Deltares

Greenhouse gas emissions from the clay ripening pilot "Kleirijperij"

EEMS DOLLARD 2050

EcoShape

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0.1	Martine Kox Stefan Jansen	Wouter van der Star	Rob Nieuwenhuis	

Summary

In dredging projects not only the combustion of fossil fuels by machines can be a source of greenhouse gases, but also the emission from naturally occurring organic material in the sediment can be a potential source. Greenhouse gases (GHGs) can be produced within sediment as a result of microbial degradation (oxidation and reduction) of the organic matter. Under certain conditions these GHGs can escape to the atmosphere. Little is known about how much greenhouse gas is released from sediments as a result from dredging operations or beneficial sediment use application. One of these potential beneficial uses is related to ripening (salty) dredged material to produce clay for dike strengthening.

Within EcoShape framework, Deltares has carried out a research project to determine the emissions of greenhouse gases during the maturation of the organic-rich dredged material from the Eems-Dollard estuary in the clay ripening project “Kleirijperij de Kwelder”. The specific goals of this study were to:

- Determine emissions of greenhouse gases (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) during maturation of dredged material in the Kleirijperij de Kwelder sedimentation basins; and to
- Evaluate factors that influence these GHG emissions.

After filling the ripening plots with sediment from December 2019 to March 2020, GHG emissions were measured in April and June 2020. Over the course of these two months GHG emission (CO_2 and CH_4) from ripening salty estuarine sediment was very low: CO_2 and CH_4 concentrations in gas chambers at the sediment were insignificant compared to control chambers at the adjacent salt marsh. One of the reasons for this was probably the high concentration of sulfate in the sediment, which can lead to the inhibition of methane production. Another factor that might explain the low GHG emissions observed is slow oxygen transport into and slow GHG transport out of the ripening clay. Increase in salt and sulfate concentration upon drying and limitation of transport through the clay will further limit GHG emissions upon further ripening. The role of salt in limiting GHG emissions from sediment was supported by laboratory experiments of CH_4 emissions from sediment in closed vessels: freshwater samples showed significant CH_4 emissions, whereas estuarine samples did not.

The decrease in organic matter (OM) content of the sediment was used to estimate cumulative GHG emissions over longer timescales (months). The calculated emissions were significant, but the alternative of using clay from Belgium was calculated to generate more GHGs from transport alone.

We recommend expanding the GHG emission study to a range of sediments that are typically dredged and deposited on land. This will help to bring perspective in maturation of dredged sediment and accompanied emissions. A range of aquatic sediments varying in salt concentrations (freshwater, brackish, salt) would allow to study the effect of salt and sulfate. A range of organic matter (OM) content and age of this OM would allow to study whether OM maturation occurs once the sediment is deposited on land, and how this affects GHG emissions.

In this study we only focused on emissions of the ripening sediment. However, there are many process steps prior to the ripening period of the sediment that were not taken into account. It would be good to study GHG emissions all the way from pre-dredging up to the moment the ripened clay has been implemented in the dikes. The pre-dredging scenario gives an indication of a reference GHG emission to compare the other emissions too. Pre-dredging or natural emission are particularly important to provide the correct perspective on how significant the effective contribution of dredging, or more generally, of human intervention is with respect to the naturally occurring GHG emission. Ultimately, this data should be incorporated into a tool that is able to calculate (predict) C-emissions, C-capture and C-storage for these types of dredging activities.

Using the results from this study, Wetlands International, Boone Kauffman from Oregon State University, U.S and Deltares jointly made a calculation method for the GHG emission from the ripening process in the Kleirijperij, which was published in Terra and Aqua (see Appendix C). The calculation method presented offers a practical tool to estimate GHG emissions based on relatively simple input parameters (most notably organic matter content). This also offers a way to estimate effects of dredging or ripening conditions (e.g. freshwater vs. salt, availability of oxygen) and comparison to alternatives (e.g. alternative sources of sediment). It should be noted that the calculation method merely provides a start: many uncertainties still must be addressed, and the calculations have to be further supported by measurements.

Contents

1 Introduction

1.1 Background

1.1.1 The Kleirijperij pilots

The port of Delfzijl must be regularly dredged because of high sedimentation, which would otherwise hamper passage of vessels. The Eems-Dollard estuary (to which the port of Delfzijl is connected) has a high sediment content, which has a negative effect on the ecological value (Ecologie en Economie in Balans, 2012). Under normal dredging operations, the dredged material is transported and disposed back in the estuary. The “Kleirijperij” pilot (Dutch for “Clay Ripening”) came into existence within the program “Beneficial Use of Dredged Material". The aim of this program is to study how 2.5 million tonnes of (*in situ*) dredged material per year can be structurally extracted from the system, and thereby improve the ecological value of the Eems-Dollard estuary. At the Kleirijperij, the material dredged from the Eems-Dollard is converted into clay soil suitable for use in embankments. This clay is then used in another pilot, the “Wide Green Dike” (“Brede Groene Dijk” in Dutch) and applied to transform about one kilometre of the Dollard dike into a wider and greener dike: an embankment with a slope which is less steep than in normal embankments, and which is fully grass covered. Multiple parties are collaborating to make the Kleirijperij a success: EcoShape[1],the Province of Groningen, Rijkswaterstaat, Groninger Landschap, Groningen Seaports and the Water Authority Hunze and Aa's.

In the Kleirijperij pilots, the dredged sediment is deposited in 1 ha plots, where it is allowed to mature (ripen), after which it can be applied on the dike. To be suitable for application on dikes, the clay should fulfil the standard guidelines for application of clay in dike (TAW, 1996). One of the requirements of these guidelines is that the organic matter content should be <5%. However, dredged material can have a very high organic matter content (> 25%). During ripening of the dredged material, the percentage of organic material decreases, mainly due to oxidation. Furthermore, the strength and stiffness of dredged material increases when allowed to ripen. Ripening or maturation of the dredged material is a process in which the largely anaerobic dredged material turns into a more compact, more aerated and more permeable material by drying and (microbial) oxidation. As a result, the dredged material gradually changes from wet slurry into a more solid (clay) soil.
The Kleirijperij pilots were conducted at two locations: one near Delfzijl (Oosterhorn) (‘Kleirijperij Delfzijl’), were sediment was placed in 2018, and one along the Dollard Dike in Polder Breebaart (‘Kleirijperij De Kwelder’), where sediment was placed between 2019 and 2020.

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[1] Within EcoShape, the project is carried out by Arcadis, Boskalis, Deltares, HKV, RHDHV, Van Oord, Wageningen Marine Research and Witteveen+Bos

1.1.2 Greenhouse gas (GHG) emissions from sediment: existing knowledge

Within dredging projects not only the combustion of fossil fuels by machines can be a source of greenhouse gases, but also the naturally occurring organic material in the sediment can be a relevant source. Greenhouse gases (GHGs) can be produced from the dredged material as a result of microbial degradation (oxidation to CO_2 and/or reduction to methane) of the organic material in the dredged material. Limited knowledge is present on how much greenhouse gas is released in the process of ripening of dredged material, dredged material movement and natural background emissions in general. This is an important gap to quantify the overall impact of dredging activities to GHG emission, beyond machineries, and in relation to emissions from natural systems. Natural systems are indeed important contributors to the GHG cycle with significant emissions (e.g. from oxidizing mud) and sequestration (e.g. in salt marsh or mangroves). Quantifying GHG emissions from sediment, machines and natural background is therefore important information to quantify and reach international climate mitigation targets.

The quantity of organic carbon within a sediment will give an idea of the potential for GHG production. Silt fraction rather than the sand fraction is an important determinant for the GHG production potential, as much as nutrients, specifically nitrogen (ammonium, nitrate(Gebert et al2019). Within shallow lakes nitrogen and phosphorus also stimulate GHG production, as was shown in the BlueCAN project (Schep et al. 2020).
Not only the quantity of organic matter determines GHG emissions, but also the quality of organic carbon, or the degradable share of organic carbon, is relevant. Typically, older material sediment contains organic carbon that is already more degraded than young, fresh sediment For dredged sediment in a landfill, sediment depth was found to correlate to sediment age, with deeper layers being older, and also there the same relationship between age of organic carbon and degradability was found (Wijdeveld, 2004; Zander et al., 2020).
Furthermore, some environmental conditions are known to limit production of GHGs. High salt and sulfate content (like in marine sediments), limit CH_4 production. Therefore, many freshwater sediments tend to have higher potential emission of the strong GHG CH_4. High oxygen availability prevents formation of CH_4, but allows for oxidation of organic carbon to CO_2, causing high CO_2 fluxes. Last but not least. high iron (hydr)oxide availability is able to bind CO_2 as bicarbonate and thereby lower the CO_2 emission.
To estimate the amount of GHG released for marine and freshwater sediments, degradation tests can be performed in the lab or can be estimated using models. Several models have been been made using sediment age, quantity of organic carbon present and degradability constants as important determinants for the GHG production potential (Gebert et al., 2006; Grasset et al., 2021; Lovelock et al., 2017; Middelburg, 1989; Wijdeveld, 2004).

Beside the geochemical characteristics of the sediment and its environment, project specific design and operations may have a large role on influencing how much of the potential emission is released. Selecting the optimum construction method, for example capping or underwater placement instead of upland disposal, can be important to limit GHG emission, or potentially improve sequestration

The GHGs produced within soft sediment will at some point escape the sediment and reach the atmosphere. In the past, researchers at Deltares have studied how gas can escape soft sediment (Van Kessel & Van Kesteren, 2002) and when the critical threshold for CH_4 is reached (Wijdeveld, 2004). Gas produced will create bubbles, that can only escape the soft sediment when these bubbles have enough volume to migrate to the surface and by doing so create channels and cracks. How much gas needs to be produced for these cracks to form depends on the soft sediment characteristics. Deposited soft sediment may hold 25 up to 37% of gas prior to it escaping to the atmosphere. Within Deltares models have been built within the numerical model Delcon based on formula by Middelburg et al. (1989) for organic matter decomposition in sediment. This has been applied in various projects amongst which Stryker bay, IJsselmeer, Ketelmeer and de Slufter (WL - Delft Hydraulics 2002, Wijdeveld, 1999).

Over the past years beneficial use of sediment gained ground, considering dredged sediment a valuable resource. By using dredged sediment to aid in the development of ecosystems that have a high potential to capture and store carbon, the sediment can even become a carbon sink. The creation of natural islands in the Dutch Marker Wadden, and a mangrove forest restoration project in Indonesia led to an initial increase in carbon captured in the sediment (Veld, 2018). The Kleirijperij is another example of sustainable use of dredged sediment.

Sustainable use of dredged sediment and finding best practices to keep carbon locked in the sediment, will help the Hydraulic Engineering (HE) sector to become carbon neutral by 2050. Furthermore, it offers economic potential for the carbon market, where the storage of carbon may be sold as carbon credits. A first plea for ecosystem-based carbon footprinting in in marine engineering projects has been published (Dekker et al., 2014; Fiselier et al., 2015)Currently a prototype tool is built for assessing circularity of inland dredging activities including effects on ecosystem emissions (Besseling et al., 2021). This is created with the objective of helping the Dutch Water Authorities to dredge sediments more circularly. Other exist to assess the sustainability of civil construction projects, such as “DuBoCalc” and “CO2 prestatie ladder”, both developed bij Rijkswaterstaat[2]. These tools are beginning to incorporate ecosystem- and sediment-releated GHG emissions to some degree and with difference between tools.
In this specific project, the GHG emissions from dredged material from the Kleirijperij pilot De Kwelder were studied, as a parallel study that leverages on the ongoing pilot, its data and results. Potential follow-up studies will focus on measuring emission from other dredged material related applications as well as various natural systems.

1.2 Scope

The aim of this research project was to measure the emissions of greenhouse gases during the maturation of the organic-rich dredged material from the Eems-Dollard estuary in the clay ripening project “Kleirijperij de Kwelder” (Figure 1.1 and 1.2). This location was chosen because the GHG emission project, which started in 2019, could thus measure GHG emissions as quickly after filling with sediment as possible, whereas the other pilot (Delfzijl), was already in place for some time. The original goal was to measure GHG emissions over more than one year; however, as partly ripened sediment was taken out for tests of applicability in dikes, in situ GHG emissions were stopped after a few months. Longer term effects were estimated based on additional measurements and theoretical calculations.

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[2] https://www.rijkswaterstaat.nl/zakelijk/zakendoen-met-rijkswaterstaat/inkoopbeleid/duurzaam-inkopen

The goals were to:

- Determine emissions of greenhouse gases (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) during maturation of dredged material in the Kleirijperij De Kwelder sedimentation basins; and to
- Evaluate factors that influence these emissions.

Figure 1.1 *The “Kleirijperij de Kwelder” from above in 2020. At the bottom, the plots with dredged sediment can be seen.*

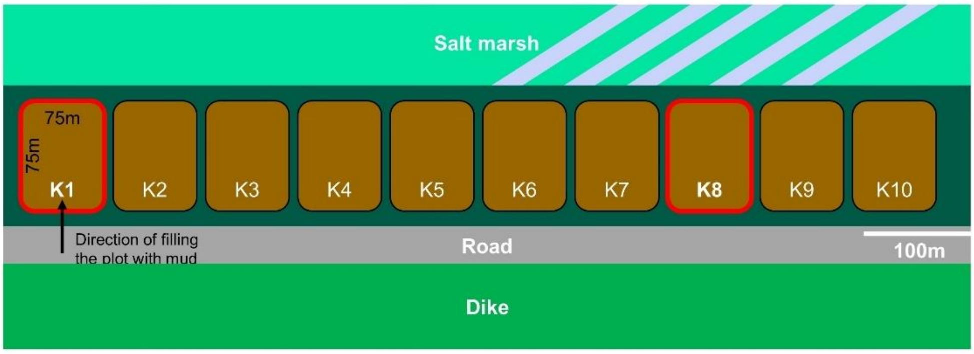

Figure 1.2 *Schematic drawing (not scaled) of the plots at “Kleirijperij de Kwelder” plots (K1-K10).*

2 Materials & Methods

2.1 Study design

Study site

Measurements were done at the ‘Kwelder’ (salt marsh) clay ripening pilot in the north of Holland. Here, in total 10 depots were filled with dredged sediment (Figure 2.1 and 2.2) in Winter 2019-2020. The source of the sediment was the natural reserve “Polder Breebaart”, where the material was dredging using a cutter dredger, and transported through a pipeline. The sediment deposited in the ripening plots is from Breebaart polder, which is connected to the Eems-Dollard intertidal flat.

To determine GHG fluxes from the ripening sediment, experiments were performed both in the field and in the lab.

Field experiment

In the field, GHG fluxes were determined using respiration chambers developed by Deltares (Figure 2.3). The chamber dimensions were: diameter: 19 cm, height (relative to sediment surface): ~22 cm. Six chambers were installed, three in each of two plots (plot K1 and K8; see Figure 2.4) in 2020. In week 5 (April 2020) and week 13 (June 2020) after completion of filling the ripening plots, GHG fluxes were measured and sediment samples were collected. Sediment samples were taken at 3 depths (0-15, 50-60 and 75-100 cm). The chambers were taken out in week 35 after filling. This was earlier than planned; the reason for this was that sediment was taken out for testing for application in dike material earlier than planned.
Key parameters were determined from the sediment samples directly in the field: electrical conductivity, pH and redox potential. Other parameters were determined in the lab: water content, organic matter content and grain size.
As a reference, GHG measurements with chambers were also performed at the adjacent salt marsh.

***Figure 2.3** Cylindrical respiration chamber for sampling greenhouse gases in the field.*

***Figure 2.4** Schematic overview (not scaled) showing placement of the respiration chambers in plot K1 and K8. SK = Dewatering cistern.*

Lab experiments

In addition to the field experiment, two lab experiments were performed to assess GHG fluxes of different stages of the ripening dredged material (fresh estuary sediment, ripe (partly matured) dredged material (sampled in week 16) and freshwater sediment to control for impact of salinity) (table 2.1).

The first experiment focused on controlled ripening in a closed vessel (mesocosms). This was achieved by placing samples of dredged material in a closed vessel with silica gel acting as desiccant (Figure 2.5). Unfortunately, the vessel appeared to be leaking CO_2 and CH_4 hence this experiment was left out of the report.

Figure 2.5 *Closed vessel with sample of dredged material to measure GHG emissions semi-continuously under controlled conditions.*

The second experiment aimed to quantify CH_4 fluxes in detail. To this end, fresh estuarine dredged material, (partly) ripe dredged material and freshwater sediment was incubated (see table 2.2) in flasks (120 ml), closed with a butyl rubber stopper and crimp cap, with a headspace of ambient air. Flasks remained closed for 40 hours. The CH_4 concentration in the headspace was followed in time.

Table 2.2 *Overview of flask incubations for* CH_4 *flux determination*

Incubation	**Dilution**	**Abbreviation**
Estuary sediment	no	ES
Ripe Clay	no	RC
Freshwater sediment	no	FWS
Diluted (1:4) estuary sediment	1 to 4	ES d1:4

2.2 Organic matter content

In the lab, sediment pore water was extracted using rhizons (0.2µm pore size) and vacuum vials. Water content was determined by determining fresh weight of a 15 ml cup filled with the sample, after which the sample was dried for 7 days at 40˚C after which dry weight was determined. Organic matter (OM) content was assessed via thermogravimetric pyrolysis (TGA 701, Leco, MI, US).

From the TGA OM content (%), we calculated the rate of loss of OM between the two sampling points. The OM content was corrected for salt sediment (factor 0.45, based on Waterloopkundig laboratorium, 1985). Organic carbon (OC) within the total OM content was calculated as OM/1.724 (g/g) = OC (Waxman & Stevens, 1930). Subsequently, OC was multiplied by dry density (g/cm^3) of the sediment to get OC as mass per volume. To express loss per area unit, we assumed C loss from the top 1 cm of 1 m^2. Next, we calculated the GHG emission in CO_2 equivalents for the two extremes where all converted C in these 8 weeks was emitted into either CO_2 or CH_4 (using the GHG equivalents for CH_4 being 34 g/g CO_2 , Myhre et al., 2013).

Furthermore, Rock Eval 6 analysis was performed to measure total organic carbon (TOC), mineral carbon (Min C) and asses the quality of the organic matter by calculating I and R indices (according to Sebag et al., 2016).

2.3 Gas concentration

To measure greenhouse gas fluxes (CO_2, CH_4, N_2O), the respiration chambers in the field site were closed by screwing on the lids and closing the luer-lock 3-way connector. Gas samples were collected after closing the chamber for at least 1 hour. In week 5 gas samples were collected at the start and at the end of the closing period. In week 13 chambers were closed for up to 4 hours and sampled at least 5 times from start to end. To take a gas sample, a 20 ml syringe was connected to the luer-lock connector piece and, after flushing three times with gas in the chamber, a gas sample was pulled from the chamber. Gas samples were injected in vacuum 12 ml gas vials and returned to the lab for quantitative analysis.

In the lab gas samples were injected on a gas chromatograph (Agilent G1530a) using the TCD detector (detection limits in table 2.3) to measure CO_2, CH_4 and N_2O. This GC was equipped with an autosampler (PAL, CTC analytics).

The flask incubations were measured by collecting gas samples (250 µl) using a Hamilton syringe and direct injection on a gas chromatograph (Varian 3800 with a tandem column (CP-Molsieve5Å and PoraBOND Q, Agilent Technologies) using the methanizer FID detector, over the course of 40 hours.

Table 2.3 *Detection limits for the GC-TCD analysis*

Compound	%	Ppm
CO2	0.14	1400
N2O	0.15	1500
O2	0.08	800
CH4	0.13	1300

2.4 Cat- and anion analysis

To analysis of cations and anions (sulfate, chloride, sodium) present in sediment porewater was sampled using 0.2µm rhizons and a syringe. Porewater was analysed using the ICP-OES method on a Perkin Elmer Avio 500.

3 Results

3.1 Field

3.1.1 Abiotic conditions

The sediment deposited in the ripening plots showed typical characteristics for mud from the Eems-Dollard estuary intertidal mud flats. This sediment is rich in salt (pore water concentrations: sodium: 3,78 ± 0,35 g/L; chloride: 6,65 ± 0,68 g/L and sulfate: 1,51 ± 0,11 g/L).

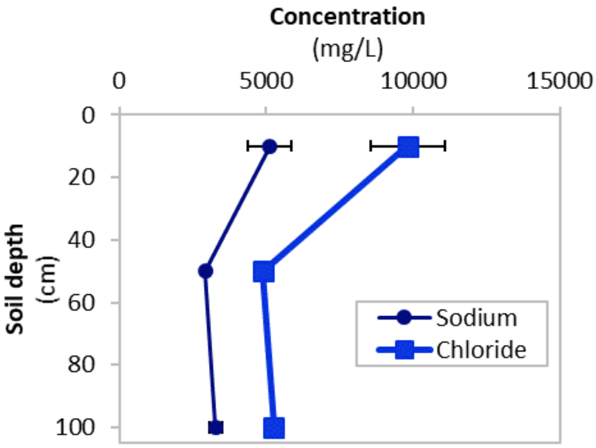

Figure 3.1 *Concentrations of sodium and chloride along a depth transect (10, 50 and 100 cm depth) collected in June 2020 (week 13). Markers represent the mean (n=6) with error bars as SEM.*

After 8 weeks of ripening, the pore water of the top layer showed higher concentrations of sodium and chloride (Figure 3.1). This is most likely due to drying of the top layer, which lost 20% of its moisture in the same period (Figure 3.2 and Figure 3.3). Furthermore, upon ripening, the sediment became even more reduced (redox potential -62 ± 25 mV SHE in April went to -162 ± 17 mV SHE in June; table 3.1), indicating that despite the cracks that form during ripening, oxygen is not able to reach into the blocks of clay (Figure 3.4). Analysis of sediment along a depth transect in June showed that the sediment is relatively homogenous in terms of pH, temperature, electrical conductivity and redox potential (table 3.2). Sulfate concentrations were moderate (June average 1506 mg/L ± 113 (n=6)) compared to seawater (~2688 mg/L); partly, sulfate will have decreased due to conversion into sulphide by sulfate reduction, and partly sulfate will have increased due to drying.

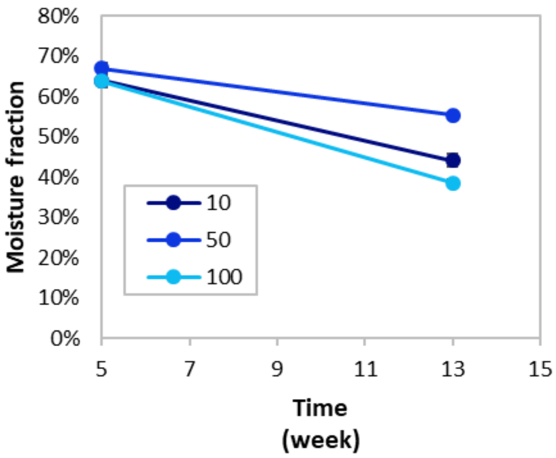

Figure 3.2 *Moisture fraction in de sediment along a depth transect (10, 50 and 100 cm) from samples in April (week 5) and in June (week 13). Markers represent the mean (n=6) with error bars as SEM.*

Table 3.1 *Abiotic conditions in the top layer (0-10cm) measured in the field in April and June (mean (n=6) ± SEM)*

	pH			**Temperature** (° C)			**EC** (mS/cm)			**Redox potential** (mV -SHE)		
April	7,6	±	0,1	12,6	±	0,7	10	±	0,8	-62	±	25
June	7,3	±	0,1	24,4	±	1,0	137	±	69	-162	±	17

Table 3.2 *Abiotic conditions in the field at different depths in June (mean (n=6) ± SEM).*

Soil depth (cm)	**pH**			**Temperature** (° C)			**EC** (mS/cm)			**Redox** (mV -SHE)		
10	7,3	±	0,1	24,4	±	1,0	137	±	69	-162	±	17
50	7,5	±	0,1	22,6	±	0,6	151	±	99	-192	±	21
100	7,3	±	0,1	23,3	±	0,7	199	±	122	-185	±	31

Figure 3.3 *Pictures of the development of the ripening clay week 5 (A), week 13 (B), week 35 (C).*

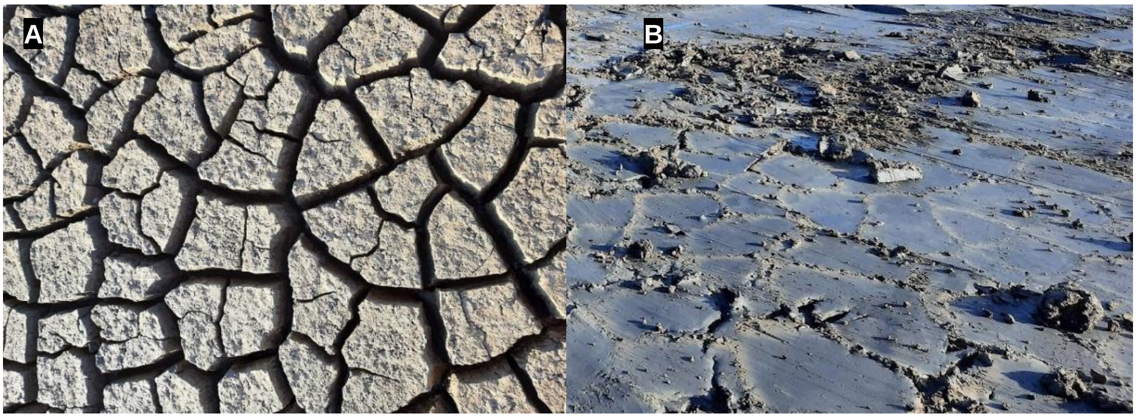

Figure 3.4 *Pictures of the ripening clay in week 35 of a plot that is not ploughed (A) and a ploughed plot (B). The latter shows clear markings of oxidation on position were previously cracks were placed. Also, evident that oxidation is only present on the outer edges of the blocks, in the cracks.*

3.1.2 Organic matter change

Organic matter (OM) content as measured by thermogravimetric analysis (TGA) showed that OM content was highest in the top 10 cm of the ripening sediment in April (12.5% ± 0.3%; mean ± SEM; Figure 3.5). From April to June the organic matter content decreased mostly in the top 10 cm of the sediment (total loss of 3.5% ± 1.3%). The deeper layers showed much less loss of OM (Figure 3.1 and table 3.1). On average, OM content in the top layer decreased with 0.43% ± 0.16% per week (table 3.3). There was no correlation between OM content and moisture fraction in time for the different depths (10, 50 and 100 cm).

Table 3.3 *Average organic matter (OM) loss (% week^{-1}) for the different soil depths (mean (n=6) ± SEM)*

Soil depth (cm)	**OM loss** (% week^{-1})		**SEM**
10	0.43%	±	0.16%
50	0.14%	±	0.02%
100	0.21%	±	0.18%

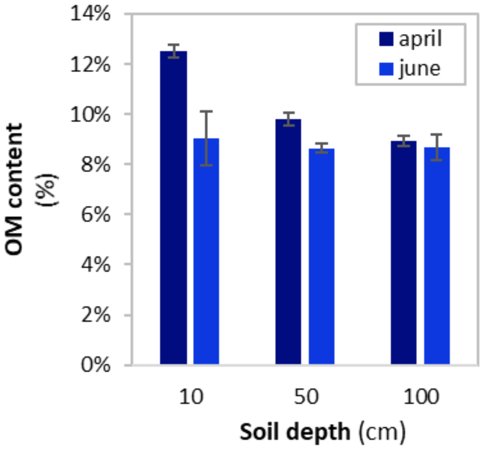

Figure 3.5 *Mean (n=6) organic matter (OM) content of the ripening sludge in April (dark blue) and June (light blue) for the different soil depths (10, 50 and 100 cm). Error bars represent SEM.*

The quality of organic matter was tested at the start of the experiment using Rock Eval 6 analysis. Details on this analysis are given in Appendix B, here a brief summary will be given. This showed a substantially lower amount of total organic carbon (TOC), which was on average 2.6% TOC ± 0.1 (SEM). Mineral carbon content was on average 1.22% mineral C ± 0.02 (SEM). Both TOC and Mineral C showed a similar pattern across all sampling locations, however, like the TGA with depth, the TOC content as measured by Rock Eval decreased, from 3.06% TOC ± 0.06 in the top layer to 2.34 ± 0.05 at 1 m depth.

The calculated I and R indices (Sebag et al. 2016) places the quality of the organic matter within the sediment into a organo-mineral to mineral fraction. This means that the OM pools might be protected from degradation by adsorption onto mineral surfaces. The samples all follow the humic trend, corresponding with organic matter from biological origin in the process of maturation (humification). The organic matter is localized in the middle of the I-R plot, which means that the material is in between very labile (top left part of the diagram) and very stable material (bottom right part of the diagram).

3.1.3 Net GHG emissions field

The net efflux of greenhouse gases (CO_2, CH_4, N_2O) was measured in the field in April and June using the respiration chamber depicted in Figure 2.3. The collected gas samples from April were measured at the gas chromatograph (TCD) and all measurements were at or below the detection limit. In June the measurements were repeated, this time the respiration chambers were closed for a longer period (at least 4 hours). However, even with this prolonged period of collecting gas samples, all measurements were at or below detection limit (exemplified for CO_2 shown in Figure 3.7). The emissions from the clay ripening site were not significantly higher than those from the adjacent salt marsh.

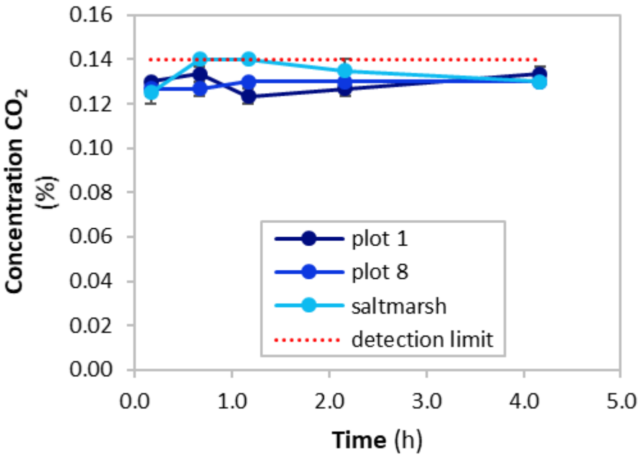

Figure 3.7 *CO_2 concentration (% CO_2) as detected in the gas respiration chambers over the course of time (hours). Markers show mean (n=3) and error bars represent SEM.*

3.2 CH_4 emissions study by flask incubations

To study CH_4 emissions in detail, flask incubations were performed with Estuary Sediment (ES), Ripe Clay (RC) and Fresh Water Sediment (FWS). CO_2 could not be monitored due to the specific column available on the gas chromatograph. Only one treatment was performed: dilution (1:4) of ES with freshwater. CH_4 concentrations were sampled directly from the headspace regularly, over the course of 40 hours and injected on a gas chromatograph (methanizer). Concentrations and rates were calculated (Figure 3.8). The CH_4 emission was absent in the estuary sediment (ES), ripe clay (RC) and diluted estuary sediment (ES d1:4) but was present in a positive control of freshwater sediment 72,8 µg CH_4 / day (fit: R^2 =0,999)).

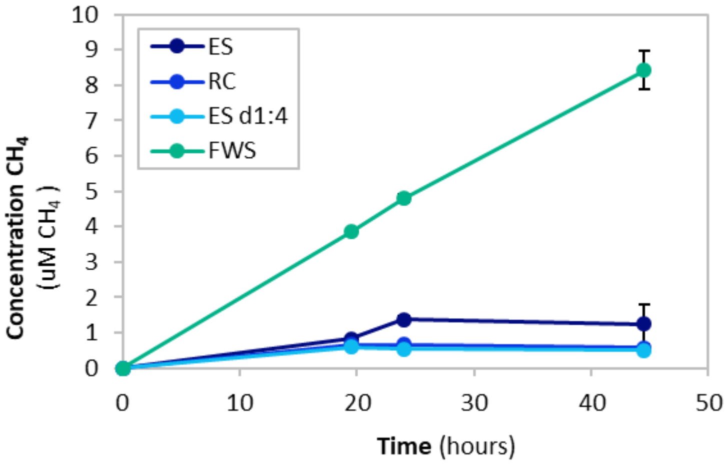

Figure 3.8 *Headspace concentrations of CH_4 in time in the flask incubations. Abbreviations: ES: Estuary Sediment; RC: Ripe Clay; FWS: Fresh Water Sediment. Markers show the mean (n=2) and error bars represent standard deviation.*

4 Discussion

This project set out to determine fluxes of greenhouse gases (CO_2, CH_4, N_2O) during maturation of dredged material in the Kleirijperij depots. Next to that, development of related parameters such as organic matter content and relevant chemical parameters in the sediment (water content, redox potential, macro-ions) were assessed during maturation (ripening) of the dredged material. To explain assess the effect of salt and sulfate, some control experiments were done including freshwater sediment. In this chapter we will discuss the results of these measurements in relation to clay ripening and greenhouse gas emissions. We will also discuss calculated estimates for GHG emissions based on measured organic matter decrease.

4.1.1 Clay ripening

The sediment in the depots is typical of Eems-Dollard estuary sediment, rich in salt and sulfate. From April 2020 to June 2020 it was a very dry and warm period, resulting in 20% loss of moisture from the sediment. Consequentially, the sulfate and chloride concentrations increased, and cracks formed, resulting in a block pattern (Figure 3.3 and 3.4). Blocks of partially matured clay had a more ripe and dry top layer and were wetter in their core. Based on pH, EC and redox potential results, the sediment stayed quite homogenous during this ripening period. Only the redox conditions changed and became even more reduced with ripening. This is a clear indication that oxygen was not able to access the sediment. Visually, the same result was observed. The blocks showed oxidation on the outside (lighter gray coloring of the sediment) along the cracks, but once cut, oxygen proved to only access the first few mm of the surface that was exposed to the atmosphere, leaving the rest of the block anaerobic (Figure 3.4). Transport out and into undisturbed ripening sediment was thus very limited.

The maturation of the organic matter appeared to be low: very little OM loss was observed in time. Only in the top 10 cm OM decreased in time, but with high variation. This might indicate that CO_2 and CH_4 emissions were limited.
The limited oxygen diffusion combined with the increased sulfate and chloride concentrations due to moisture loss, are likely to hamper the ripening process. (An experimental side effect of the high salt concentration is that it might affect the TGA method that was used to determine the organic matter content and cause high variation in OM content.)

4.1.2 GHG emissions

Emissions of CO_2 and CH_4 from ripening sediment measured in field experiment were below detection limit.
There are several possible reasons that can explain this. Firstly, it was observed that the sediment remained very dense and reduced over the timescale of the measurements. This will also mean that the rates of transport from oxygen into the sediment is slow, slowing down any potential breakdown of organic matter that might result in GHG formation. Secondly, the same slow transport will lead to slow transport of any potentially formed GHGs. A third potential reason for low GHG emissions from the sediment is the fact that the high sulfate and salt concentration in the sediment might limit CH_4 emissions via competition between sulfate reducing microorganisms and methane producing microorganisms. The sulfate reducers are known to outcompete methane producing microorganisms under anaerobic conditions (Oremland & Taylor, 1975), resulting in limited CH_4 formation. The flask incubation experiment (Figure 3.9) confirmed this by showing that freshwater sediment produced CH_4, whereas sulfate rich sediment (estuary sediment and partially mature clay) produced little to no CH_4.

CO_2 was not measured in this experiment, but as CH_4 is a much stronger greenhouse gas, this has important consequences for the greenhouse gas emissions from sediments.

GHG emissions can also be calculated based on the measured decrease of organic matter (OM) (see Appendix C for more details on the calculation method).
In order to calculate GHG emissions the data collected from specific sampling points and depths were assumed to represent certain depth ranges within the ripening sediment:

- 10 cm sediment samples: representative of the bulk density and soil organic matter at the 0 - 30 cm[3]
- 50 cm sediment samples: representative of sediments at the 30 - 60 cm depth
- 100 cm sediment samples: representative of sediments at the 60 – 100 cm depth.

We assumed that losses in carbon stock were largely emitted in the form of CO_2 rather than CH_4, based upon our field and lab experiments and given the high salinity contents of sediments. Under this assumption, we report the carbon losses as potential CO_2 emissions, or CO_2 equivalents (CO_2e) – obtained by multiplying C values by 3.7, the molecular ratio of CO_2 to C.
Two scenarios for GHG emissions were calculated: scenario 1 corresponding to the shallow and limited organic matter degradation observed in the field study, and scenario 2 being a more theoretical scenario corresponding to deeper and higher organic matter degradation, assuming that the desired quality of 5% organic matter content would be reached.

Scenario 1: shallow and limited organic matter degradation (representing field data)
Based on the measured organic matter content for the different layers measured in April and June, a mean carbon loss between April and June was calculated of 2 kg C/m^2. This corresponds to a mean potential GHG emission during the April to June clay ripening period of 7.7 kg CO_2e m^{-2} of sediment (Table 1). Extrapolating to an entire single sampling pond (75 x 75 m or 5625 m^2), we estimate the total emissions from a single clay ripening plot was 43.1 ton CO_2e per plot. The CO_2e emissions from the entire 10 plots ripening sites was 431.5 ton CO_2e (Table 4.1). This comes down to 0.012 ton CO_2e per ton clay.

Scenario 2: deep, high (5%) OM degradation (desired quality)
This scenario assumed higher loss of OM (to 5%) across the whole depth transect of the sediment. As can be expected, the estimated CO_2 emission in this scenario was significantly higher (27 kg CO_2e m^{-2}) than in scenario 1. In general, the CO_2 emission will be proportional to the depth to which the organic matter will be degraded, and the extent to which organic matter content decreases.

Table 4.1 *Estimated CO_2 emissions per m^2, per plot and per ton of clay for two scenarios.*

Scenario	**CO_2 emission**		
	(kg CO_2e m^{-2})	(ton CO_2e $plot^{-1}$)	(ton CO_2e ton^{-1} clay)
1. Shallow, low OM loss (represents field data)	8	43	0.012
2. Deep, high OM loss (desired quality)	27	149	0.05

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[3] The depth range of 0-30 cm for the shallowest layer are probably an overestimation, as visual observations made clear that the oxidized zone extended less deeply, say 10 cm. On the other hand, the formation of cracks might propagate exchange.

Given the small sample size, short sampling time and indirect measures of organic carbon, data and results must be viewed with caution as there is likely a high degree of uncertainty. However, the results do suggest that carbon emissions from the clay ripening process may be significant. Further investigations using accurate portable infrared gas analyzers and intense sampling of sediment carbon pools during the entire clay ripening process is recommended in order to obtain more precise estimates of greenhouse gas emissions from the clay ripening process.

The differences in GHG emissions calculated for the two scenarios demonstrates the influence that environmental conditions, time and sediment management may have on the CO_2 emissions. The depth and degree to which organic matter is degraded strongly determines the CO_2 emission, and this is likely strongly dependent on ripening strategy (e.g.: ploughing, influencing salinity) as well as the carbon quality of the sediment organic matter. Thus, the different ripening strategies would influence the rate and quality of CO_2 emissions. Furthermore, we demonstrated a much higher GHG emission from freshwater sediment compared to saltwater sediment. Thus, also the choice between saltwater and freshwater sediment can strongly influence sediment-related GHG emissions, as the production of CH_4 and consequently the emission of GHG is most likely much higher in freshwater sediments.

To put the numbers calculated here into context, it is necessary to compare these to potential alternatives. For building dikes, an alternative source of dike clay is the import of freshwater sediment from Belgium, low in organic matter. An important component of the GHG emission related to this alternative is the emission due to fossil fuel combustion. An estimate can be made of this emission based on known numbers (https://www.co2emissiefactoren.nl/). The emission of a truck (of 10-20 ton) is estimated to be 0.256 kg CO_2/ton kilometer (well to wheel). When we assume one-way transport of a loaded truck over 200 km (from Belgium to the pilot locations), this leads to an emission of:

200 km * 0.256 kg CO_2/ton kilometer = 51.2 kg CO_2 / ton clay or 0.05 ton CO_2 / ton clay.

This comes down to the same order of magnitude as the GHG emissions from the clay ripening process under the worst-case scenario 2. Actual emissions from transport will be higher, as we did not take into account other activities, and only included one-way transport.

4.1.3 Speeding up clay ripening

Speeding up oxidation of soft sediment after deposition might be possible by ploughing, which means it can contribute to the ripening process. It is a question whether ploughing can cause enough mixing to sufficiently improve transport. Furthermore, we doubt whether ploughing sufficiently improves ripening of the OM in the sediment, as the maturation of the OM likely remains hindered by the high salt and sulfate concentrations. Reducing salt/sulfate concentration naturally will occur microbiologically, through plants, crack formation and through precipitation. Further research is required to assess how these processes can be stimulated to improve ripening of the OM. On the other hand, the fact that OM content decreases slowly during ripening indicates that the estuary sediment has high C-storage potential. This might mean that beneficial use so sediment, as in clay ripening and subsequent use (e.g. in dikes) might have the positive side effect of carbon storage. This additional effect deserves further quantification.

5 Conclusions & Recommendations

Conclusions

Over the course of two months (April-June), GHG emission (CO_2 and CH_4) from ripening estuarine sediment was very low. This is probably caused by the high concentration of sulfate, which can lead to the inhibition of methane production due to competition with Sulfate Reducing Bacteria (SRBs). Increase in salt and sulfate concentration upon drying and limitation of transport through the ripening clay will further limit GHG emissions.

The role of salt in limiting GHG emissions from sediment was further supported by laboratory experiments of CH_4 emissions from sediment in closed vessels: freshwater samples showed significant CH_4 emissions, whereas estuarine samples did not.

Based on decrease in organic matter content over time, GHG emissions over longer timescales (months) were calculated. The calculated emissions, while higher than measured, were lower than emissions related to alternative sources for organic matter for the eventual use of the sediment.

Although the calculation method needs further quantification, it provides an easy tool to estimate GHG emissions from sediment management and enable to evaluate effects of different alternatives.

Recommendations

We recommend expanding the GHG emission study to a range of sediments that are typically dredged and deposited on land. This will help to bring perspective in maturation of dredged sediment and accompanied emissions. It will also help to objectively determine the extent of GHG emissions from sediments with respect to other emission sources within the project and as background. A range of aquatic sediments varying in salt concentrations (freshwater, brackish, salt) would allow to study the effect of salt and sulfate. A range of organic matter content and age of this OM would allow to study whether OM maturation occurs once the sediment is deposited on land, and how this affects GHG emissions.

In this study we only focused on emissions of the ripening sediment. However, there are many process steps prior to the ripening period of the sediment that are not taken into account. It would be good to study GHG emissions all the way from pre-dredging up to the moment the ripened clay has been implemented in the dikes. The pre-dredging scenario gives an indication of a reference GHG emission to compare the other emissions too. Ultimately, should GHG emissions from sediment be proven significant, this data should be incorporated into a tool that is able to calculate (predict) C-emissions, C-capture and C-storage for these types of dredging activities.

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A Ripening of the plots over time in pictures

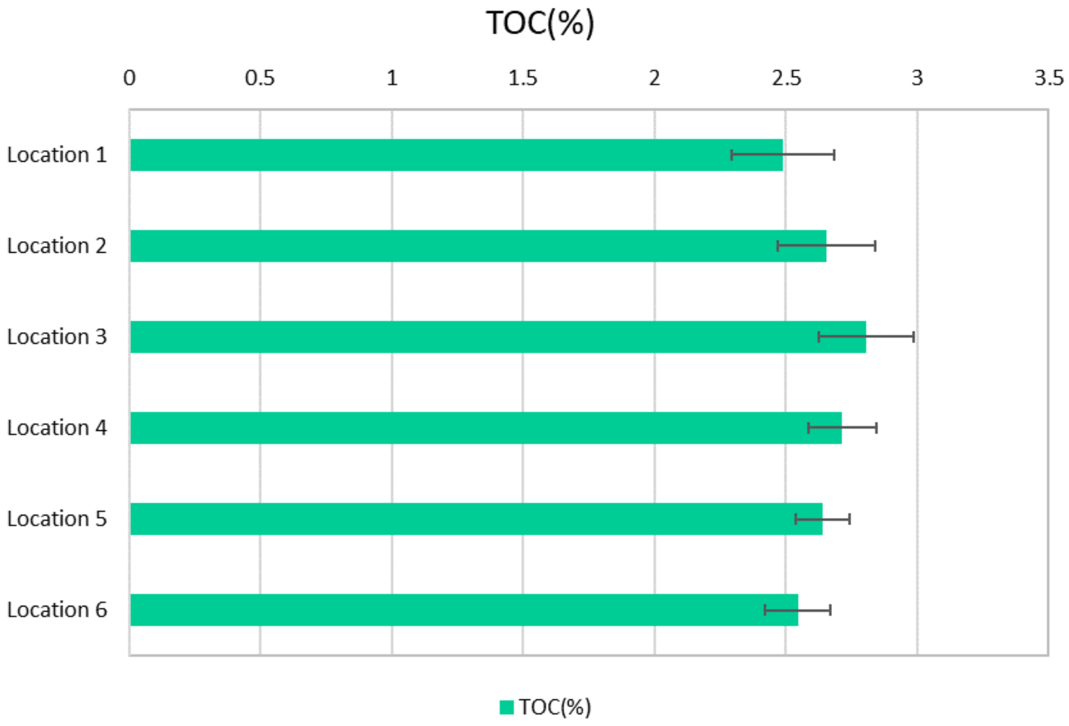

Depot K8
(April) Week 5

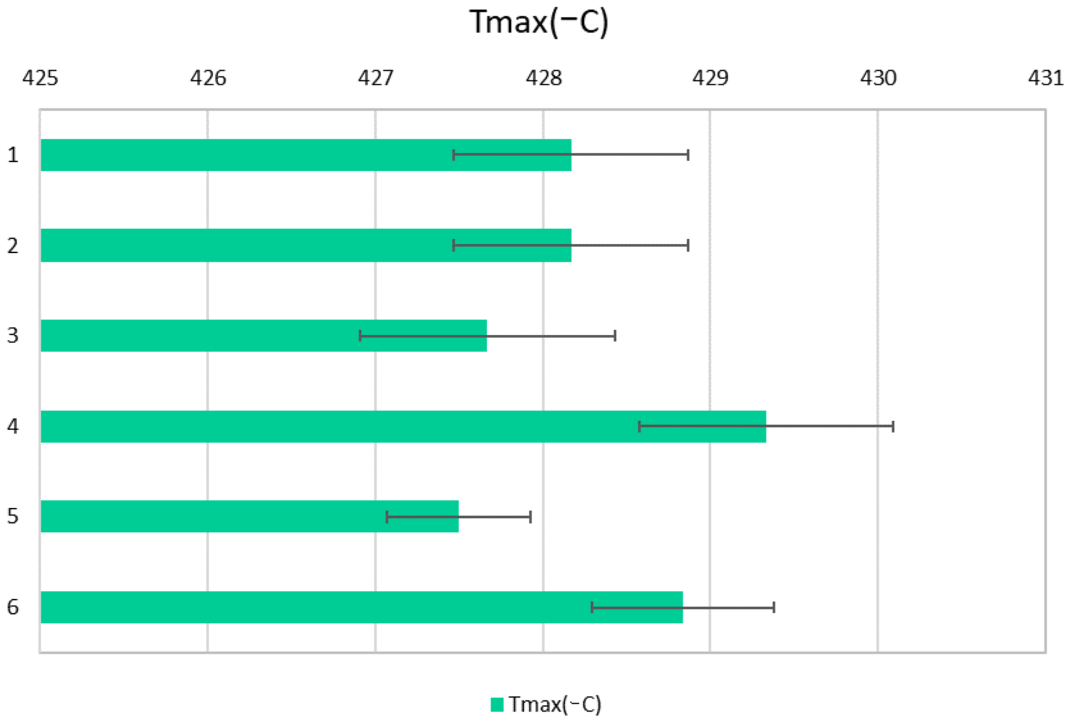

Depot K8
(June) Week 13

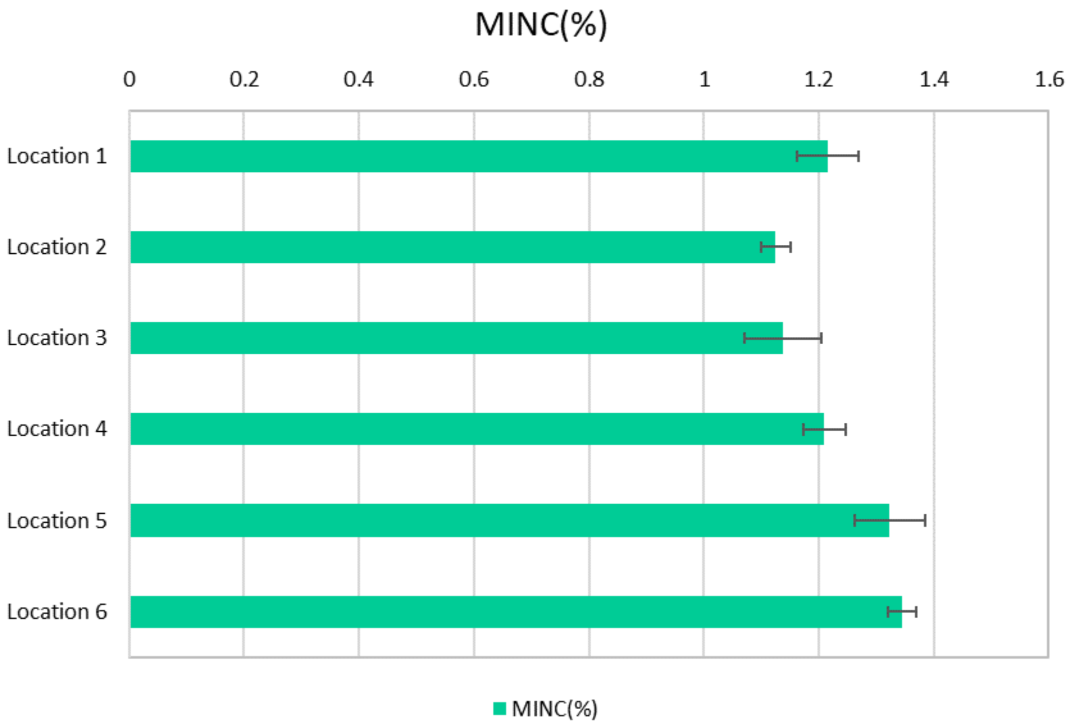

Depot K8
(November) Week 35

B Rock Eval 6 results

Rock Eval results per location



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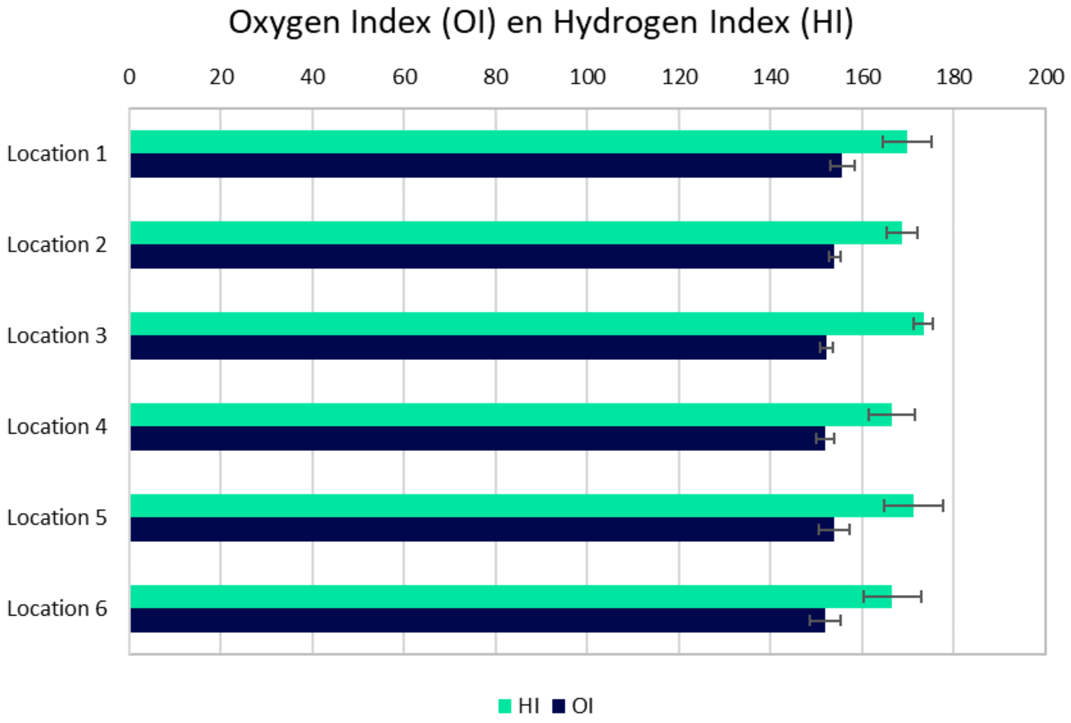
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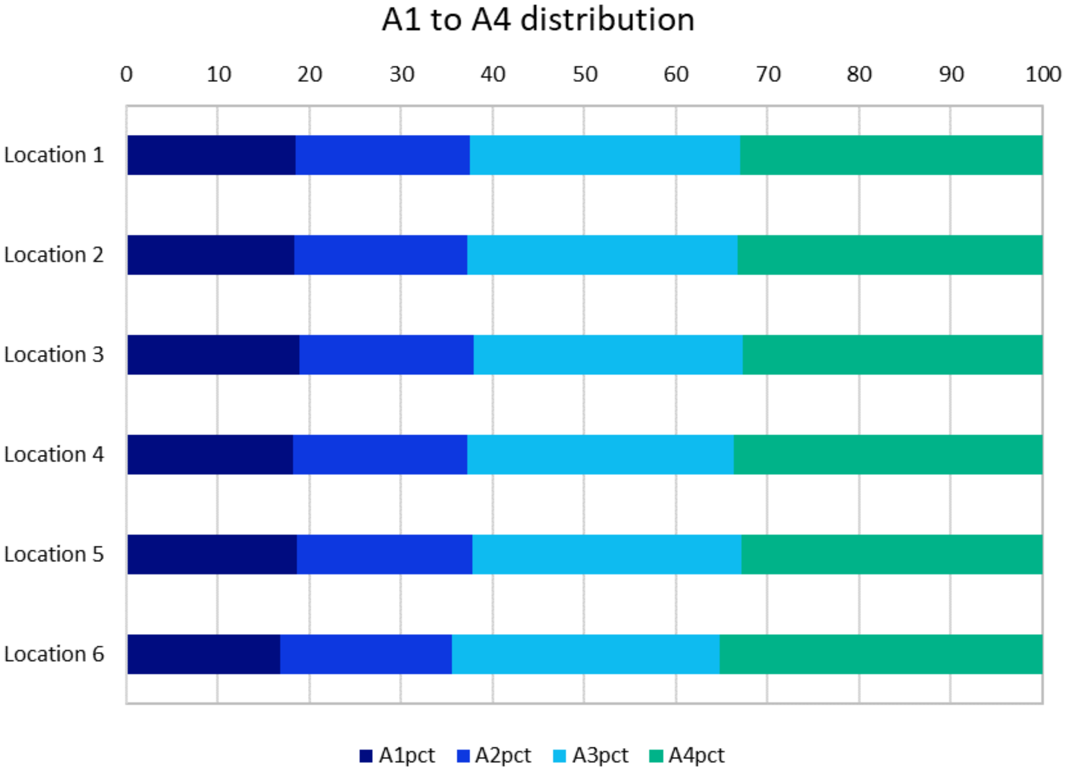
A1 to A4 distribution
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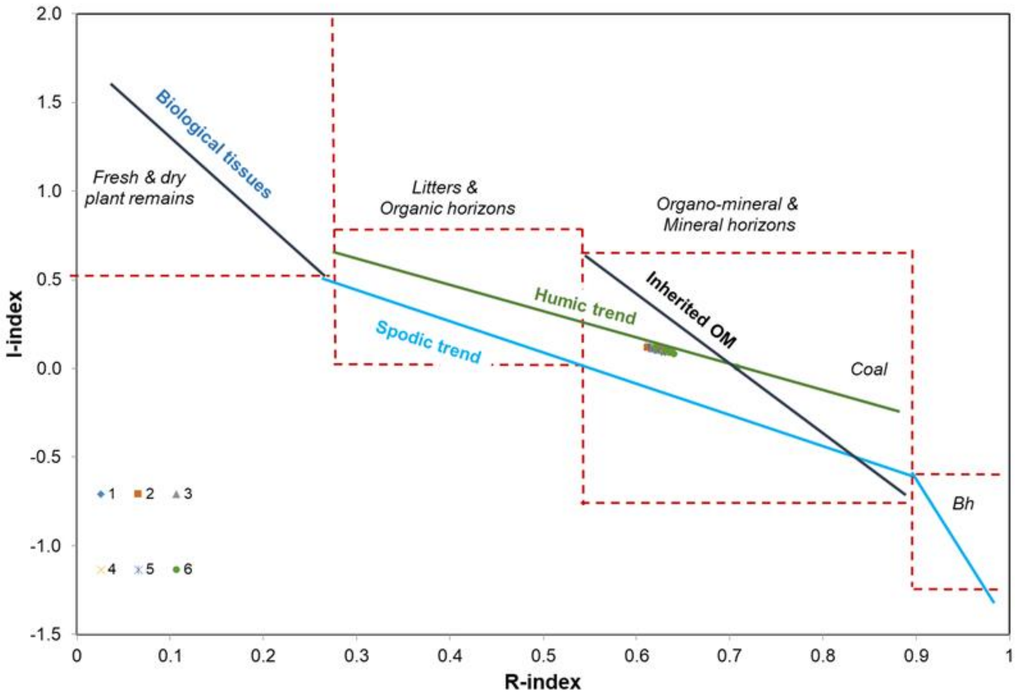
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Litters & Organic horizons
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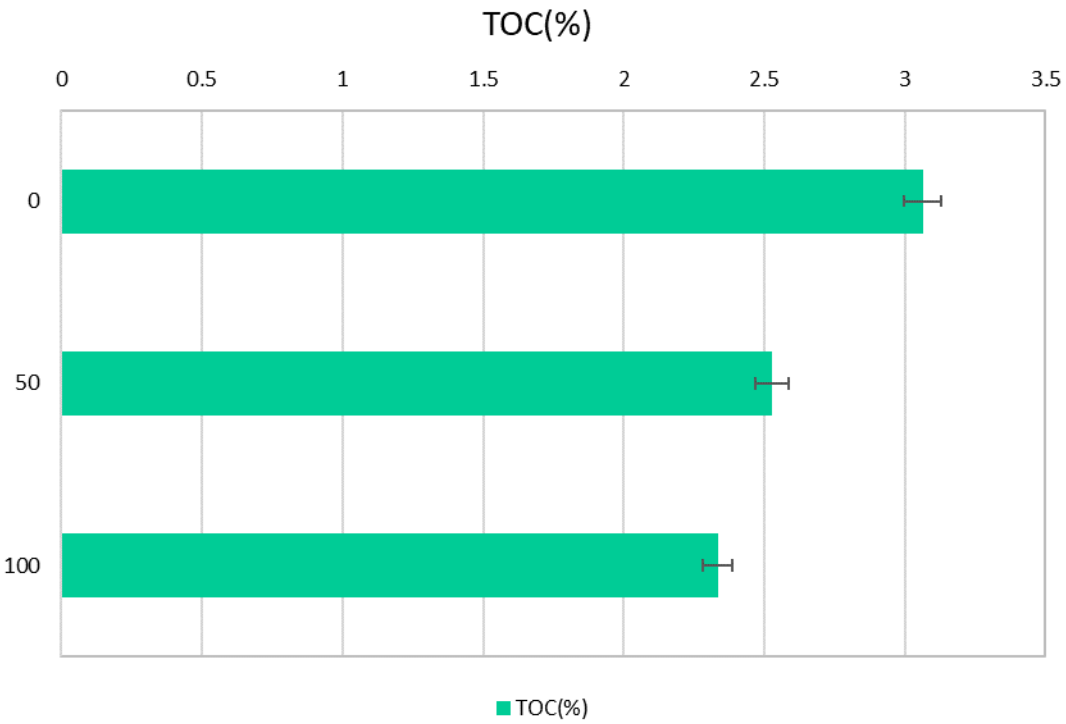
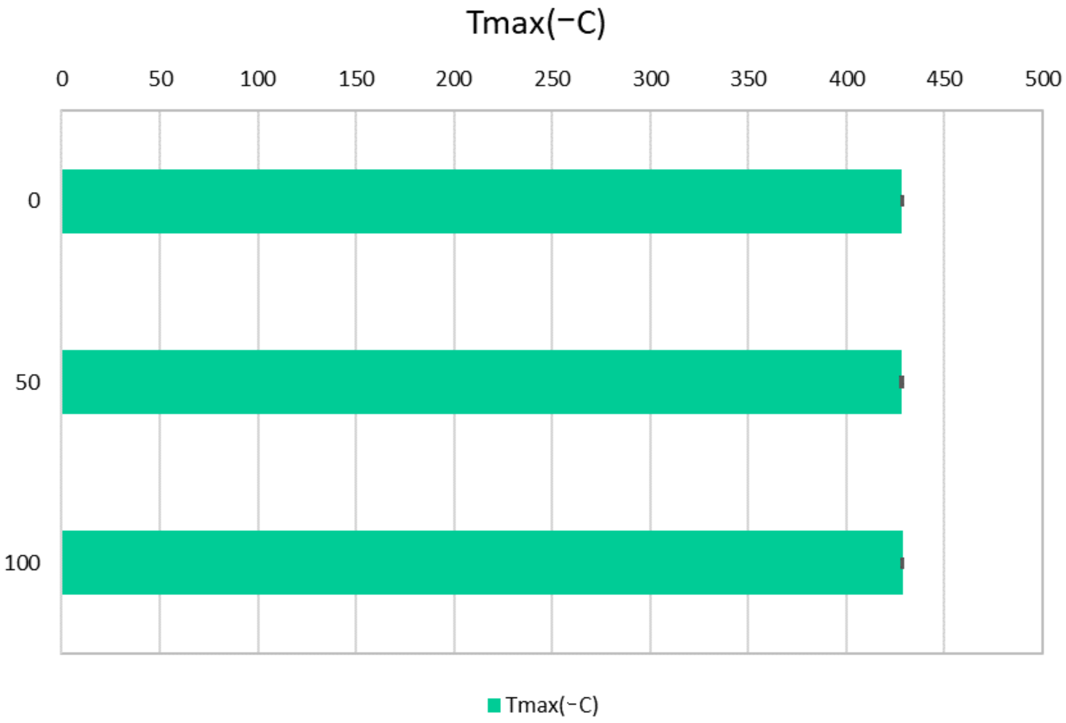
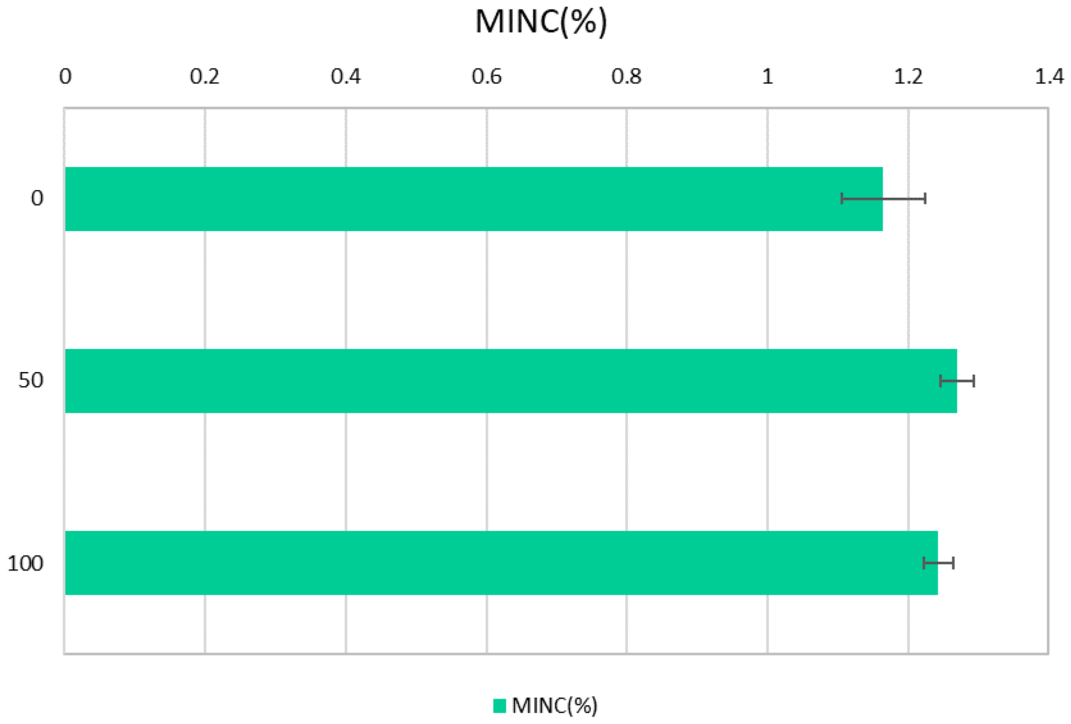
Rock Eval results per depth



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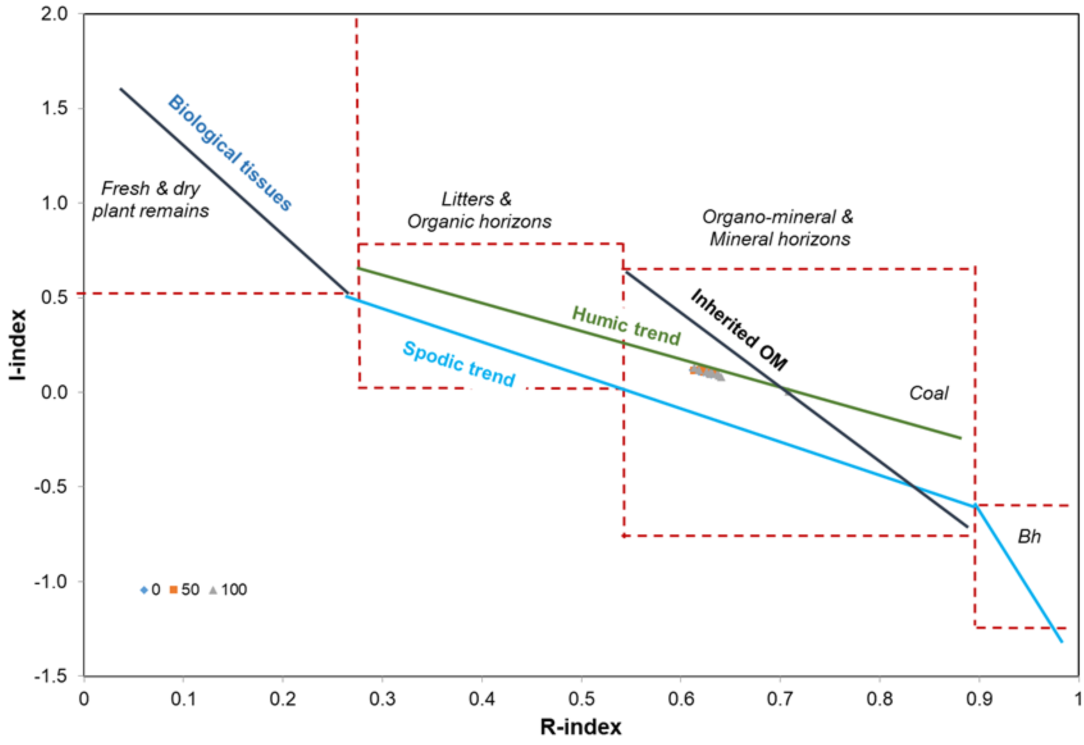
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R-index
Biological tissues
Fresh & dry plant remains
Litters & Organic horizons
Organo-mineral & Mineral horizons
Humic trend
Inherited OM
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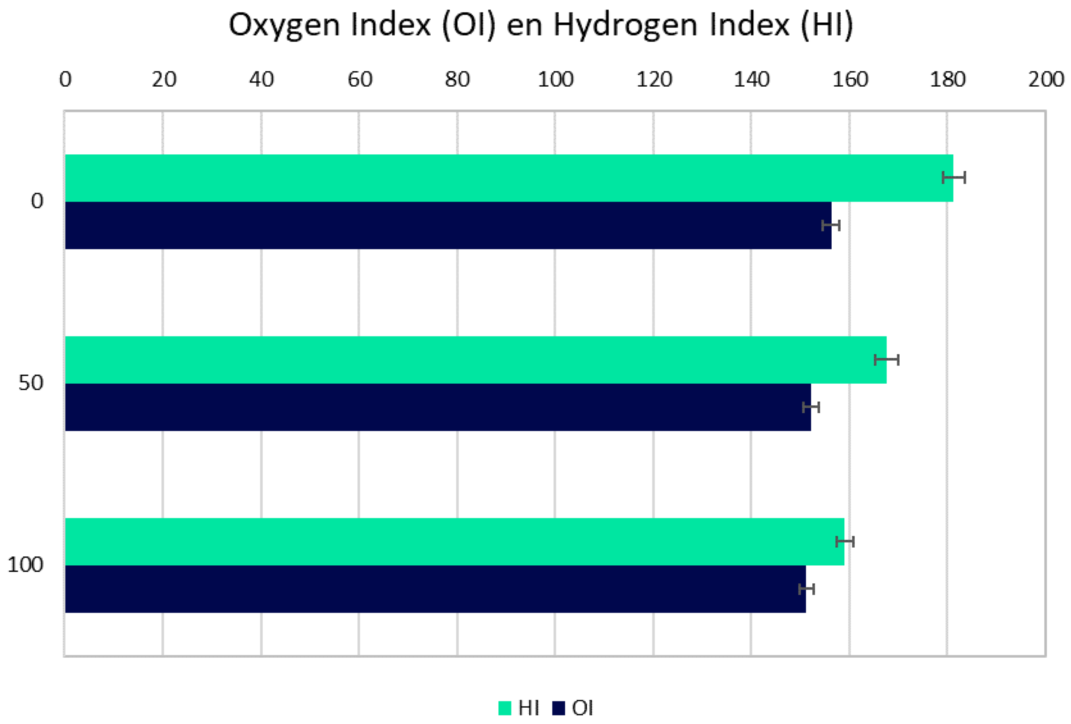
Oxygen Index (OI) en Hydrogen Index (HI)
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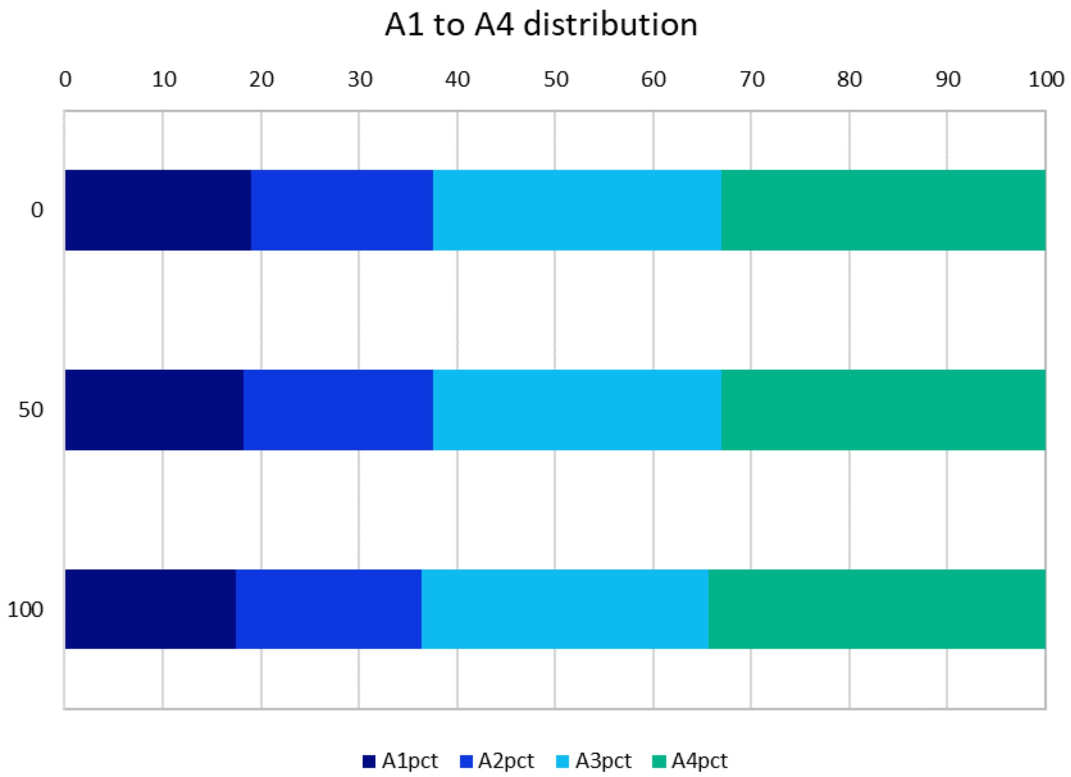
A1 to A4 distribution
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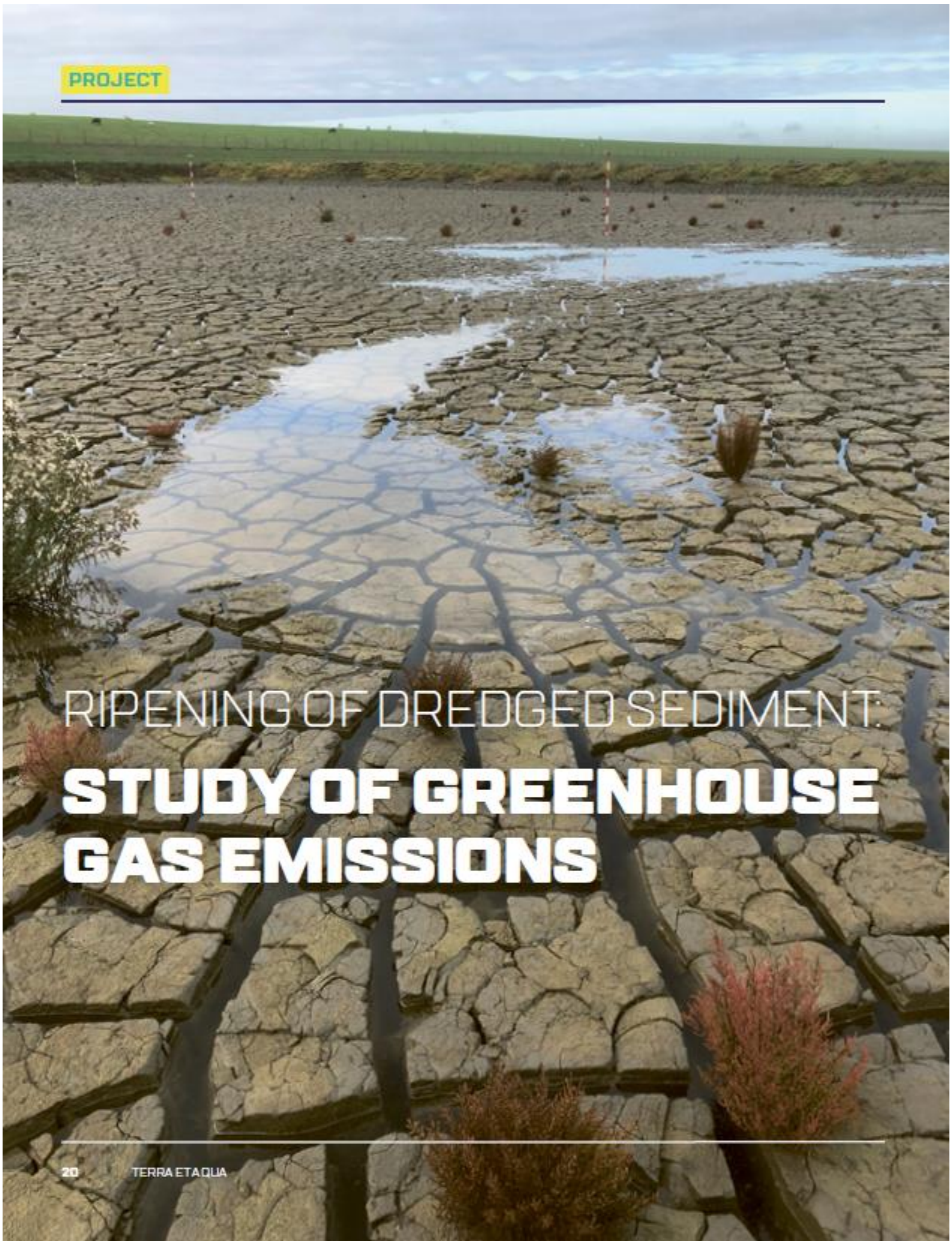
C Publication ‘Greenhouse gas emissions during ripening of dredged marine sediment – case study from the Eems-Dollard estuary’



PROJECT

RIPENING OF DREDGED SEDIMENT:

STUDY OF GREENHOUSE GAS EMISSIONS

20 TERRA ET AQUA

Two clay ripening pilots were constructed in the province of Groningen: Delfzijl and Kwelder, established in 2018 and 2020 respectively.

As increasing greenhouse gas (GHG) emissions contribute to global warming, it is becoming more important to consider the carbon footprint of hydraulic engineering projects. This carbon footprint is more complex than previously thought however, as it can also include the carbon dynamics of the sediments from which projects are built. The purpose of this study was to provide a first approximation from sediment-related GHG emissions of dredged sediments. Using the case study of the clay ripening pilot project ('Kleirijperij') in Groningen, the Netherlands, one phase of sediment processing was examined: the ripening of dredged sediments for use as a clay material in dyke construction.

When calculating GHG emissions from dredging and dyke infrastructure developments, the focus is on the emissions arising from operations and transport (e.g. fossil fuel combustion). However, the carbon stock concealed in ecosystem sediments, has the potential to be released as GHGs by dredging, drying, processing and further use. To date, these sediment-related GHG emissions arising from disturbance are often not accounted for in life cycle analysis (LCA) of hydraulic engineering projects. It also not known how much of the stored carbon is released via GHG emissions upon disturbance.

Clay ripening pilot project ('Kleirijperij'): a win-win

The aim of the clay ripening pilot project ('Kleirijperij') is to study innovative methods to transform locally dredged soft sediments into clay soil suitable for dyke construction. The pilot project monitored a range of physical and chemical characteristics over 2 years and assessed the suitability of the clay product for dyke construction. Two clay ripening pilots were constructed in the province of Groningen: Delfzijl and Kwelder, established in 2018 and 2020 respectively (Figure 1). Both pilot projects consisted of multiple test beds to test whether the conditions, such as the deposition layer thickness, the ploughing frequency and the presence of plants, aid the ripening process and eventually the clay quality. Ultimately, the finished ripened clay product will be used for the construction of the 'Brede Groene Dijk' (The Wide Green Dyke). Clay ripening from soft sediment is a form of beneficial use of dredged sediment. The Eems-Dollard estuary has to be dredged annually for transport purposes (mainly for the harbours of Delfzijl and Eemshaven) as well as ecological purposes. The high turbidity of the estuary is an ecological concern. Structurally removing approximately 1 million tonnes of sediment per year can have significant effects on this turbidity (Van Maren, 2016), which is a driver for larger scale future dredging. To evaluate large-scale use on land of dredged Eems-Dollard sediment in the future, several pilots are being conducted within the Eems-Dollard 2050 programme. The clay ripening pilot is one of them (Sittoni, 2019) and is executed by the Province of Groningen, Groninger Landschap foundation, Groningen Seaports, Rijkswaterstaat North Netherlands, water authority Hunze en Aa's and the EcoShape foundation.

Ripening soft sediment to clay for dyke construction

The large quantities of dredged sediment can be beneficially used in a number of ways. One is the use of clay for dyke construction. In the Netherlands, there is a great need for high-quality clay to reinforce and raise dykes, in order to adapt to the challenges of climate change. Due to the demand, foreign clay is often imported. Costs of dyke clay and transport, in terms of market value, ecological degradation and the carbon footprint from foreign clay extraction and transport, suggests locally produced dredge spoils provide a promising alternative.

In order to be suitable for use as dyke clay, the estuarine soft sediment from the Eems-Dollard estuary (i.e. the sediment trap Delfzijl harbour for the clay ripening pilot Delfzijl and soft sediment deposits in polder Breebaart, deposited over the past 20 years) has to undergo maturation or ripening. This involves

PROJECT

FIGURE 1

Location and overview of the clay ripening pilot projects in the Netherlands. (A) Delfzijl and (B) Kwelder (showing the salt marsh before sediment filling.

dewatering, desalinisation and degradation of organic matter. Desired targets are a porewater chloride content below 2.4 g L^{-1} and an organic matter content of below 5% (dry matter basis). To efficiently reach this composition, various strategies that may improve aeration and availability of labile organic matter were tested, such as aeration of the sediment through ploughing, flushing with freshwater prior to deposition in the test bed, and stimulating biological factors (i.e. plants and worms).

FIGURE 2

Schematic drawing (not scaled) of the test beds at 'Kleirijperij Kwelder' (K1-K10). Schematic overview (not scaled) showing placement of the respiration chambers in test bed K1 and K8. SK = Dewatering cistern. The photo on the right shows cylindrical respiration chambers for sampling greenhouse gases in the field, prior to placement.

How ripening can result in GHG emissions

Coastal estuarine sediments are carbon sinks (Macreadie et al., 2019). Multiple conditions in the coastal estuary, including anaerobic (oxygen-free) conditions in sediments, result in this long-term storage. Dredging activities in the coastal estuary disturb processes both in these ecosystems and in the dredged material. This can result in the release of the stored carbon in the form of GHG emissions. GHG emissions are likely to vary depending on the dredging method, approaches to deposition and the composition of the dredged material. Oxygen is the key element that, when available, facilitates fast microbial degradation of organic carbon stored in the fine sediment. This results in loss of organic carbon as GHG carbon dioxide (CO_2).

During and after dredging, GHGs escape from the dredged material as a result of microbial degradation of organic matter. Following dredging, increased availability of oxygen to the sediment speeds up the degradation process, resulting in the reduction of organic carbon content through increased CO_2 emissions. There is a growing awareness that this source of GHG emissions might be significant for hydraulic engineering infrastructure projects (dykes, harbours, aquaculture, etc.) (Fiselier et al., 2018). However, few reliable measurements or assessments of GHG emissions due to ecosystem-derived carbon losses in hydraulic engineering projects are available.

In this study, we made a preliminary assessment of the carbon loss and resulting GHG emissions from dredged sediments during the clay ripening phase of a hydraulic engineering pilot project. Our goals were to illustrate an approach to assess carbon losses, GHG emissions and key processes involved. In the end, we hope to propose a framework for comparing emissions between different practical options.

Monitoring GHG emissions during the clay ripening pilot project

GHG emissions were measured in the first 3 months of the ripening of fine coastal sediments at the clay ripening pilot project Kwelder. Measurements were performed both in the field and in the laboratory. In March 2020, the clay ripening pilot Kwelder was established and filled with fine sediment.

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FIGURE 3

Development of the ripening clay week 5 (A), week 13 (B) and week 35 (C).

This sediment originated from Polder Breebaart, a salt marsh area connected to the Eems-Dollard estuary and thus subjected to (dampened) tidal effects, which resulted in a net increase in sediment over the years. The material was removed using a cutter dredger and pumped at a low density (ca 1.05–1.10 kg/m^3) to the mud ripener over ca 10 km, where it was deposited in ten test beds (K1-K10) (see Figure 2).

Sampling sites to gather GHG emissions were carefully selected to contain fine sediments that made up the largest part of the deposit. These sites were far away from the entry location of the sediment, as mainly heavy particles (sand) settled near these locations. In two of these test beds (K1 and K8), GHG measurements were taken. These test beds were not treated by ploughing, desalinisation methods or introduction of plants. In each test bed, three sediment sampling points were selected and respiration flux chambers (see Figure 2) were installed. Measurements were performed in April and June 2020, 5 and 13 weeks respectively after deposition of the material.

To monitor GHG emissions in the field, flux chambers were closed for 4 hours to collect gas samples, which were later analysed in the laboratory. To support field measurements, sediment from both clay ripening pilot projects was incubated under controlled conditions in the laboratory to measure methane (CH_4) production, by monitoring headspace CH_4 concentration. For comparison, fresh salt marsh sediment (starting material in the Kwelder pilot), ripe salt marsh clay from the Delfzijl pilot and freshwater sediment (from a ditch) were incubated in parallel in the laboratory.

To analyse physical and chemical ripening in test bed K1 and K8, sediment samples were collected at three soil depths (10, 50 and 100 cm[1]) below the surface and analysed for electrical conductivity, pH, redox potential, bulk density (BD) and organic matter (OM) content. OM was analysed by loss on ignition with thermogravimetric pyrolysis (using the TGA-701 by Leco.).

Physical and chemical ripening of the clay

Over time, the fine sediments dried and consolidated to form a denser substrate. The shrinking, compaction and formation of cracks is clearly seen in the photos in Figure 3. Over the first 3 months, the moisture content decreased on average from 65% in April (±3% Standard Deviation (SD), over entire profile of 1 m) to lower values in June: 44% (±4% SD) in the top layer, 55% (±1.5% SD) in the intermediate layer and 39% (±2% SD) in the deep layer. Chloride and sulfate concentrations increased in the top layer due to evaporation of water. Electrical conductivity (EC) was used as a proxy for salinity and increased over ten times during the initial 3 months of ripening (April 10 ± 0.8 mS/cm; June 137 ± 69 mS/cm).

Furthermore, the redox potential decreased, reflecting increasingly reduced conditions (average redox potential -62 ± 25 mV vs SHE[2] in April and -162 ± 17 mV in June) and low availability of oxygen. Analysis of sediment along a depth gradient showed that after 13 weeks sediment was similar in terms of pH (range 7.3–7.5), temperature, EC (range of averages 137–199 mS/cm) and redox potential (range of averages 162–192 mV). All average values are based on six measurements per sediment layer.

Change in organic matter and field GHG emissions

There was a significant decrease in organic matter in the top layer of the ripening sediment (Figure 4). The OM content in April was highest in the top 10 cm and declined by 28% in 3 months (mean 12.5% in April and 9% in June).

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[1] In some analyses in the first period 75 cm.

[2] SHE = Standard Hydrogen Electrode

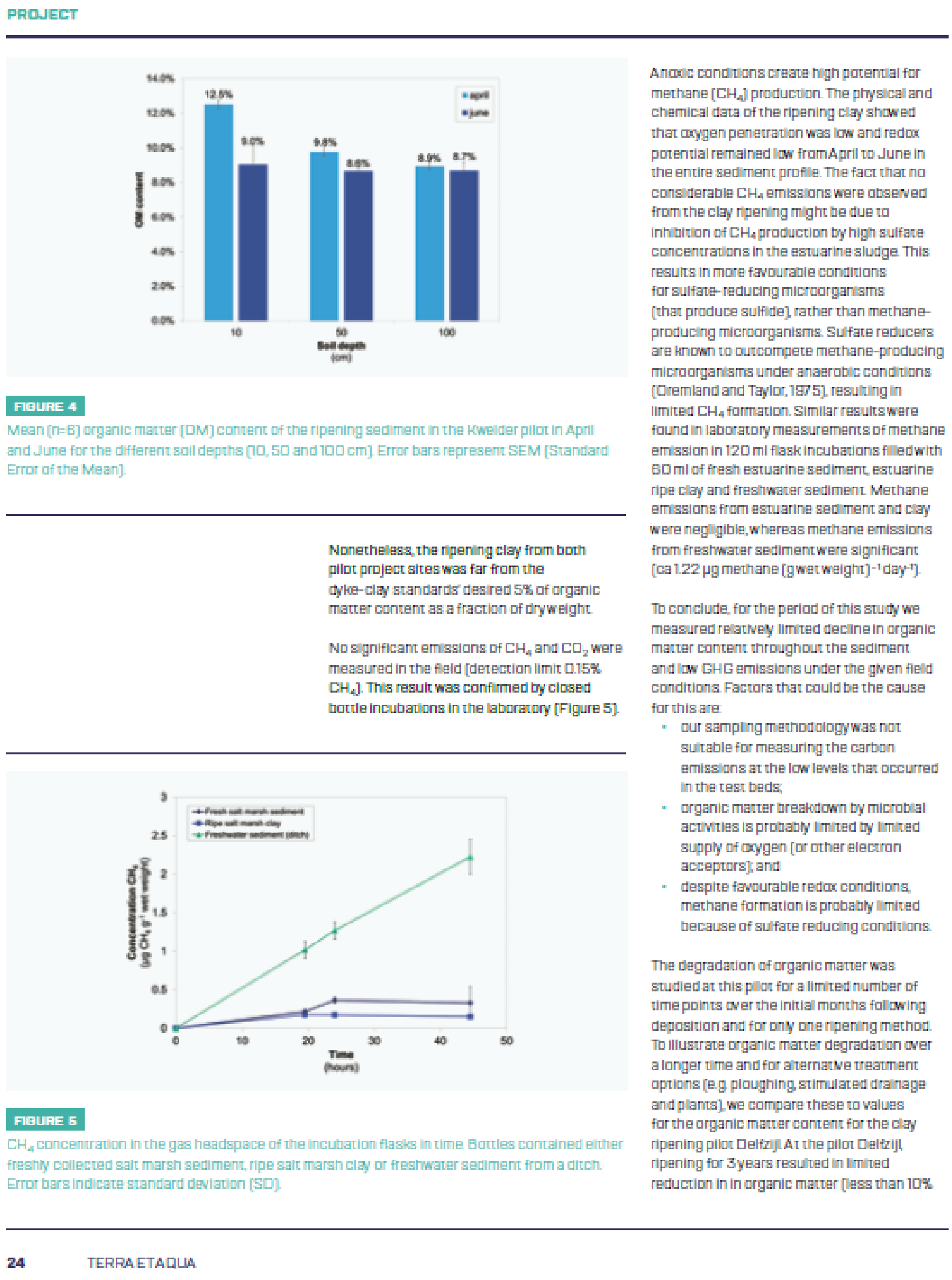

FIGURE 4

Mean (n=6) organic matter (OM) content of the ripening sediment in the Kwelder pilot in April and June for the different soil depths (10, 50 and 100 cm). Error bars represent SEM (Standard Error of the Mean).

Nonetheless, the ripening clay from both pilot project sites was far from the dyke-clay standards' desired 5% of organic matter content as a fraction of dry weight.

No significant emissions of CH_4 and CO_2 were measured in the field (detection limit 0.15% CH_4). This result was confirmed by closed bottle incubations in the laboratory (Figure 5).

FIGURE 5

CH_4 concentration in the gas headspace of the incubation flasks in time. Bottles contained either freshly collected salt marsh sediment, ripe salt marsh clay or freshwater sediment from a ditch. Error bars indicate standard deviation (SD).

Anoxic conditions create high potential for methane (CH_4) production. The physical and chemical data of the ripening clay showed that oxygen penetration was low and redox potential remained low from April to June in the entire sediment profile. The fact that no considerable CH_4 emissions were observed from the clay ripening might be due to inhibition of CH_4 production by high sulfate concentrations in the estuarine sludge. This results in more favourable conditions for sulfate-reducing microorganisms (that produce sulfide), rather than methane-producing microorganisms. Sulfate reducers are known to outcompete methane-producing microorganisms under anaerobic conditions (Oremland and Taylor, 1975), resulting in limited CH_4 formation. Similar results were found in laboratory measurements of methane emission in 120 ml flask incubations filled with 60 ml of fresh estuarine sediment, estuarine ripe clay and freshwater sediment. Methane emissions from estuarine sediment and clay were negligible, whereas methane emissions from freshwater sediment were significant (ca 1.22 µg methane (g wet weight)$^{-1}$ day^{-1}).

To conclude, for the period of this study we measured relatively limited decline in organic matter content throughout the sediment and low GHG emissions under the given field conditions. Factors that could be the cause for this are:

- our sampling methodology was not suitable for measuring the carbon emissions at the low levels that occurred in the test beds;
- organic matter breakdown by microbial activities is probably limited by limited supply of oxygen (or other electron acceptors); and
- despite favourable redox conditions, methane formation is probably limited because of sulfate reducing conditions.

The degradation of organic matter was studied at this pilot for a limited number of time points over the initial months following deposition and for only one ripening method. To illustrate organic matter degradation over a longer time and for alternative treatment options (e.g. ploughing, stimulated drainage and plants), we compare these to values for the organic matter content for the clay ripening pilot Delfzijl. At the pilot Delfzijl, ripening for 3 years resulted in limited reduction in in organic matter (less than 10%

of the initial amount) (results not shown). This shows that also for the Delfzijl pilot, organic matter degradation remained limited during 2 years of ripening, and alternative methods of ripening did not show significant increase in degradation.

Potential GHG emissions based on changes in sediment carbon stock

Based on the decrease in organic matter concentrations, carbon stocks of sediments and the concomitant GHG emissions of the ripening process can be estimated. This is similar to many studies that have quantified carbon stocks of ecosystems, such as salt marshes and mangroves (Kauffman et al., 2020a; Kauffman et al., 2020b). This is the stock-change approach (SCA) that is also described in the Intergovernmental Panel on Climate Change (IPCC) as an approach to measure carbon stock losses and emissions.

In the current assessment, this SCA approach was performed for two scenarios:

- **Scenario 1:** This scenario represents the results obtained from the field. As described in the results section, transport of oxygen in the sediment and of GHGs out of the sediment was slow, resulting in a thin layer of sediment likely actively emitting GHGs. For this scenario, we assumed the organic carbon degradation values obtained from the clay ripening pilot Kwelder.
- **Scenario 2:** In this scenario, we assumed the clay to reach target values of organic matter content, which were set for the final stage of ripening of this sediment (5% organic matter of dry matter). Starting with 10% of organic matter content in the freshly dredged sediment, this equals a decrease of 5% organic matter loss. In this scenario, this loss of organic matter was assumed for the whole sediment mass. As illustrated with the 2-year data for the Delfzijl pilot, this organic matter degradation was not found in practice, despite efforts to reach the target by aeration through ploughing and the addition of plants. Therefore, this scenario represents a case scenario, not likely to be reached in practice in a short time frame of 2 years. As the Delfzijl pilot showed, even ploughing did not result in this degradation over 2 years.

TABLE 1

Estimated CO_2 emissions per m², per test bed and per tonne of clay for two scenarios.

Scenario	CO_2e emission (kg CO_2e m-2)	CO_2e emission (tonne CO_2e test bed-1)	CO_2e emission (tonne CO_2e tonne-1 clay)
1. Shallow, low OM loss (represents field data)	8	43	0.012
2. Deep, high OM loss (desired quality)	27	149	0.050

Calculation

In order to calculate GHG emissions for the entire test bed for scenario 1, the data collected from specific sampling points and depths were assumed to represent certain depth ranges within the ripening sediment:

- 10 cm sediment samples: representative of the bulk density and soil organic matter at the 0–30 cm[3];
- 50 cm sediment samples: representative of sediments at the 30–60 cm depth; and
- 100 cm sediment samples: representative of sediments at the 60–100 cm depth.

Organic carbon was determined from measured organic matter concentrations from the Kwelder pilot using a relation presented by Fourqurean et al. (2012) and Howard et al. (2014).

$$Y = 0.21 + 0.4X \qquad [1]$$

Where Y is organic C (%) and X = organic matter (%), r²= 0.97.

As is apparent through photos taken at the time of sampling, dramatic changes in soils between the different time periods were observed. In a period of less than 3 months, large cracks had formed in surface layers and a concomitant increase in soil bulk density was observed. Soil bulk density of the surface layers was 0.35 g/cm³ in April compared to 0.67 g/cm³ in the June. Similar responses were found at the middle depths (30–60 cm). Due to differences in the soil bulk density between time periods, comparisons of carbon stocks through examination of the same soil volume would yield incorrect estimations of carbon flux through time.

Therefore, to compare soil carbon stocks adequately, estimates were made using equivalent masses of the mineral soil fraction for April and June (Kauffman et al., 2016; Arifanti et al., 2020). The mineral soil mass is determined through subtraction of the soil organic matter density from the total soil bulk density. Then the total mass of the mineral fraction is determined for the top 100 cm of sediment in April, followed by calculation of the mineral soil mass for the June samples.

We assumed that losses in carbon stock were largely emitted in the form of CO_2 rather than CH_4, based upon our field and lab experiments and given the high salinity contents of sediments. Under this assumption, we report the ecosystem carbon losses as potential CO_2 emissions, or CO_2 equivalents (CO_2e) – obtained by multiplying C values by 3.7, the molecular ratio of CO_2 to C.

Methane emissions from estuarine sediment and clay were negligible, whereas methane emissions from freshwater sediment were significant.

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[3] The depth range of 0–30 cm for the shallowest layer is probably an overestimation, as visual observations made clear that the oxidised zone extended less deeply, say 10 cm. On the other hand, the formation of cracks might propagate exchange.

The presented analyses in this study demonstrate a useful methodology and indicative numbers for sediment-related GHG emissions.

Scenario 1: shallow and limited organic matter degradation (representing field data)

Based on the measured organic matter content for the different layers measured in April and June, a mean carbon loss between April and June was calculated of 2 kg C/m². This corresponds to a mean potential GHG emission during the April to June clay ripening period of 7.7 kg CO_2e m-² of sediment (Table 1). Extrapolating to an entire single test bed (75 x 75 m or 5625 m²), we estimate the total emissions from a single clay ripening test bed was 43.1 tonnes CO_2. The CO_2e emissions from the entire ten ripening sites was 431.5 tonnes CO_2e (Table 1). This comes down to 0.012 tonne CO_2 per tonne of clay.

Scenario 2: deep, high (5%) OM degradation (desired quality)

This scenario assumed higher loss of OM (to 5%) across the whole depth transect of the sediment. As can be expected, the estimated CO_2 emission in this scenario was significantly higher (27 kg CO_2e m-²) than in scenario 1. In general, the CO_2 emission will be proportional to the depth to which the organic matter will be degraded and the extent to which organic matter content decreases. There are currently no indications such a scenario is taking place when ripening sediment from the Ems-Dollard estuary[4].

Results

Given the small sample size, short sampling time and indirect measures of organic carbon, data and results must be viewed with caution, as there is a high degree of uncertainty. However, the results do suggest that carbon emissions from the clay-ripening process may be significant. Further investigation using accurate portable infrared gas analysers and intense sampling of sediment carbon pools during the entire clay ripening process is recommended in order to obtain more precise estimates of GHG emissions.

The differences in GHG emissions calculated for the two scenarios demonstrates the influence that environmental conditions, time and sediment management may have on the CO_2 emissions. The depth and degree to which organic matter is degraded strongly determines the CO_2 emission, and is likely strongly dependent on ripening strategy (e.g. ploughing and influencing salinity) as well as the carbon quality of the sediment organic matter. Thus, the different ripening strategies would influence the rate and quality of CO_2 emissions. Furthermore, with the laboratory incubations, we demonstrated a much higher GHG emission from freshwater sediment compared to saltwater sediment. Therefore, the choice between saltwater and freshwater sediment can also strongly influence sediment-related GHG emissions, as the production of CH_4 and consequently the emission of GHG is most likely much higher in freshwater sediments.

To put the numbers calculated here into context, it is useful to compare these to potential alternatives. For building dykes, an alternative source of dyke clay is the import of freshwater sediment from Belgium or other parts of the Netherlands, low in organic matter. An important component of the GHG emission related to this alternative is the emission due to fossil fuel combustion. An estimate can be made of this emission based on known numbers (www.co2emissiefactoren.nl). The GHG emissions of a truck (weighing 10–20 tonnes) is estimated to be 0.256 kg CO_2/tonne kilometre (well-to-wheel). When we assume one-way transport of a loaded truck over 200 km (from Belgium to the pilot locations), this leads to an emission of: 200 km * 0.256 kg CO_2/tonne kilometre = 51.2 kg CO_2/tonne clay or 0.05 tonne CO_2/tonne clay.

This comes down to the same order of magnitude as the GHG emissions from the clay ripening process under the worst-case Scenario 2. Actual emissions from transport will be higher, as we did not take into account other activities, and only included one-way transport. In general, these calculations should be seen as a first estimate to illustrate the methodology; for a complete comparison, emissions from several activities in both scenarios should also be taken into account.

Discussion

The urgency of all business sectors to address climate change mitigation through reduction of emissions and the sequestration of GHGs is well recognised. If we are to attain the target of the Paris Agreement, all stakeholders must act. In line with this, the Dutch government aims to reduce the Netherlands' greenhouse gas emissions by 49% by 2030, compared to 1990 levels, and a 95% reduction by 2050. In addition, the Dutch Ministry of Infrastructure and Water Management adopted a target to achieve net-zero emissions by 2030. Companies in the maritime and dredging sector have started to adopt net-zero targets. This has already resulted in serious efforts to minimise emissions from hydraulic engineering, particularly in relation to minimising use of fossil fuels and optimising construction materials. The presented analyses in this study demonstrate a useful methodology and indicative numbers for sediment-related GHG emissions, however, many uncertainties remain that deserve further attention. Improved approaches to the sampling of GHG emissions and quantification of the carbon mass within the sediments would facilitate accurate quantification of carbon stocks and GHG emissions from the ripening process.

Direct measurements of GHG emissions are needed to confirm the findings and validate the range of applicability both in the field and in the lab (i.e. Gebert et al., 2019). The use of portable infrared gas analysers would facilitate accurate field measures of trace gas emissions (CO_2, CH_4, N2O). Furthermore, intense sampling of sediment carbon pools

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[4] Evaluations are taking place to see if clay soil of a higher organic matter and salt levels can nevertheless be safely used in embankments.

during the entire clay ripening process is recommended to obtain more precise estimates of GHG emissions from the clay ripening process. This would entail repeated sampling of carbon concentrations and concentration bulk density at varying depths of the sediment beds over time.

The carbon composition and quality are also unknown. The carbon quality is a measure of the quantity of labile and recalcitrant fractions. This is important as microorganisms can readily decompose labile forms while recalcitrant carbon may indefinitely persist in sediments. Knowledge of carbon quality provides information on the time required and potential to reduce organic matter in dredged sediments. In addition, other components of the sediment, such as clay and salt, may have an impact.

Once salts are washed out of the clay and with greater oxygen penetration (via plant roots, bioturbation or after soil is mixed with sand), the degradation of the organic material will be enhanced. However, this could increase methane emissions, which would increase the global warming potential of the gases arising from this ripening process. The sediment in the two pilot projects described here is both estuarine, salt-water sediment from the same region, which could differ in several more specific parameters such as organic matter quality. For other cases, sediment properties (such as organic matter quality, salinity and clay content) might be different and it is recommended to take this into account.

If the objective of a project involving hydraulic engineering is to minimise GHG emissions or even sequester carbon, then building with nature based solutions can be applied, such as using the soft sediments for salt marsh creation where vegetation could uptake CO_2 and store organic carbon in sediments. Salt marshes not only sequester carbon but also reduce wave heights. When dykes are combined with vegetated foreshores, they can be lower and still provide safety (Temmerman et al., 2013). Lower dykes require less clay and thus involve fewer emissions from clay ripening.

In this pilot study, we only focused on processes during the first stage of the ripening of the sediment and a transition from dredged sediment to clay material. Microbial degradation of organic carbon will result in loss of this organic carbon as CO_2 and/or CH_4. Our pilot study already suggested some parameters that affect the sediment-related GHG emissions during ripening of soft sediments: the concentration of organic carbon in the sediment, the quality of that carbon and the salinity that affects whether emissions will be limited to CO_2 or not. The slow transport in and out of the sediment probably also affects degradation conditions, resulting in low availability of oxygen and slow breakdown of organic matter and slow emission of greenhouse gases.

These and other factors might differ with different dredging methods and ways of deposition during ripening. However, there are many process steps before and after the ripening period in the project where GHG emission estimates are still lacking, therefore more research on this topic is needed. The entire life cycle analysis, including the carbon losses and sediment-related GHG emissions from pre-dredging up to the moment the clay has been implemented in the dykes and the further fate of carbon, should be quantified. Finally, the use of relevant reference scenarios in the life cycle analysis is essential, e.g. of natural ecosystems or alternatives of the sediment use.

Conclusions

Our study aimed to assess GHG emissions from ripening soft sediment to dyke clay and to identify key processes involved. Given the small sample size, short sampling time and indirect measures of carbon loss, our results must be considered as a first exploration. The estimated CO_2 emissions suggest that carbon emissions from the clay ripening process are potentially significant and that these emissions can be affected by the type of sediment and ripening conditions. Emissions from clay ripening ranged between 0.012 tonne CO_2e/tonne of clay for our short-term field experiment up to 0.05 tonne CO_2e/ tonne of clay if the desired clay quality with an organic matter content of 5% would be reached. Alternatively, if a similar amount of clay would have been collected from abroad, GHG emissions from transport alone may equal these emissions.

The results from this study offer an approach to compare GHG emissions from soft sediments to alternatives and give information on control parameters by which GHG emissions from soft sediments can be minimised. Firstly, working with saline sediment it is less likely that organic matter is converted to CO_2 instead of into the more potent greenhouse gas CH_4 than working with freshwater sediment. Secondly, gas exchange between sediment and atmosphere can be limited, minimising GHG emissions directly, and indirectly by maintaining anaerobic conditions. However, for freshwater sediments anaerobic conditions may stimulate emissions of the much stronger GHG CH_4. Minimising gas exchange works against the aim to reduce the organic carbon content in the sediment to 5%. Therefore, studies are performed with ripened to see whether saline clay with a higher than desired OM content (>5% OM) is affecting dyke building strength.

We recommend that sediment-related emissions are addressed in life cycle analysis (LCA) of hydraulic engineering projects, so that different options can be properly compared and well-informed decisions can be made. To achieve this, GHG emissions from and carbon sequestration in sediments need to be integrated into existing tools, such as the ones used by the Dutch government 'DuboCalc' or the 'CO_2 performance ladder'. Meanwhile, hydraulic engineering projects that involve soft sediments should measure and report carbon stocks and fluxes of GHGs to build up the required knowledge base.

D Review of ecosystem and sediment-related GHG emission from Hydraulic engineering activities

A review of ecosystem and sediment-related GHG emission from Hydraulic engineering activities.

The 2021 IPCC report has clearly indicated that current global warming is unprecedented in scale and speed and is induced by humanity, causing rapid changes in the atmosphere, oceans, cryosphere and biosphere. In order to limit further changes in future climate, we need to reduce our impact on our climate immediately. Within the Paris Agreement (2015) governments agreed to keep global warming at max 1.5 degrees Celsius warming. In order to do so, all net CO_2 emissions need to be limited, to at least net zero CO_2 emissions. The European Union set out to reduce CO_2 emissions by 55% in 2030 and to be carbon neutral by 2050. The Dutch government aims to reduce CO_2 emissions by 50% in 2030 and net zero in 2050. Many sectors have adopted these goals. This memo reviews current knowledge on ecosystem- and sediment-related GHG emissions from hydraulic engineering (HE) activities.

Within the water sector, climate change is considered an urgent topic. Project owners (such as governments or private developers), have the ambition to be climate neutral by 2050Current investments within the Hydraulic Engineering (HE) sector focus mostly on: using sustainable energy sources, increasing carbon efficiency, creating sustainable solutions, carbon offset/credits. In practice, this means most investments are focused on the activity that takes up the largest part of the HE carbon footprint: fleet-related GHG emissions.
In the Netherlands most tenders within the HE sector already include assessment based on the amount of carbon reduction or sustainability that can be achieved, using tools and certification via the “CO_2 prestatie ladder” and “DuBoCalc”. Internationally the Greenhouse Gas Protocol is used.

BOX 1 - Scopes in CO_2 emissions
The Greenhouse gas Protocol is the most widely internationally used standard for accounting and reporting GHG emissions. It offers an accounting tool for GHG emissions on project- and company level and categorizes GHG emissions into 3 scopes:
Scope 1 – direct emissions from owned or controlled sources (buildings, fleet, production activities, etc).
Scope 2 – indirect emission from electricity or heat that are purchased and consumed by the reporting company
Scope 3 – all other indirect emissions that occur in a company’s value chain. (i.e. business travel, purchased goods, disposal of products, investments)
A company takes at least responsibility for the emissions in scope 1 and 2. A GHG emission reduction target for scope 3 emissions is only required when a company’s scope 3 emissions are 40% or more of the total of scope 1,2and 3 emissions.
More info: www.ghgprotocol.org & http://www.sciencebasedtargets.org

Ecosystem and sediment related emissions
It is clear that fleet-related GHG emissions (scope 1 and 2,Box 1) are biggest emission factors within the HE sector. Yet, there are components within HE projects that fall outside scope 1 and 2, which are currently not considered in the footprint and have the potential to be a large source of GHG emissions: the disturbance of the natural system through HE activities and ultimately GHG emissions from the sediment that is handled.

The activities within HE projects such as dredging have impact on GHG fluxes from the natural system, but it is unclear to what extent.
. At this moment it is not known how dredging affects emissions from the affected extraction location, the deposition location, or the extracted material, let alone which dredging methodologies or deposition options are more effincient in terms of ecosystem and sediment-related emissions.

Greenhouse gas emissions from ecosystem and sediment
Next to sand, silt and clay particles, dredged sediment contains also organic matter, nutrients (like nitrogen and phosphorus) and potentially contaminants. Sediment holding organic matter has the potential to produce and release the greenhouse gases carbon dioxide (CO_2) and methane (CH_4) under certain conditions. It is well known that dredged sediment has gas production potential, as gas production is a risk when depositing soft sediment in landfills (Gebert et al. 2019; Gebert et al. 2006; Van Kessel & Van Kesteren 2002). In the past within Deltares and its predecessors much focus has been on modelling the risk of swell due to gas production, providing risk-management measures to prevent basins filled with soft-sediment from overflowing (Gebert et al. 2006; Van Kessel & Van Kesteren 2002). While this literature relates to almost two decades back, this remains the current state-of-the-art. To date much remains unknown regarding the type, magnitude and rate of greenhouse gas production and release from deposited soft-sediment, let alone how this relates to sediment properties and which measures can help reducing these emissions.

Greenhouse gas production potential
What we do know is that the quantity of organic carbon within a sediment will give an idea of the *potential* for GHG production, as it gives an indication of the maximum emission that is possible. Whether that emission will eventually materialize is subject to more debate.

Silt rather than sand is an important determinant for the GHG production potential, as much as the amount of nutrients, specifically nitrogen (ammonium, nitrate) (Gebert et al. 2019). Within shallow lakes and river sediments, nitrogen and phosphorus are good proxies for GHG production, as was shown in the BlueCAN project (Schep et al. 2020).
The quality of organic carbon, or the degradable share of organic carbon, is also relevant. Typically, older sediments contain organic carbon that is more degraded than young, freshly deposited sediment. Sediment depth correlates to sediment age, with deeper layers being older, and also there this relationship between age of organic carbon and degradability is found (Wijdeveld, 1999; Zander et al., 2020).
Furthermore, some conditions are known to limit production of certain GHGs. Sediments with a high salt (and especially sulfate) content like in marine environments, limit CH_4 production. Therefore, many freshwater sediments tend to have higher expected emissions of the strong GHG CH_4. High oxygen availability prevents formation of CH_4, but allows for oxidation of organic carbon to CO_2, causing high CO_2 fluxes. Last but not least, high iron (hydr)oxide availability is able to bind CO_2 as bicarbonate and thereby lower the CO_2 emission (Wijdeveld, 1999).
Beside the geochemical characteristics of the sediment and its environment, project specific design and operations may have a large role on influencing how much of the potential emission is released. Selecting the optimum construction method, for example capping or underwater placement instead of upland disposal, can be important to limit GHG emission, or potentially improve sequestration.

To determine the GHG production potential, degradation tests can be performed in the lab or can be estimated using models. Lab measurements involve incubation of sediment samples under aerobic and anaerobic conditions. To estimate the amount of GHG released for marine and freshwater sediments, several models have been made using sediment age, quantity of organic carbon present and degradability constants as important determinants for the GHG production potential (Gebert et al., 2006; Grasset et al., 2021; Lovelock et al., 2017; Middelburg, 1989; Wijdeveld, 1999). Previous work of Deltares has used the formula formulated by Middelburg (1989), as defined below.

$$k = a \, . \, t^{-b}$$

Where:
a = constant
b = constant
t= age of organic matter in profile (defined as t_{ini}+ t)
t_{ini} = age of organic matter at sedimentation.

Greenhouse gas emission from soft sediment land fills

The GHGs produced within soft sediment will at some point escape the sediment and reach the atmosphere. In the past, researchers at Deltares have studied how gas can escape soft sediment (Van Kessel & Van Kesteren, 2002) and when the critical threshold for GHG to escape from the sediment is reached . Gas produced will create bubbles, that can only escape the soft sediment when these bubbles have enough volume to migrate to the surface and by doing so create channels and cracks. How much gas needs to be produced for these cracks to form depends on the soft sediment characteristics. Deposited soft sediment may hold 25 up to 37% of gas prior to it escaping to the atmosphere. Within Deltares models have been built within Delcon based on formula by Middelburg et al. (1989) for organic matter decomposition in sediment. This has been applied in various projects amongst which Stryker bay, Ijsselmeer, Ketelmeer and de Slufter (WL - Delft Hydraulics 2002, Wijdeveld, 1999, 2002).

Sustainable use of dredged sediment

Over the past years sustainable use of sediment gained ground, considering dredged sediment a valuable resource. By using dredged sediment to aid in the development of ecosystems that have a high potential to capture and store carbon, the sediment can even become a carbon sink. The creation of natural islands ”Marker Wadden” in the Dutch lake Markermeer were able to start capturing carbon within 1 year in locations with reed present (carbon sequestration of 5 to 12 g CO_2 m^{-2} day $^{-1}$; (Temmink et al., 2022)).Also a mangrove forest restoration project in Indonesia (Semarang, Northern Java) led to an initial increase in carbon captured in the sediment (Veld, 2018). The Kleirijperij (mud ripener) is another example of sustainable use of dredged sediment, and showed limited GHG emissions. Overall very little organic matter was lost over the course of 3 years of maturation, indicating quite some organic carbon remains stored. Sustainable use of dredged sediment and finding best practices to keep carbon locked in the sediment, will help the HE sector to become carbon neutral by 2050. Furthermore, it offers economic potential for the carbon market, where the storage of carbon may be sold as carbon credits.

How to deal with ecosystem and sediment-related emissions in the future?

Ecosystem based carbon footprinting in in marine engineering projects has been published (Dekker et al., 2014; Fiselier et al., 2015). Currently a prototype tool is developed that assesses the circularity of inland dredging activities including effects on ecosystem emissions (Besseling et al., 2021). This is created with the objective of helping the Dutch Water Authorities to dredge sediments more circularly. Other tools exist to assess the sustainability of civil construction projects, such as “DuBoCalc” and “CO2 prestatie ladder”, both developed by Rijkswaterstaat. These tools are beginning to incorporate ecosystem and sediment-related GHG emissions, to some degree and with differences between tools.
Efforts are made to get a grasp of ecosystem and sediment-related emission associated to HE activities. The “Programmatische Aanpak Grote Wateren” (PAGW) is a program within

the Netherlands that aims to improve ecological quality of large water bodies within the Netherlands via restoration of estuarine dynamics and restoring shallow land-water transitions. The first exploration phase of the pilot study (proeftuin) for cost-effective and sustainable engineering of these shallow land-water transitions gave an important insight: next to GHG emissions from equipment used to dredge, the sediment itself and disturbance thereof may cause high GHG emissions (Raadgever et al. 2020). Based on the dominant mechanisms that are associated with GHG emissions from soft sediment, the study also identifies several measures (eg the time of year that dredging takes place) that are likely to impact the size of this emission source.

Concluding remarks

To achieve the climate goals of the EU and Dutch government quick action is required. The Hydraulic Engineering sector is participating by monitoring yearly carbon footprints that are published publicly and shifting to green energy resources. At the moment, a lot remains unknown about the impact of disturbance (i.e. dredging) of the sediment layer on the GHG flux balance. Nor do we know how to treat dredged soft sediment, so that GHG emissions are kept low or to maximize capture. Researchers need to identify

- How much GHG soft sediment can emit,
- Sediment types and locations (salt / fresh waters) with most potential for release
- How we can limit these GHG emissions and maximize capture
- How disturbance affects these emissions.

Future research into practical aspects linking GHG production potential to HE methods can include

- Execute testing in field and lab on various sediment types during different parts of the dredging process and in various meteorological conditions
- Plot results over time with regards to release and fixation (vegetation growth)
- Compare the results with other GHG emission sources during the project (context and significance)
- Compare results with other operational aspects in projects, like transport of clay towards site if Beneficial Use of dredged sediment will not be possible.

These insights will offer practical guidance towards a net-zero HE sector that includes ecosystem related emissions.

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