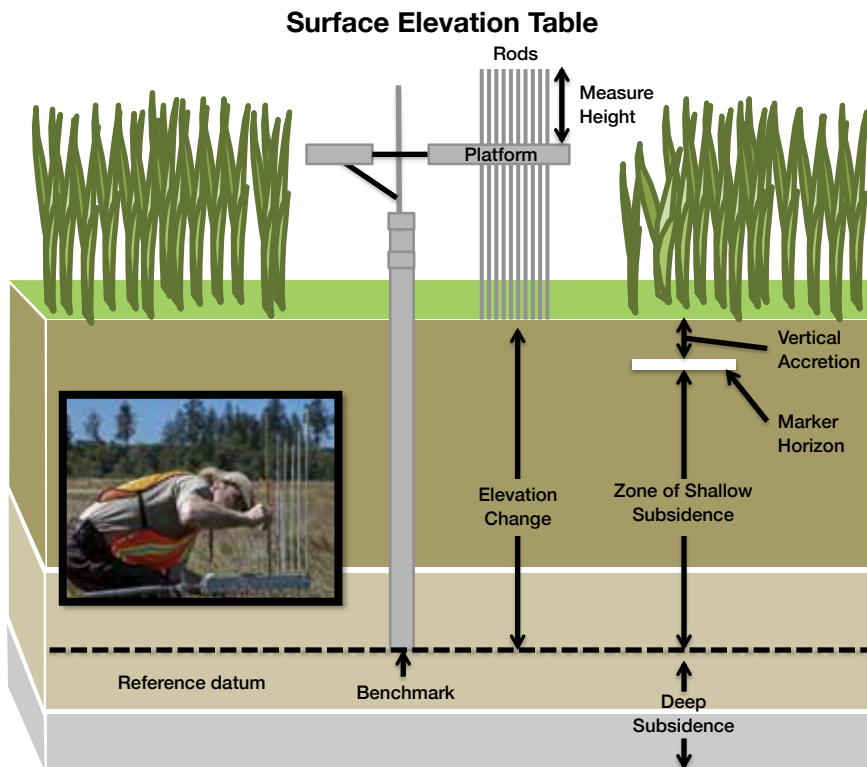
To obtain high-resolution measures of wetland soil elevation, a portable, mechanical device called a Surface Elevation Table (SET) can be used (**Fig 5.3**). The SET device has an accuracy of 1.5 mm and can be used to determine elevation trends in any wetland setting (Cahoon *et al.* 2002). It is more accurate to measure soil elevation seasonally every year between T1 and T2 to establish trends and determine an average gain/loss per year. However, if it is not possible to routinely make measurements, soil elevation can be determined from a different but similar location where routine sampling is easier, but the uncertainty will increase. Rates of soil elevation gains/losses over time may also be available from government agencies or researchers.

A detailed explanation of the SET marker horizon (MH) methodology, a list of the scientists using it, and the locations where it is being used can be found on the US Geological Survey (USGS) web site: [www.pwrc.usgs.gov/set](http://www.pwrc.usgs.gov/set). Briefly, the SET-MH methodology comprises a



**Figure 5.2** Effect of accretion and erosion on soil samples. A reference datum is usually given the designation “0,” and future measurements should refer to it. For example, soil accretion of 3 cm at T2 would be reported as “+3 cm relative to the datum” where the initial datum would also be reported. Note that the “0” reference datum in this example is established by measuring the distance to a second, primary reference datum below the soil surface. The primary reference may be bedrock or a distinct soil layer such as clay (**Fig. 5.1**), or the bottom of an SET benchmark (**Fig. 5.3**).

platform that attaches to a benchmark (pole driven permanently into the ground that does not move and remains from T1 to T2). The platform is suspended above the ground, and rods of known length are fed through openings in the platform and rested on the surface of the soil (**Fig 5.3**). The height from the top of the platform to the top of the rod is measured. The



**Figure 5.3** Diagram of a Surface Elevation Table (SET) marker horizon (MH) method used to detect changes in soil surface elevation. Inset image: USFWS biologist Bill Bridgeland taking measurements on a SET (© Roy Lowe, U.S. Fish and Wildlife Service).

height of the rod changes as the soil surface changes, representing change in soil surface elevation, which corresponds to a change in soil volume. Change in soil volume is the only parameter required for calculating stock change. However, the associated marker horizon can give additional insights that are interesting for research purposes, namely a calculation of shallow subsidence (e.g., due to root decomposition) (Cahoon *et al.* 2002).

#### EXAMPLE

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- Rod height at T1 = 100.46 cm
- Rod height at T2 = 100.98 cm
  - Elevation change = 100.98 cm – 100.46 cm = 0.52 cm = 5.2 mm (common unit used for describing elevation changes in soil and sea level)
- Depth of the marker horizon at T1 = 0
  - The marker horizon is established during the first assessment
- Depth of the marker horizon at T2 = 8.6 mm
  - Vertical accretion = 8.6 mm – 0 mm = 8.6 mm
- Shallow subsidence = Vertical accretion – elevation change
  - 8.6 mm – 5.2 mm = 3.4 mm

In the case of soil accretion, the top sub-sample is used to determine the change in the soil carbon due to change in the soil volume (Lovelock *et al.* 2014).

#### EXAMPLE

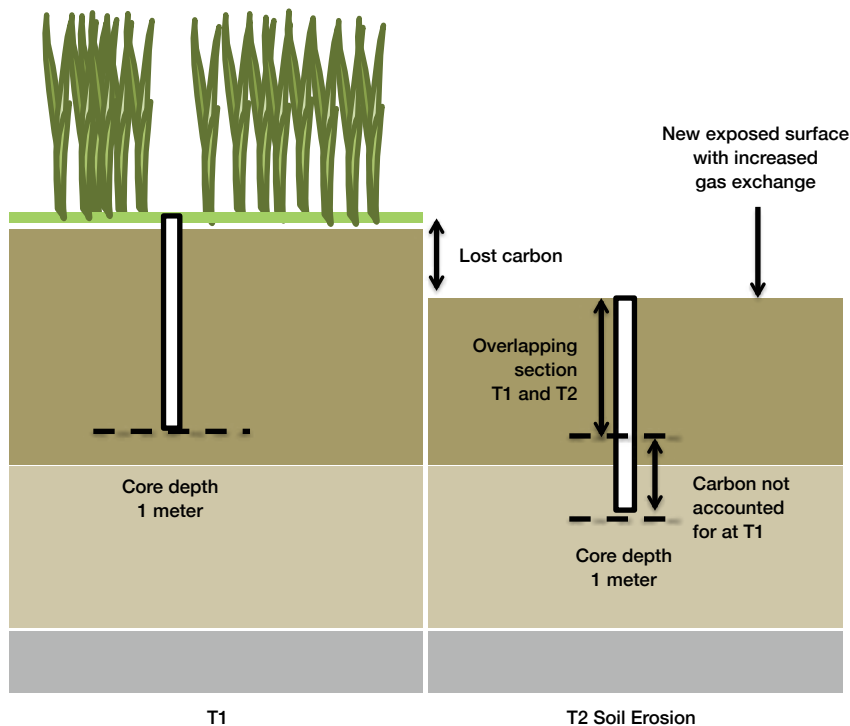
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- T2 soil core top 10 cm = 0.195 g/cm<sup>3</sup>
- Soil elevation rate (determined by SET-MH measurements) = 0.52 cm per year
- Time between T1 and T2 = 10 years
- Total soil accretion occurring in that time = 5.2 cm
- 5.2 cm \* 0.195 g/cm<sup>3</sup> = 1.014 g/cm<sup>2</sup> additional soil carbon due to accretion (in a single core)

This number can then be extrapolated to the hectare scale:

- Total carbon content in the core due to accretion = 1.014 g/cm<sup>2</sup>
- 1 Mg = 1 000 000 g
- 1 hectare = 100 000 000 cm<sup>2</sup>
- 1.014 g/cm<sup>2</sup> \* (Mg/1 000 000 g)\*(100 000 000 cm<sup>2</sup>/ha) = 101.4 Mg C/ha
- **101.4 Mg C/ha was added to the system**

If the area has undergone erosion, a one meter soil core at T2 will go deeper than the sample taken at T1, and the new surface that was once buried is now exposed to the air/water. In this case, a change in the soil carbon at T2 is determined by doing an assessment of carbon content for the overlapping depth (**Fig. 5.4**).



**Figure 5.4** Diagram of how carbon losses may be accounted for where the soil surface has subsided or been eroded.

### EXAMPLE

- Soil elevation rate (determined by SET measurements) = -0.86 cm per year
- Time between T1 and T2 = 10 years
- Total soil erosion occurring in that time = -8.6 cm
  - So the one-meter core at T2 went 8.6 cm deeper than a one-meter core at T1, and, thus, the bottom 8.6 cm cannot be used to determine change
- The carbon content of the remaining top 91.4 cm (as determined by methods in outlined in Chapter 3) = 69.2 g/cm<sup>2</sup>

This number can then be extrapolated to the hectare scale:

- Total carbon content in the top 91.4 cm of the core = 69.2 g/cm<sup>2</sup>
- 1 Mg = 1 000 000 g
- 1 hectare = 100 000 000 cm<sup>2</sup>
- 69.2 g/cm<sup>2</sup> \* (Mg/1 000 000 g)\*(100 000 000 cm<sup>2</sup>/ha) = 6920 Mg C/ha
- T1 soil carbon pool = 7205 Mg C/ha
- T2 soil carbon pool – T1 soil carbon pool = change in soil carbon
  - 6920 Mg C/ha – 7205 Mg C/ha = -285 Mg C/ha
- **285 Mg C/ha was lost from the system**

## DETERMINING CHANGES IN CARBON STOCKS/POOLS

The change in carbon stocks for a given blue carbon ecosystem over a specific area can be determined as follows:

- Change in carbon stock (Mg C) = total carbon stock at T2 (sum of all carbon pools) – total carbon stock at initial measurement T1 (sum of all carbon pools)
- For example in a degraded salt marsh:
  - Measurements taken from a salt marsh site in 2002 (T1) estimated the total carbon stock (sum of all relevant pools) to be 34 667 Mg C.
  - Measurements taken for the same salt marsh site using the same methods in 2012 (T2) estimated the total carbon stock (sum of all relevant pools) to be 25 167 Mg C.
  - $25\,133\text{ Mg C} - 34\,667\text{ Mg C} = -9534\text{ Mg C}$  (carbon lost from the ecosystem)

If we assume all carbon lost is emitted to the atmosphere/water column (see Additional Considerations for Carbon Accounting below) then the total change in carbon can be used to estimate the annual rate of change and potential CO<sub>2</sub> emissions.

- Annual change in carbon stock (Mg C/year) = (total carbon stock at T2 – total carbon stock at initial measurement T1) / (T2 – T1)
- Using the example above
  - $(25\,133\text{ Mg C} - 34\,667\text{ Mg C}) / (2012 - 2002) = -953\text{ Mg C/year}$ .
  - Associated CO<sub>2</sub> emissions =  $953\text{ Mg C} * 3.67$  (conversion factor) =  $3498\text{ Mg CO}_2$

The stock-change approach is used to make Tier 3 estimates of carbon stock/pool change. It is applicable in countries that have national inventory systems, or ongoing monitoring programs, where the stocks of different biomass pools are measured at periodic intervals (IPCC 2013). However, the stock-change approach requires measurements of each carbon pool and significant resources, making this approach not feasible in all situations.

### Gain-loss Method

This method is used following an initial blue carbon inventory when subsequent inventories from direct measurements are not feasible. Instead, the gain-loss method uses activity data to account for the change in carbon stock between T1 and T2. There are numerous activities that result in carbon stock gains (transfer between pools, restoration activities, annual plant growth, soil accretion, etc.) and losses (transfer between pools, land-use changes, extractive activities, natural disturbances such as storms, etc.) in coastal marine ecosystems, the impacts of various activities are complex and as yet not well characterized. Thus, this method yields Tier 1 and 2 results. Tier 1 results are calculated based on the general conversion factors obtained from globally compiled databases (i.e., 2013 IPCC Wetlands Supplement), Tier 2 results are based on country-specific conversion factors. Both Tier 1 and Tier 2 estimates require country-specific or project-specific activity data.

If we assume all carbon lost is emitted to the atmosphere/water column (see Additional Considerations for Carbon Accounting below), using the Gain-Loss method change in carbon stocks for a given blue carbon ecosystem over a specific area can be determined as follows:

- Change in carbon stock (Mg C) = Carbon stock at T1 – (carbon losses (land use change, natural disasters, erosion, etc.) + carbon gains (soil accretion, growth, restoration, etc.))



## FOR EXAMPLE

- Measurements taken from a 1000 ha salt marsh site in 2002 (T1) estimated the total carbon stock to be 34 667 Mg C.
- In 2007: 200 ha were drained resulting in an emissions rate = 7.9 Mg C/ha/yr (IPCC 2013, **Table 4.13**, p. 31)
- In 2010: 50 ha were rewetted resulting in an emissions rate = -0.91 Mg C/ha/yr (IPCC 2013, **Table 4.12**, p. 29)
- Second assessment conducted in 2012 (T2)
- Assuming all else remained the same carbon stock change can be estimated as follows:
  - $(200 \text{ ha} * 7.9 \text{ Mg C/ha/yr}) * (2010-2007) = 4740 \text{ Mg C}$
  - $(150 \text{ ha} * 7.9 \text{ Mg C/ha/yr}) * (2012-2010) = 2370 \text{ Mg C}$
  - $(50 \text{ ha} * -0.91 \text{ Mg C/ha/yr}) * (2012-2010) = -91 \text{ Mg C}$
- Total carbon difference = 4740 Mg C + 2370 Mg C + -91 Mg C = 7019 Mg C lost
- Associated CO<sub>2</sub> emissions = 7019 Mg C \* 3.67 (conversion factor) = 25 739 Mg CO<sub>2</sub>

## Additional Considerations for Carbon Accounting

When calculating carbon emissions, it is important to consider where any additional ecosystem carbon originated from and where any loss of carbon ends up. The calculations above apply when the difference in carbon between T2 and T1 is released to the atmosphere/water column as CO<sub>2</sub>, providing an upper estimate of carbon emissions. However, in some cases carbon is not emitted but redeposited to adjacent habitats or the deep ocean. Corrections must be made to include carbon that is lost but not emitted. However the science to support choosing the correction factor describing the proportion of carbon emitted to the atmosphere is still developing.

Autochthonous organic carbon (carbon originating or forming in the place where it is accumulated) can be included in carbon accounting. Allochthonous organic carbon (carbon originating outside the project boundary and being deposited in the project area) can be challenging to account for at the project level because the carbon was lost from an adjacent location but not emitted. For example, in tidal wetlands erosion may reduce carbon stocks at one location, but some of this carbon may be deposited and trapped in a neighboring system. Indeed, up to 50% of seagrass soil carbon is allochthonous (Kennedy *et al.* 2010). In this case, it is incorrect to presume that the carbon lost at the first site resulted in carbon emissions.

In the case of live vegetation, any carbon additions are due to increased plant biomass from growth (autochthonous) and any loss is from degradation (e.g., deforestation or weather event); therefore, this carbon is transferred to the non-living vegetative carbon pool. Carbon in the non-living vegetative pool either gets released to the atmosphere/water column, buried in the soil as the plant material decomposes, or is removed to a neighboring ecosystem by wind/currents. Carbon lost to the atmosphere/water column can be counted as emissions, carbon buried in the soil will be accounted for during the soil analysis, and carbon that is moved to a new location can either be ignored (if < 5%) or estimated based on conservative conversion factors or modeling. Similarly, deposited allochthonous vegetative carbon (i.e., leaf litter, branches) can either be ignored (if < 5%), estimated based on conservative conversion

factors or modeling, or—if it is obvious (i.e., tree branches in low marsh area)—can be calculated as a separate carbon pool.

In the case of soil, organic carbon additions result from either increased belowground biomass or sedimentation. Where soil organic carbon accumulates on the project site due to sedimentation, a compensation factor must be applied. This factor is based on a percentage of the carbon stock that is derived from allochthonous soil organic carbon and may be derived from published values, field-collected data (e.g., stable isotopes of carbon can be used in seagrass beds to establish the origin of accreted carbon (Kennedy *et al.* 2010), proposed conservative default factors (e.g., 50% allochthonous carbon) or modeling. When soil carbon is lost due to erosion, the fate of the lost carbon (e.g., redeposited or oxidized to CO<sub>2</sub>) determines if this loss can be accounted as CO<sub>2</sub> emission. Therefore, a compensation factor is needed here, too. The science on this subject is not sufficiently developed to propose a default factor, and, therefore, must be based on either field data or modeling. Studies to date have used a range e.g., 25–100% of organic carbon in the top meter of the soil as being emitted to the atmosphere (Fourqurean *et al.* 2012a; Pendleton *et al.* 2012; Siikamäki *et al.* 2012).

## Gas Flux Techniques for Directly Assessing Carbon Flux

To understand how coastal wetlands influence climate, there is a need to understand how much GHG actually leaves coastal ecosystems and enter the atmosphere. While the methods described above can be used as a proxy to estimate the carbon emissions from coastal ecosystems, flux measurements are the only way to measure the emission of methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O), which do not accumulate in the system. Flux measurements can also be used to measure CO<sub>2</sub> emissions directly, as an alternative to the stock change methods described above.

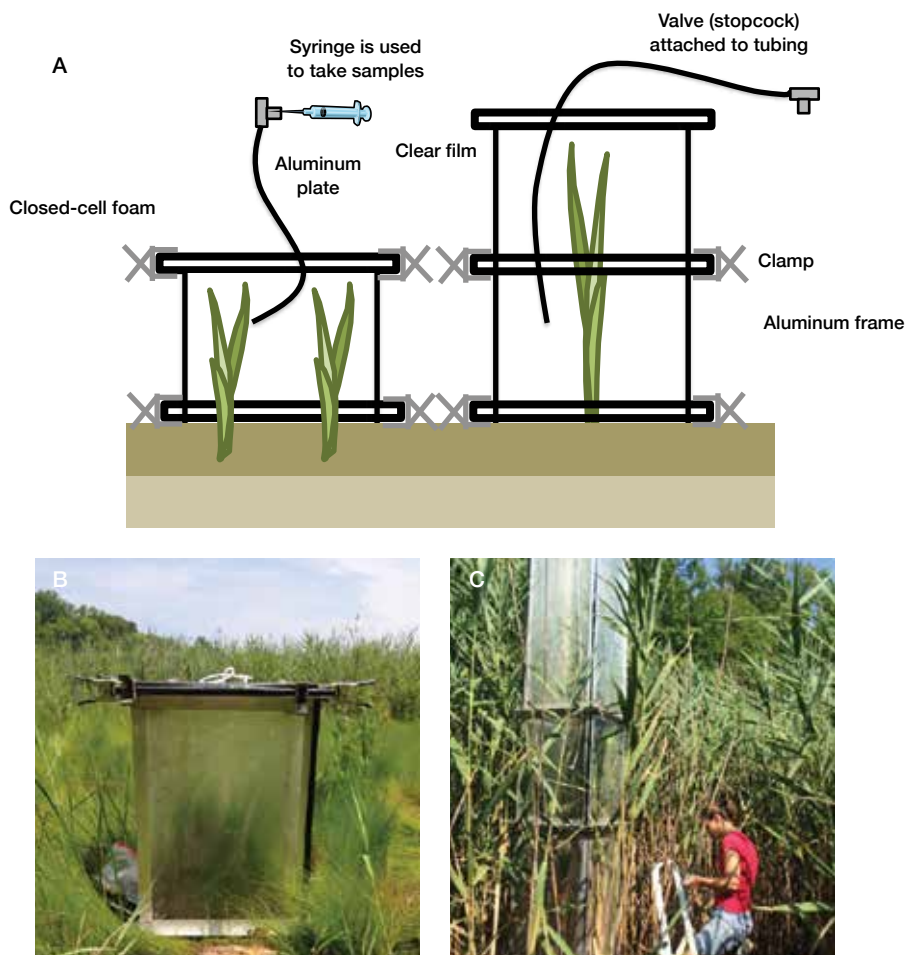
In general, management activities resulting in extraction of soils, such as construction of aquaculture ponds, can result in large CO<sub>2</sub> emissions in mangroves and tidal salt marshes (IPCC 2013), N<sub>2</sub>O emissions originate mostly from aquaculture activities and upland agricultural practices, and rewetting areas increases CH<sub>4</sub> from drained freshwater tidal systems. The mass of CH<sub>4</sub> and N<sub>2</sub>O exchanged between wetlands and the atmosphere is smaller than the exchange of CO<sub>2</sub> by orders of magnitude; however, they are far more effective greenhouse gases than CO<sub>2</sub>, with global warming potentials 25 and 298 times higher than CO<sub>2</sub> over a 100-year time horizon. For this reason, relatively low emissions of these gases have the potential to significantly change calculations on the climate benefits of conserving, creating, or restoring wetlands.

N<sub>2</sub>O emissions are generally negligible unless the ecosystem has a source of nitrate loading, such as from fertilizer runoff. CH<sub>4</sub> production is directly related to salinity (Purvaja & Ramesh 2001; Poffenbarger *et al.* 2011). Systems with salinity levels greater than 18 ppt can assume a zero value for CH<sub>4</sub> emissions. For the purposes of monitoring emissions from blue carbon ecosystems, N<sub>2</sub>O emissions will most likely be minimal to nonexistent in pristine areas but may be significant in degraded areas. Methods below use CH<sub>4</sub> as the example, but the principles of measurement are the same for all three gases.

## Static Chamber Method

The most common technique for measuring CH<sub>4</sub> (or any GHG) flux is to trap the gases inside a closed chamber that is open only to the soil and plants rooted therein, then measure

the rate at which gas concentrations change over time (Meronigal & Schlesinger 2002; Lovelock *et al.* 2011; Sidik & Lovelock 2013). Flux measurements require the construction or purchase of static chambers and boardwalks to avoid disturbing the site where fluxes will be measured. Static chambers are relatively inexpensive to construct, can be designed to function with a wide range of plant types and site conditions, and can be deployed in groups to save field time (**Fig. 5.5**). However, static chambers may also change the temperature and light environments of plants, they do not capture  $\text{CH}_4$  escaping from the soil as bubbles (as opposed to diffusion), and assumptions are needed in order to scale short-term emissions data to annual fluxes.



**Figure 5.5** Chamber volume must be adjusted to enclose plants of different stature (B & C). (B, © Rachel Hagar, SERC; C, Eric Hazelton, SERC)

The basic design of the chamber requires a base that extends into the soil or the floodwater and a chamber that can be placed over the plants and sealed to the base. Once the chamber is in place, the headspace concentration of gases being emitted or consumed by the plant-soil system will begin to change. These changes can be quantified by taking samples from the headspace. The sensitivity of the method is determined largely by the ratio of the chamber headspace to the area of soil from which  $\text{CH}_4$  is emitted.  $\text{CH}_4$  emissions are often quite low from tidal wetlands (compared to  $\text{CO}_2$ ), so it may be necessary to enclose fairly large areas (minimum  $0.25 \text{ m}^2$ ). The measurement period can be lengthened to increase sensitivity, but longer time periods increase the risk of the chamber heating up in the sun and impacting the results.

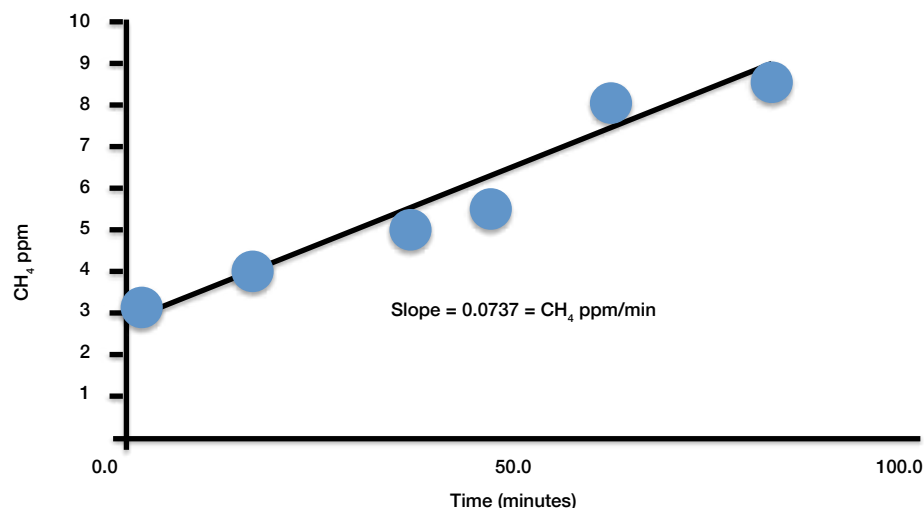
The chambers must be constructed of gas-impermeable material. The side of the chambers can be transparent or opaque depending on the vegetation. Transparent chambers are used when enclosing the foliage of plants with pressurized gas transport systems (e.g., *Phragmites*), but are not necessary for plants that have diffusion transport (e.g., *Spartina*). Opaque chambers are easier to use because they do not require temperature regulation and can be made easily from common items like plastic trashcans. If long incubation times are needed because of low  $\text{CH}_4$  flux rates, it may be necessary to cool the air inside transparent chambers in order to maintain temperature within about 2 °C of ambient.

Additional issues to consider include the time of day measurements are taken and approaching the chamber properly. If  $\text{CH}_4$  emissions are affected by light, then both light and dark emissions should be measured in order to estimate a 24-hour rate. It may be easier to measure emissions when the site is not flooded; a few studies report that flooding does not affect  $\text{CH}_4$  emissions, but there is at least one exception (Van der Nat & Middelburg 2000). Care must be taken not to step or stand on the soil surface within 1–2 m of the chamber; the pressure may cause  $\text{CH}_4$  bubbles trapped in the soil to escape and artificially increase emission rates. For this reason it is useful to build a simple boardwalk elevated 5–10 cm off the soil surface for approaching the chamber, and to install the chamber base several days before taking the first samples. The headspace  $\text{CH}_4$  concentration is typically quantified by drawing a headspace sample into a gas-tight syringe, injecting it into a leaf-free vial for storage (e.g., Exetainer vial), and analyzing against standards using gas chromatography.

Water motion affects seagrass photosynthesis, gas exchange, sediment transport, and epiphyte load. Therefore, estimates of flux that constrain or alter water motion will confound the estimate of flux. Benthic chambers used in seagrass ecosystems rely on the temporal measurement of water properties, but do not replicate the physical conditions of the aquatic environment (flowing water).

## CALCULATIONS

The rate of gas flux is determined by measuring changes in headspace concentration over time as determined by the slope of a best-fit line through the data (**Fig. 5.6**). To do this, samples are taken at several consecutive points in time (e.g., 2 min, 15 min, 35 min, 45 min,



**Figure 5.6** Proportion of  $\text{CH}_4$  gas in the chamber is determined by plotting gas concentration per samples against the time between closing the chamber and collecting the sample. The slope is determined by calculating a best-fit line.

60 min, and 80 min) and the amount of CH<sub>4</sub> is determined for each sample. The slope is converted from concentration to a mass-based flux of CH<sub>4</sub> by accounting for the total mass of all gases in the chamber headspace. Mass flux can then be divided by the area of soil inside the chamber in order to express emissions on an area basis.

Measurements needed to determine CH<sub>4</sub> flux (**Table 5.1**):

- Atmospheric Pressure (assume this value = 1 atmosphere)
- Volume of the chamber (Liters)
- Universal gas constant = 0.0820 (L\*atmos/K\*mol)
- Temperature in Kelvins at the time of each measurement (K = 273 + temp in °C)

**Table 5.1** Example of how to determine moles of gas molecules within a flux chamber

PRESSURE (atmos)	VOLUME OF CHAMBER (L)	GAS CONSTANT (L*atmos/K*mol)	TEMPERATURE (Kelvins)	GAS MOLECULES IN THE CHAMBER (moles)
P	V	R	T	$n = PV/RT$
1	515	0.0820	288	21.8072

Run each sample through a gas chromatograph or similar instrument to determine the proportion of the total gas in the chamber that is CH<sub>4</sub>; use the results of each sample to calculate the slope of the data (**Fig. 5.6**).

### CALCULATE FLUX

- n (total amount of gas in the chamber) is in moles, but the CH<sub>4</sub> is measured is in parts per million (ppm) so these need to be converted to similar units
  - Amount of gas molecules in the chamber = 21.8072 moles
  - Amount of CH<sub>4</sub> = 0.0737 ppm/minute
  - 1 ppm = 1 μmole CH<sub>4</sub> per mole of gas
  - 0.0737 ppm = 0.0737 μmole/mole/minute
  - So for every mole of gas, 0.0737 μmole is CH<sub>4</sub>
    - Thus, the total amount of CH<sub>4</sub> being emitted in the chamber is:
    - (0.0737 μmole/mole/minute) \* 21.8072 moles = 1.6072 μmole/minute
- To determine the amount of CH<sub>4</sub> emitted per minute per unit area, divide by the area of the chamber (assumed to be 0.5 m<sup>2</sup> in this example)
  - (1.6072 μmole/minute)/0.5 m<sup>2</sup> = 3.2 μmole/m<sup>2</sup>/min
- Extrapolate to Mg/ha/day
  - 1440 min = 1 day
  - 1 ha = 10 000 m<sup>2</sup>
  - 1 mole = 1 000 000 μmole
  - 1 mole of methane = 16.042 g (according to molecular weight of methane)
  - 1 Mg = 1 000 000 g
  - 3.2 μmole/m<sup>2</sup>/min \* 10 000 m<sup>2</sup>/ha \* 1 mole/1 000 000 μmole \* 16.042 g/mole \* 1Mg/1 000 000 g = 51\*10<sup>-8</sup> Mg CH<sub>4</sub>/ha/min
  - 51\*10<sup>-8</sup> Mg CH<sub>4</sub>/ha/min \* 1440 min/day = 7.4\*10<sup>-4</sup> Mg/ha/day

- If the project area was 1500 ha then the total CH<sub>4</sub> emissions for the project area is
  - 1500 ha \* (7.4\*10<sup>-4</sup> Mg/ha/day) = **1.11 Mg CH<sub>4</sub>/ha/day**

When extrapolating these measurements to an annual scale (Mg/ha/year), assumptions must be made about the level of variation of emissions over time, or variation over time must be measured and emissions scaled up using modelling techniques. Confidence decreases when scaling up to longer time scales as emissions may vary over seasons and with short- and long-term variation in temperature, salinity, and other factors (Poffenbarger et al. 2011), among other factors.

## Eddy Covariance Method

The eddy covariance (EC) method is an alternative, more sophisticated, method for directly measuring the exchange of CO<sub>2</sub> between ecosystems and the atmosphere. It is non-intrusive, *in situ*, and measures fluxes of the entire ecosystem. However, EC is expensive because it requires buying flux towers and paying personnel to perform complex data processing. For more information on how to set up an EC system, please see Aubinet *et al.* (2012) and Burba (2013).

Mangroves and tidal salt marshes can use standard terrestrial EC towers (Barr *et al.* 2010). However, these systems were originally designed for terrestrial forests, and the close proximity of large water bodies will impact the readings and require an additional layer of data processing to correct. Underwater EC systems used for seagrasses currently only measure O<sub>2</sub> because CO<sub>2</sub> sensors are not fast enough to measure CO<sub>2</sub> in turbulent water (Chipman *et al.* 2012).

When conducting flux measurements it is important to consider the form of carbon being lost. Carbon may be lost from coastal wetlands in the form of dissolved inorganic carbon, organic carbon, or particulate organic compounds that move into adjacent ecosystems (mangrove-seagrass-coastal ocean). Typically, the loss of dissolved organic compounds is only a small fraction of the of the carbon budget (Wetzel & Penhale 1979), but losses of DIC from mangroves may be large (Bouillon *et al.* 2008). Horizontal fluxes of carbon are generally missed by atmospheric measurements of flux, which supports why carbon flux between the atmosphere and the soil surface may not be equivalent to changes in the coastal wetland storage of carbon. Assessment of the magnitude of different pathways of carbon fluxes from mangroves, salt marsh and seagrass ecosystems is an active area of research that will reduce uncertainty around the carbon budgets of these ecosystems.