SCIENCEFORECAST
A digital science library

Review Article
Published: 24 Sep, 2018

SF Journal of Environmental and Earth Science

Review of Methane Emissions and Soil Carbon in Wetlands in Dry Landscapes, Macquarie Mashes, Australia

*Wali SU**

Department of Geography, Federal University, Birnin Kebbi, Nigeria

Abstract

There is growing number of emission measurements of methane (CH_4) to the atmosphere worldwide, as a result of its atmospheric chemical and radiative roles becomes prevalent. In this review progress on estimating and understanding both the magnitude of, and environmental controls on CH_4 emissions from natural wetlands in comparison with other natural landscapes surfaces. Global CH_4 emission was estimated based on time series data published by Ciais et al. [1]. Natural wetlands release 177 – 284 Tg CH_4 yr^{-1}. Estimates from rice fields, animals and wastes contribute 85-105 Tg CH_4 yr^{-1}. Fossil fuels and related emissions contribute 61-200 Tg CH_4 yr^{-1}. Geological sources, termites and freshwater release 32 - 39 Tg CH_4 yr^{-1}. Analysis of methane fluxes in inland freshwater wetlands in dry landscapes (WIDS) has shown that WIDS can emit CH_4 at similar rates to coastal saline wetlands. Similarly, freshwater wetland soils rich in organic matter have greater CH_4 production rate. The formation of CH_4 from decomposing ^{14}C-labelled by *Phragmatis australis* was also reported from some wetlands. Therefore, additional measurements and refinement of wetland CH_4 emissions and their controlling factors is required from other regions particularly in Africa, Middle East and Eastern Europe. This would allow more accurate valuations of seasonal active periods, and precise up-to-date measurements of environmental controls, landscape classification and GHG quantification.

Keywords: Greenhouse gas; Methane budget; Isotopic signature; Groundwater; Environmental controls

Introduction

Soil carbon and emissions of greenhouse gasses (GHGs) including carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) vary between soil types, vegetation communities, landscape units (e.g. rivers, wetlands, drylands), land use types, and climate zones. Understanding biophysical and chemical controls on soil carbon and GHG flux processes is critical for environmental management, for wetlands in drylands (WIDS) that act as hotspots of ecosystem services. While GHG fluxes are highly variable over landscape environment, controls on soil carbon and GHG emissions are still debated and there are few datasets to draw conclusions from, especially in relation to WIDS. The rates and characteristics of GHG emissions from WIDS have implications for the global atmospheric GHG budget and may be affected by future changes in rainfall and temperature patterns associated with climate change, ecological productivity, and human land-use.

In spite of the importance of wetland vegetation in carbon sequestration, key controls and drivers of wetland GHG fluxes are yet to be fully understood [2]. While inundation, for instance, can result in bursts of biomass production, the contributions of specific landscape surfaces such as rivers and wetlands to global atmospheric GHGs concentrations vary with landscape types, inundation frequency and other biophysical and geochemical processes [3]. For example, recent studies revealed that the historical increase in global CH_4 emissions are due largely to the activities of microbes in wetlands, rice paddies, and the guts of ruminants [4].

Net fluxes of GHGs change after inundation, for instance, when inundated soils are drained, the uptake of CO_2 by vegetation increases, but for N_2O and CH_4 fluxes, soil moisture is the major control [5]. Similarly, the rate at which oxygen is depleted depends on the ambient temperature, the availability of organic substrates for microbial respiration, and sometimes the chemical oxygen demand from reductants such as ferrous iron [2]. Rates of organic decomposition are most rapid in the presence of oxygen and slower for electron receptors such as nitrates and sulphates. Nitrogen is often the most limiting nutrient in flooded soils [2]. Therefore, oxidation seemed to the controlling subscale process for the high CH_4 emissions from wetlands common soils [6].

OPEN ACCESS

Correspondence:
Saadu Umar Wali, Department of Geography, Federal University, Birnin Kebbi, Nigeria.
Tel: *+234(0)8065555170*
E-mail: *saadu.wali@fubk.edu.ng*

Received Date: *15 Aug 2018*
Accepted Date: *20 Sep 2018*
Published Date: *24 Sep 2018*

Citation: *Wali SU. Review of Methane Emissions and Soil Carbon in Wetlands in Dry Landscapes, Macquarie Mashes, Australia. SF J Environ Earth Sci. 2018; 1(2): 1024.*

ISSN 2643-8070

Table 1: Global methane budget for the decade 2000 to 2009.

Sources	Tg CH_4 yr^{-1}
Natural wetlands (bottom-up estimates)	177 – 284
Agriculture and waste (rice, animals and waste)	85 – 105
Fossil fuel related emissions	61 - 200
Other natural emissions (geological, termites and freshwater)	32 - 39
Biomass and biofuel burning	-

Source: Ciais et al. [1].

Australian research has shown the importance of incorporating knowledge from geographical and environmental science for remediation planning for rivers and wetlands that have been impacted by human activities over long periods [7]. Many conservation strategies for floodplain wetlands, such as the Macquarie Marshes in central New South Wales, tend to prioritise water distribution and response by ecological communities to flooding [8,9] instead of critical biophysical controls of organic carbon and GHG fluxes in fluvial systems that produce, store, transform and emit GHGs. In response to water regulation and new conservation policies, the extent of inundation and responses by vegetation communities have been well documented in systems like the Macquarie Marshes [10,11], but the amount of soil carbon produced as a result of inundation as well as the GHG flux characteristics of the wetlands remains unknown.

Therefore, the knowledge gap for GHG flux changes driven by changes in inundation frequency must be addressed. We need to better understand how inundation regimes affect soil carbon production, carbon sequestration by vegetation and CH_4 emissions, as well as CO_2 fluxes affected by seasonal soil moisture removal from wetlands and floodplains [12]. It is critical, therefore, to understand the links between soil carbon and GHG flux from wetlands in an Australian and global context, as well as the environmental factors (including soil moisture, soil temperature, above ground biomass, humidity, soil temperature, pH and conductivity) that can act as major controls on soil carbon and GHG flux. The ability to extend our knowledge of the controls of soil carbon and GHG flux from WIDS that are likely to affect the global atmospheric GHG budget, is critical for future global warming mitigation and WIDS management.

GHG Flux Research for Wetlands in Drylands

Greenhouse gas flux research has been carried out in many types of landscape units, including: rivers [3,13-19], wetlands [20-26], drylands [18]; soils [27], groundwater systems [28,29] and vegetation [6,30-33]. Despite the considerable number of previous researches, GHG emissions from wetlands in drylands are still not well studied. Many wetland ecosystems are recognised as important carbon sinks; however, the capacity of different wetlands to sequester and emit GHGs is thought to be highly variable. Therefore, wetlands in drylands are necessarily be assumed to be carbon sinks and further research is required into CH_4 flux.

Wetlands in drylands may be permanent, seasonal or ephemeral and are able to support large, deep, shallow water bodies or overlay a very rich aquifer. Methane flux from dry water courses can be substantial, comparable to that from rivers, wetlands and vegetation [18]. Although drained soils may have shallow groundwater tables, their annual CO_2 release, nearly doubled compared to deep groundwater levels [5], and the average annual CH_4 emissions is about 10 times greater than from deep subsoils. Further, the seasonal pattern in the depth-integrated CH_4 production rates is strongly influenced by temperature [24]. While anaerobic decomposition continuously produces CH_4 emissions [29], soil carbon of drier patches will decompose more rapidly, there by producing more CO_2 effluxes. The three main GHGs, CO_2, CH_4 and N_2O, are stored in wetlands and as such have different global warming potentials [29].

Greenhouse gas emissions from landscapes depends on the type of GHG source, effects associated with producing organisms and cellular GHG budget [34]. Similarly, the potential of a particular wetland to sequester, transform and emit GHGs change with soil moisture regime, vegetation and soil types as well as time due to environmental conditions and ecological composition [35]. For example, GHG emission models can be used to simulate the global atmospheric GHG emissions. These models are typically based on a broad spatial scale with relatively large uncertainties in bottom up and top down flux characteristics of GHGs. Therefore, site specific studies of GHG flux are advantageous as they enable identification of specific source of a particular GHG, quantification of emission rates and understand how a particular type of GHG flux pattern vary with landscape types and other environmental factors such as soil moisture, soil temperature, vegetation and soil types. Understanding the rates of efflux from a particular type of landscape and the biophysical and geochemical processes in operation, which accelerate the transformation and emissions of GHGs over distinct landscapes create the ability to address and limit the challenges and uncertainties in GHG flux behaviour by comparing independent results obtained from different landscapes.

Some of these uncertainties and challenges associated with GHG flux research include; effects of environmental change (rainfall and temperature) on GHG flux character [12,36-38]. Effects of landscapes types - wetlands and drylands [6,39]. Effects of surface-groundwater interactions [29,40]. Effects of other environmental factors – soil moisture, temperature, vegetation types [12,13] and changes in land use pattern [32]. In addition, employing site specific GHG flux analysis can allow for comparison of GHG flux characteristics of different wetlands and drylands, resulting in more accurate and robust records of GHG flux pattern from different environments.

For example, Xiao et al.'s [41] analysis of three (3) diel field campaigns, over one month (i.e. 2010-May 2011), showed average CH_4 flux was much less than that reported from reservoirs in tropic and temperate regions. The photosynthesis of phytoplankton was the dominant control on the diel gas fluxes during alga bloom in spring and summer, while maximum monthly flux occurred in June 2010 which corresponded to the lowest water level. However, water and sediment temperature, and TOC did not have significant relationship with CH_4 fluxes. Conversely, the continuous decrease in hydrostatic pressure and the low water level was responsible for more CH_4 emission at the sediment-water during the discharging period; thus, increases the CH_4 effluxes, since the diffusion time via a reedy water column is shorter and less CH_4 may be oxidized compared with that in a long water column.

Therefore, understanding the spatial variability of soil carbon and GHG flux from wetlands particularly in arid and semi-arid regions like Australia is becoming increasingly important in the context of global climate change and the associated projected changes in rainfall and temperature. This review will contribute to the scientific basis of understanding CH_4 emissions from WIDS and has implications for similar WIDS and their management in many parts of the world (e.g. Sub-Saharan Africa, Asia and America).

Table 2: Methane studies from different ecosystems around the world.

Country	Site Name	Ecosystem type	g CH_4 m^{-2} d^{-1}
Australia	Richmond Catchment	Coastal floodplain	1.04
Australia	Mary River Catchment	Tropical floodplain	25.3E-06
Australia	North-eastern NSW	Forested wetlands	1.029
Australia	North-eastern NSW	Costal upland water bodies	0.015
Canada	Haliburton Forest	Temperate Forest	39.33
Canada	Boreas fen site	Bog	7.61
China	Sichun	Peatland	71.04
China	Zoege Plateau	Wetlands	58.8
Denmark	-	Fresh water wetland	0.08-0.345
England-UK	Tadham Moor	Wetland meadow	-0.066
Finland	Boreal Forest	Peatlands	12.72-588.00
India	Mooringanag Creek	-	0.64
Malaysia		Fresh water wetland	16.25
Mongolia	Inner Plateau	Riparian Mires	234,30
Netherland	R.O.C. Zegveld	Peat soils	0.27-0.43
Poland	Lodz	Inhabited (city)	0.66-0.24
Sweden	Stordalen, Abisko	Sub-arctic wetland	2681.2
Tanzania	Ras Dege	Coastal wetland	0.160-1.12
Thailand	Thai Rice Paddies	Freshwater wetland	2.84-8.05
USA	Prudhoe Bay, Alaska	Wet coastal tundra	0.166-0.203
USA	Barrow Alaska	Wet/moist coastal tundra	0.01-0.17

After Wali, [66].

Research Knowledge Gaps

The global GHG budget is strongly influenced by ecosystem carbon production, storage, biophysical and geochemical processes that transform and emit GHGs from different landscape surfaces [42]. While dissolved organic matter (DOM) from wetlands represents a fundamental layer in the global carbon budget [17], CO_2 flux from wetlands, streams and rivers constitutes a major component of global carbon cycle [21]. Although wetlands are typically sinks of CO_2 [12], wetlands are also dominant sources of global atmospheric CH_4 [21]. Therefore, it is quite difficult to understand the role of wetlands in the global atmospheric GHG budget, because of large disparities found in bottom-up and top-down estimates of GHG fluxes. This is particularly the case for WIDS, where inundation, soil moisture and other environmental variables such as annual rainfall and temperature are highly variable.

Soil carbon production, transformation and GHG emission in rivers of Australia indicates that riparian sources dominate carbon pools in streams and catchments, while floodplain sediments represent a substantial sink of riverine particulate organic carbon (POC) [43]. This study acknowledges the interactions between geomorphology, surface flow and river regulation as well as the temporal and climatic influence on POC production. However, the study did not explain how DOC is transformed and emitted from floodplain wetlands. Although, the characteristics of soil carbon in Australian wetlands, like in many regions of the world, vary with rainfall distribution, continental vegetation, soil moisture and temperature patterns [42], how these variabilities affect GHGs flux from fluvial systems remain largely unknown particularly in Australian inland wetlands.

The status and knowledge of Australia's wetlands has increased in recent years with greater understanding of their distribution and extent, biota and ecological conditions [7]. While considerable amounts of research were undertaken in Australian inlands freshwater wetlands including [42,44,45]. Most of these studies focussed particularly on inundation extents and response by ecological communities, instead of critical biophysical and geochemical processes that act in conjunction with inundation (or soil moisture) to affect soil carbon (SOM) and GHG flux in fluvial network systems. Generally, in Australia, there has been little input of scientific principles into investigating GHG flux characteristics from typical Australian inland fresh water wetlands.

The ecological character of many WIDS in Australia [7], has deteriorated in recent years as a consequence of water regulation. This in addition to changes in annual rainfall, result in fluctuations of standing water level. Falls in groundwater table and drying up of surface water bodies in WIDS contribute greatly to increased CO_2 fluctuations, even though CH_4 fluxes can be slightly reduced [40]. There is no doubt that groundwater levels in Australia, are heavily impacted by changes in annual rainfall and landuse, but how these changes might affect soil carbon and GHG flux, is largely un investigated from Australia's inland wetlands systems located in drylands.

Over View of Methane Emission from Different Wetlands Around the World

Atmospheric CH_4 concentrations have increased since pre-industrial times. From 1750 to 2011 the atmospheric concentration of CO_2 increased by 40%, from 278 ppm to 390.5 ppm. During this

Table 3: Environmental controls and drivers of GHG flux from wetlands.

GHG flux	*Controls and drivers*	*Methods used*	*Examples/citation*
High CH_4 emission from freshwater wetlands	Inundation/soil moisture	Chambers	Macquarie Marshes, NSW, Australia [66]
1. High CH_4 flux from forested wetlands compared to open wetlands	Productivity factor, wetland area, precipitation ratio.	A process-based model (Landsat ETM+) + chambers	North-eastern NSW, Australia [71]
2. CO_2 evasion contributed to wetland C loss, CH_4 evasion contributed to CO_2 emissions	Deoxygenation and acidification in waters	Micrometeorological techniques	Richmond River Catchment, Australia [53]
3. Seasonal CH_4 flux correlate strongly with soil temperature and organic acid concentrations	Temperature and microbial substrate availability	Static chambers (squared, 10 – 3- litres)	North-western Eurasian and Greenlandic North, Greenland [72]
4. High correlation between moisture content and reduction of CH_4 uptake rate.	Soil moisture content and temperature	Diffusion chambers (*H* = 600mm, *D* = 150mm)	Landfill, Schoten-Antwerp, Belgium [51,52]
5. Variability in GHG flux attributable to contrasting dry matter liability and soil physical properties	Soil water depth and soil properties	Drilling-based technique (Lysimetre)	Orke and Majne-gaden, Sweden [68]
Variation of CH_4 flux correlate to seasonal variation of temperature and precipitation.	Temperature and precipitation	Open-ended static chambers	Sanjian Mire, Sanjiang Plain, Hongyuan county, China [73]
6. CH_4 fluxes are 30 times higher from ponds compared to adjacent vegetated area.	Underlying peat and N_2 fixing cyanobacteria	Micrometeorological techniques	Boreal and Sub-arctic, USA [22]
CH_4 flux rates varies with planting date, straw addition. Diel variation of CH_4 flux correlates strongly with temperature.	Solar radiation, temperature and straw incorporation	Static chamber technique	Rice fields, Texas, USA [75]
7. Large hourly variation in CH_4 fluxes, no systematic diurnal variation in CH_4 fluxes. CH_4 flux was exponential to peat and temperature	Peat depth and annual temperature	Eddy covariance technique	Boreal fen, Finland [75]

same period, the atmospheric concentration of CH_4 increased by 150%, from 0.722 ppm to 1.803 ppm [1]. Global atmospheric CH_4 concentrations and estimates of atmospheric lifetime, limit total CH_4 emissions between 500 to 600 Tg per yr^{-1}. Estimates of global CH_4 emissions from wetlands ranged from 80 to 280 Tg per yr^{-1} [46]. After a decade of stability in atmospheric CH_4 concentrations, atmospheric measurements indicate that since 2007 there has been renewed annual increase in the concentration of CH_4 in the atmosphere. The drivers of this renewed growth are still debated [1]. Atmospheric CH_4 budget between 2000 -2009 is summarised in Table 1. During this period natural wetlands, Agriculture and waste dominated emissions.

Wetlands constitutes small portion of global land area (5 to 8%), but they are the largest natural source of atmospheric CH_4, with median emissions of ~164 Tg yr^{-1}, which constitutes about 1/3 of global emissions [46]. Emissions from northern high altitude wetlands [47] showed that between 1990 to 2009, ~48.7 Tg yr^{-1} is being released. While global wetlands emit large quantities of CH_4, methane emissions from wetlands in both dry and wet climates show a large temporal and spatial variation, which can partly be described by correlations with environmental variables [48]. Methane emission from landscapes surfaces have extensively being studied (Table 2), with most studies focusing on wet and peatlands.

However, there is disagreement if wetlands are actually carbon sources or sinks [49]. In addition, there is also disagreement in the interpretation of internal drivers of carbon sequestrations and emissions in wetlands (Table 3). Further, the environmental conditions in wetlands are also dependent on other external forces, like climate. For instance, Fest et al [50] compared CH_4 flux from dry and wet forests. Their study revealed that, variation in CH_4 uptake in both dry and wet conditions was strongly controlled by soil moisture. This study agrees with Boeckx, et al., [51] and Gatland et al [52]. Table 3 summarised an overview of GHG flux and their environmental controls and drivers. Methane production and consumption in both wet and dry ecosystem is a microbiological process, controlled by the energy sources required for methane producing and oxidising bacteria and archaea. There are now many extensive studies on CH_4 emissions in different landscape setting (Table1).

Nevertheless, the controls and drivers of CH_4 emission from wetland are still debated, as it is not clear which environmental factor is the most influential when they all operate simultaneously. Methane flux is highly variable in both wetlands and drylands, because CH_4 fluxes tend to be characteristic of ecosystem types and the prevailing environmental conditions (Table 2). Michael and Sabmine [23] studied groundwater levels and redox potentials of common wetland soils in a temperate-humid climate, Germany. Their study revealed high emissions of CH_4, fluxes ranged from 5-73 g $m^{-2}yr^{-1}$ and increased with inundation of wetland soils. This finding concurs with Gatland et al. [52]. Their study on CH_4 emissions in coastal wetland, showed high CH_4 fluxes occurred during inundation period. Morin et al. [12] concluded that CH_4 emissions in wetlands increased as a result of rising water level and atmospheric temperature. Diel variation of CH_4 strongly correlated with temperature, atmospheric pressure and the height of the atmospheric boundary layer [53].

Isotopic composition of methane from wetlands

The development of stable isotope techniques is one of the major breakthrough of the last century [54]. The technique is increasingly being used to solve biogeochemical problems in ecosystem [55]. For carbon stable isotope ratios $\delta^{13}C$ (‰), the calculation is defined as $\delta^{13}C = (R_s / R_{std} - 1) \times 1000$ [56]. Where R_s is the isotopic ratio $^{13}C/^{12}C$ of the sample and R_{std} the carbon stable isotope standard. Atmospheric CH_4 has a mean $\delta^{13}C$ value of around -47‰ [54]. Measurements of spatial and temporal variation in global $\delta^{13}C$ and D, showed a slight enrichment in southern hemisphere (-47.2‰) relative to northern hemisphere (-47.4‰). However, CH_4 derived from air bubbles in polar ice, up to 350 years in age, has a ^{13}C value, which is 2% lower than at present. This suggests that anthropogenic burning of fossil fuels and biomass may be the fundamental driver of the present ^{13}C enrichment in methane [54]. Table 4 summarizes global estimates of CH_4 from wetlands. Schaefer et al. [57] review of global fossil fuel CH_4 emissions based on isotope database indicates that CH_4 emissions from natural gas, oil and coal production and their usage are 20–160% greater than the reported estimates by past studies.

Historical records of $\delta^{13}C$-based source attribution for different periods, showed upward trend of emissions from fossil fuel, industries and geological sources between 0-1700 and 1985-2002 [58]. From 2003 to 2013, there were declines in CH_4 emissions from these sources. However, biomass and microbial sources showed consistent upward production from 0-1700, through 1985-2002 and from 2003-

Table 4: $\delta^{13}C$ CH_4 isotopic signature (‰) from the Macquarie Mashes.

Study sites	CH_4 flux (kg ha^{-1} d^{-1})	$\delta^{13}CCH_4$ (‰)	Summary	
Site 1	NFC	-54.6	Reed bed	
Site 2	4.81E+04	-49.2	**Units (median)**	**CH_4 flux**
Site 3	1.38E+01	-53.4	kg ha^{-1} d^{-1}	1.73E+03
Site 4	3.44E+01	-49.3	kg ha^{-1} yr^{-1}	6.31E+05
Site 5	3.44E-02	-	kg ha^{-1} $yr^{-1}$for	1.40E+06
Site 6	5.50E-02	-60.6	g ha^{-1} y^{-1}	1.40E+03
Site 7	1.38E-01	-63	Tg yr^{-1} f	4.97E-03
Site 8	-6.88E-02	-64	$\bar{x}$ $\delta^{13}C$ CH_4 (‰)	-56.3
Site 9	2.75E+04	-	-	-

After Wali, [66].

2013. In addition, two separate studies of $\delta^{13}C$ CH_4 isotopic trends for 1990-2005 arrived at different conclusions. Continuous fossil fuel emission and decreasing microbial emissions in the Northern Hemisphere were first inferred [58]. In contrast, Kirschke et al. [59] and Zang et al. [60] inferred that $\delta^{13}C$ CH_4 isotopic trends were driven by decreasing rice paddies.

Many recent studies [56,61-65] have used stable isotope analysis to measure CH_4 flux in wetlands. Isotopic signatures in Richmond River Catchment, for instance, were characteristic of temporal variability rather than spatial trend. The DIC compositions were driven by changes in groundwater flows and rainfall events [61]. During warmer months DIC pools is decreased by aquatic photosynthesis, there by enriching $\delta^{13}C$-DIC by preferentially removing the $^{12}CO_2$ isotopologue. During cooler months, $\delta^{13}C$-DIC was lower indicating that groundwater flows outweighed aquatic photosynthesis as the predominant control of the DIC pool during the drier seasons [61].

The combined atmospheric modelling environment and inventory analysis by France et al. [62] indicates air mass in the planetary boundary layer over Russia and Barents Sea, with wetlands being the likely dominant source of CH_4 in that region. Holmes et al. [63] analysis of factors influencing CH_4 and CO_2 production and oxidation pathways from 58 wetlands, showed a combination of environmental factors including soil pH, vegetation type, soil EC and latitude correlate to the dominant methanogenic pathway. However, their analysis indicates that tropical wetlands do not correlate with these factors in the same way as northern wetlands do, suggesting the effects climate change as responsible for different correlations.

However, plant mediated diffusion through aerenchyma, a process that discriminates ^{13}C CH_4, dominates CH_4 emission pathways from wetlands to the atmosphere [64]. The CH_4 emitted in the atmosphere from wetlands in subarctic tundra, was found to be lighter compared to that of surface pore water, and $\delta^{13}C$ in the emitted CH_4 correlated negatively with vascular plant cover [64]. In the same vein, Thompson et al. [56] analysis of $\delta^{13}C$ and $\delta^{2}H$ of CH_4 showed higher concentrations of CH_4 in the hypoxic deep water coincided with decreasing dissolved CO_2 concentrations. While most depleted values of $\delta^{13}C$ and ^{2}H isotopic values occurred in profundal sediments, and methanotrophic oxidation dominated in epilimnion based upon decreasing concentrations and increasing values of $\delta^{13}C$ CH_4 and $\delta^{2}H$ CH_4 [56]. Their comparison of $\delta^{13}C$ and $\delta^{2}H$ observations of CH_4, showed acetate fermentation was likely the dominant production pathway throughout the system. This finding concurs with Vaughn et al. [65]. Their study also found that stable isotope signatures of CH_4 and DIC was dominated by acetate cleavage CH_4 production in low-centred polygons of arctic polygon tundra.

Methane emission from a typical freshwater wetland

Methane emission from wetlands is critical because CH_4 has 25 times the global warming potential of carbon dioxide (CO_2), and wetlands play a critical role in global carbon cycling. Using flux chambers CH_4 emission was measured in the reed bed: Macquarie Marshes. Methane flux was strongly influenced by moisture. The reed bed (809.5 ha) has the potential to release 4.97E-03 Tg yr^{-1} of CH_4. These results demonstrate that freshwater floodplain wetlands in dry landscapes can emit CH_4, at rates comparable to coastal saline wetlands [66].

Environmental drivers of soil carbon in wetlands

A range of environmental drivers affects soil carbon and CH_4 flux from wetlands. These include groundwater levels, inundation pattern, soil temperature, vegetation types, soil EC and soil pH levels. Fluctuating water table levels for instance, may hinder the effective production of CH_4 by thwarting the microbial methanogenic consortium through the introduction of oxygen. Soil temperature is also widely considered a highly influential variable on CH_4 emission from wetlands, likely owing to the stimulation of the metabolic tolls of microbial methanogenic consortium in the soil [12]. While oxygen level affects the relative rate of CH_4 generation and oxidation, both processes are controlled by increased soil temperature [12].

Battin, et al. [13] studied the biophysical controls on organic carbon fluxes in fluvial networks. The study indicates large downstream trend of declining in-stream storage volumes and storage times of dissolved organic carbon (DOC) with increasing river or streams discharge. Because discharge shapes the channel geomorphology and bed topography through fluvial networks and is thus, likely to control rich-scale DOC storage dynamics in floodplain wetlands. For instance, flow over morphological features controls surface and sub-surface fluxes, whereas sediment characteristics control the resistance to exchange. High gradient streambeds create opportunities for subsurface retention and storage of DOC under flow conditions. In contrasts, low-gradient streams and rivers are generally depositional environments during low flows. Fine-grained sediments accumulate, clog interstitial spaces and reduce bed roughness, all of which reduce the potential for surface-subsurface fluxes, storage and retention of DOC in floodplain wetlands.

Mitsch et al.'s [67] wetlands carbon and climate change showed while wetlands provide an optimum natural environment for the sequestration of atmospheric CO_2, yet they are large emitters of CH_4. The study further illustrates that when carbon sequestration is compared to CH_4 fluxes; do not have 25 times more CO_2 sequestration than CH_4 emissions. They further illustrate (using dynamic modelling) that carbon flux from temperate and tropical wetlands, that CH_4 emissions from wetlands may become unimportant within 300 years when compared to carbon sequestration in wetlands. However, within that period, most wetlands may become both net carbon and radiative sinks. This study concurs with Birol et al (2009). Findings showed that wetlands can be both sources and sinks of carbon, depending on their age, operation and the environmental boundary conditions, e.g. climate and latitude.

Effects of groundwater levels on GHG fluxes

Fluvial water courses in drylands (often connected to groundwater) have the ability to process carbon during downstream transport and emit considerable amount of CO_2 [18]. In floodplain soils,

the groundwater level is thought to be the most important control of methane emissions [23], because the depth of wet soils controls both the depth of methane production zones as well as the thickness of the oxidation zone. In the same vein, deep groundwater levels correspond to a greater vertical extension of the zone of oxidation and subsequently lead to lower emission [23]. At very deep-water levels, no significant relationship exists between emissions and groundwater levels. Groundwater table levels may be seen as unimportant but it is not positively correlated to higher methane emissions [23]. This finding is similar as Berglund and Berglund, [68]. Variability in GHG emissions from soils are attributable to contrasting dry matter liability and soil physical properties, which regulate the water table. Although, well drained soils are CH_4 sinks, however, significant emissions of CH_4 is reported from drained peatlands [68].

The importance of inundation in floodplain wetlands

Understanding the response of a wetland ecosystem to inundation over longer timeframe is essential to evaluate how changes in surface flows can affect soil carbon control and GHG flux from wetlands. Thomas et al. [45], demonstrated the extreme variability of inundation in the Macquarie Marshes by investigating spatio-temporal patterns of radiosynthetically active radiation absorbed by the vegetation canopy. However the study does not explain how inundation relates to vegetation or biomass, nor does it show how inundation drives critical biophysical processes controlling organic carbon production and GHG flux in the wetland. Despite wetland vegetation plays a central role in carbon and nutrient dynamics [67], wetland vegetation growth is dependent on floodplain inundation frequncy. While CH_4 emissions from vegetated surfaces has been extensively studied [69], there are few reports on CH_4 emissions from freshwater wetlands even though the latter correlates with inundation [22,52]. It is also important to note that many studies are focused on flood-vegetation response modelling, which does not explain how flood-vegetation responses can be extended to understand GHG fluxes in wetlands.

Wetlands are also highly sensitive to climate fluctuations and associated changes in rainfall and temperature which play a dominant role in an ecosytem primary productivity. While ecosystem productivity is dependent on climate and inflow, the controls of GHG flux is largely a product of biogeophysical and chemical activities within an ecoystem, which can in turn be impacted by changes in temperature and soil moisture regimes. As a result of varying inudation regimes and other external controls and intrinsic processes, many wetlands are highly heterogeneous, which may be translated into variability of GHG flux.

The heterogeneous compositions of aquatic species in wetlands have evolved life history strategies primarily in response to the natural flow, and the invasion and success of exotic and introduced species in wetlands is facilitated by changes of flow regimes [36]. Ecosystem composition changes in wetlands, is normally associated with changes in biophysical and chemical processes. Establishing these relationships and at the same time quantifying GHG flux in relation to biomass production has become a complex issue in environmental science, because of the difficulties and uncertainties involved in investigating and understanding the controls of soil carbon from landscapes surfaces in relation to atmospheric chemistry [70].

In addition, the presence of vascular plants has been recognised as one of the key factors controlling the scale of GHG fluxes in wetlands because it affects the biogeochemical processes that act jointly together to produce, transport, and emit GHGs [20]. However, changes in GHG flux derived from changes in wetland vegetation alone could be inadequate to draw conclusions regarding wetland GHG flux character. Because, GHG flux can be affected by grazing animals, changes in inundation as well as temperature, which act to control the biophysical and chemical processes controlling greenhouse flux from wetlands [76-80].

Conclusion

The literature highlights the importance of understanding of environmental controls on GHG production and oxidation from different landscapes. It is clear from the literature that the general links between inundation, vegetation response, soil bacteria and archaea, soil carbon and GHG flux are broadly understood. However, more research is required to refine our understanding of when, where and how CH_4 will be produced or oxidised in inland terminal wetlands systems.

References

1. Ciais P, Sabine C, Bala G, Bopp L, Brovkin V, Canadell J, et al. Carbon and other Biogeochemical cycles. In: Climate change 2013: The physical science basis. Contribution of working group to the fifth assessment report of the intergovernmental panel on climate change. Eds: Stocker TF, Qin GK, Plattner M, Tignor SK, Allen j, Boschung A, et al. Cambridge university press, Cambridge, United Kingdom and New York, Ny, USA.
2. Mitsch JW and Gosselink GJ. Wetlands: Fourth edition, john wiley & sons, inc Hoboken, New Jersy. Canada. 2007.
3. Hall RO, Tank JL, Baker MA, Rosi-Marshall EJ and Hotchkiss ER. Metabolism, gas exchange, and carbon spiralling in rivers. *Ecosystems.* 2016; 19: 73-86.
4. Nisbet EG, Dlugokencky EJ, Manning MR, Lowry D, Fisher RE, France JL, et al. Rising atmospheric methane: 2007–2014 growth and isotopic shift. *Global Biogeochemical Cycles*. 2016; 30.
5. Von Arnold K, Nilsson M, Hånell B, Weslien P and Klemedtsson L. Fluxes of CO_2, CH_4 and N_2O from drained organic soils in deciduous forests. *Soil Biology and Biochemistry.* 2005; 37: 1059-1071.
6. Langevelda CAR, Segersa Dirksa BOM, van den Pol-van Dasselaarb A, Hensend GL, A Velthofc, A Hensend. Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands. *European Journal of Agronomy.* 1997; 7: 35–42.
7. Finlayson CM, Davis JA, Gell PA, Kingsford RT and Parton KA. The status of wetlands and the predicted effects of global climate change: The situation in Australia. *Aquatic Sciences*. 2011; 75: 73-93.
8. Ralph TJ, Hesse PP and Kobayashi T. Wandering wetlands: Spatial patterns of historical channel and floodplain change in the Ramsar-listed Macquarie Marshes, Australia. *Marine and Freshwater Research.* 2016; 67: 782.
9. Ralph TJ and Hesse PP. Downstream hydrogeomorphic changes along the Macquarie river, south eastern Australia, leading to channel breakdown and floodplain wetlands. *Geomorphology.* 2010; 118: 48-64.
10. Wang B, Chen Y and Lü C. Evaluating flood inundation impact on wetland vegetation fpar of the Macquarie Marshes, Australia. *Environmental Earth Sciences.* 2015; 74: 4989-5000.
11. Wang C, Lai DY, Sardans J, Wang W, Zeng C and Penuelas J. Factors related with CH4 and N2O emissions from a paddy field: Clues for management implications. *PLoS One.* 2017; 12: e0169254.
12. Morin TH, Bohrer G, Frasson RPdM, Naor-Azreli L, Mesi S, Stefanik KC, et al. Environmental drivers of methane fluxes from an urban temperate wetland park. Journal of Geophysical Research: *Bio geosciences.* 2014; 119: 2188-2208.

13. Battin TJ, Kaplan LA, Findlay CS, Sturart MHE, Packman AIJ, Newbold D, et al. Biophysical controls on organic carbon fluxes in fluvial networks. *Nature Geoscience.* 2008; 1: 95-100.

14. Galy V, Peucker-Ehrenbrink B and Eglinton T. Global carbon export from the terrestrial biosphere controlled by erosion. *Nature*. 2015; 521: 204-207.

15. Hotchkiss ER, Hall Jr RO, Sponseller RA, Butman D, Klaminder J, Laudon H, et al. Sources of and processes controlling CO_2 emissions change with the size of streams and rivers. *Nature Geoscience*. 2015; 8: 696-699.

16. Liikanen AH, Jari T, Karjalainen SM, Heikkinen K, Väisänen TS, Nykänen H, et al. Temporal and seasonal changes in greenhouse gas emissions from a constructed wetland purifying peat mining runoff waters. *Ecological Engineering*. 2006; 26: 241-251.

17. Spencer RGM, Butler KD and Aiken GR. Dissolved organic carbon and chromophoric dissolved organic matter properties of rivers in the USA. Journal of Geophysical Research: *Biogeoscience.* 2012; 117: 1-14.

18. von Schiller D, Marcé R, Obrador B, Gómez-Gener L, Casas-Ruiz J, Acuña V, et al. Carbon dioxide emissions from dry watercourses. *Inland Waters*. 2014; 4: 377-382.

19. Young RG, Matthaei CD and Townsend CR. Organic matter breakdown and ecosystem metabolism: Functional indicators for assessing river ecosystem health. Journal of the North American *Benthological Society*. 2008; 27: 605-625.

20. Anna J, Torben RC and Bo W. Vascular plant controls on methane emissions from northern peat forming wetlands. *Reviews*. 1999; 14: 385-388.

21. Crawford JT, Stanley EH, Spawn SA, Finlay JC, Loken LC and Striegl RG. Ebullitive methane emissions from oxygenated wetland streams. *Global Change Biology*. 2014; 20: 3408-3422.

22. David HJ, Carol AK, John WMR and Raymond HH. Flux to the atmosphere of CH_4 and from wetland ponds on the Hudson Bay Lowlands. *Journal of Geophysical Research*. 1994; 99: 495-1510.

23. Michael S and Sabmine F. Methane emissions, groundwater levels and redox potentials of common wetland soils in a temperate-humid climate. *Global Biogeochemical Cycles*. 2000; 14: 1081-1093.

24. Shoemaker J, Varner R and Schrag D. Characterization of subsurface methane production and release over 3years at a new hampshire wetland. *Geochimica et Cosmochimica Acta*. 2012; 91: 120-139.

25. Song C, Xu X, Sun X, Tian H, Sun L, Miao Y, et al. Large methane emission upon spring thaw from natural wetlands in the northern permafrost region. Environmental Research Letters. 2012; 7: 034009.

26. Ström L and Christensen TR. Below ground carbon turnover and greenhouse gas exchanges in a sub-arctic wetland. *Soil Biology and Biochemistry*. 2007; 39: 1689-1698.

27. Batson J, Noe GBN, Hupp CR, Krauss KW, Rybicki BN and Schenk ER. Soil greenhouse gas emissions and carbon budgeting in a short-hydroperiod floodplain wetland. *Bio geosciences.* 2015; 120: 77-95.

28. Gleeson J, Santos IR, Maher DT and Golsby-Smith L. Groundwater–surface water exchange in a mangrove tidal creek: Evidence from natural geochemical tracers and implications for nutrient budgets. *Marine Chemistry*. 2013; 156: 27-37.

29. Jungkunst HF, Flessa H, Scherber C and Fiedler S. Groundwater level controls CO_2, N_2O and CH_4 fluxes of three different hydromorphic soil types of a temperate forest ecosystem. *Soil Biology and Biochemistry.* 2008; 40: 2047-2054.

30. Chen CR, Xu ZH and Mathers NJ. Soil carbon pools in adjacent natural and plantation forests of subtropical Australia. *Soil Science Society of America*. 2004; 68: 282-291.

31. Frank ED, Han J, Palou-Rivera I, Elgowainy A and Wang MQ. Methane and nitrous oxide emissions affect the life-cycle analysis of algal biofuels. *Environmental Research Letters*. 2012; 7: 014030.

32. Ibell Paula T, Xu Zhihong and Blumfield Timothy J. Effects of weed control and fertilization on soil carbon and nutrient pools in an exotic pine plantation of subtropical Australia. *Journal of Soils and Sediments*. 2010; 10: 1027-1038.

33. Yao H, Wilkes A, Zhu G, Zhang H, Liu X, Dan D, et al. Stable carbon isotope as a signal index for monitoring grassland degradation. *Scientific Reports.* 2016; 6: 31399.

34. Hayes JM. Factors controlling ^{13}C contents of sedimentary organic compounds: Principles and evidence. *Marine Geology*. 1993; 113: 111-125.

35. Tiunov AV. Stable isotopes of carbon and nitrogen in soil ecological studies. *Biology Bulletin*. 2007; 34: 395-407.

36. Bunn SE and Arthington AH. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management. 2002; 30: 492-507.

37. Davidson EA, Belk E and Richard DB. Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest. *Global Change Biology*. 1998; 4: 217-227.

38. Leifeld J and Fuhrer J. The temperature response of CO_2 production from bulk soils and soil fractions is related to soil organic matter quality. Biogeochemistry. 2005; 75: 433–453.

39. Ojanen P, Minkkinen K, Alm J and Penttilä T. Soil–atmosphere CO_2, CH_4 and N_2O fluxes in boreal forestry-drained peatlands. *Forest Ecology and Management*. 2010; 260: 411-421.

40. Furukawa Y, Inubushi KA, Mochamad I and Tsuruta MH. Effect of changing groundwater levels caused by land-use changes on greenhouse gas fluxes from tropical peat lands. *Nutrient Cycling in Agroecosystems*. 2005; 71: 81-91.

41. Xiao S, Liu D, Wang Y, Yang Z and Chen W. Temporal variation of methane flux from Xiangxi Bay of the three Georges. *Scientific Reports.* 2013; 3: 2500, 1-8.

42. Bui E, Henderson B and Viergever K. Using knowledge discovery with data mining from the Australian soil resource information system database to inform soil carbon mapping in Australia. *Global Biogeochemical Cycles*. 2009; 23: n/a-n/a.

43. Robertson AI, Bunn SE, Boon PI and Walker KF. Sources, sinks and transformations od organic carbon in Australian floodplain rivers. *Marine Freshwater Research*. 1999; 813-829.

44. Kinsford RT. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology*. 2000; 109–127.

45. Thomas RF, Kingsford RT, Lu Yi and Hunter SJ. Landsat mapping of annual inundation (1979–2006) of the Macquarie Marshes in semi-arid Australia. *International Journal of Remote Sensing.* 2011; 32: 4545-4569.

46. Bridgham SD, Cadillo-Quiroz H, Keller JK and Zhuang Q. Methane emissions from wetlands: Biogeochemical, microbial, and Modelling perspectives from local to global scales. *Global Change Biology.* 2013; 19: 1325-1346.

47. Zhu X, Zhuang Q, Qin Zhangcai GM and Song L. Estimating wetland methane emissions from the northern high latitudes from 1990 to 2009 using artificial neural networks. Global *Biogeochemical Cycles*. 2013; 27: 592-604.

48. Sari J. Methane fluxes and their environmental controls in the littoral zone of boreal lakes. PhD Dissertations in Biology, University of Joensuu. 2004; 25.

49. Kayranli B, Scholz M, Mustafa A and Hedmark Å. Carbon storage and

fluxes within freshwater wetlands: A critical review. *Wetlands*. 2009; 30: 111-124.

50. Fest BJ, Najera NH, Wardlaw T, Griffith DWT, Livesley SJ and Arndt SK. Soil methane oxidation in both dry and wet temperate eucalypt forest shows a near-identical relationship with soil air-filled porosity. *Bio geosciences*. 2017; 14: 467-479.

51. Boeckx P, Cleemput OV and Villaralvo I. Methane emission from a landfill and the methane oxidising capacity of its covering soil. *Soil Biology and Biochemistry*. 1996; 28: 1397-1405DS.

52. Gatland JR, Santos IR, Maher DT, Duncan TM and Erler DV. Carbon dioxide and methane emissions from an artificially drained coastal wetland during a flood: Implications for wetland global warming potential. Journal of Geophysical Research: *Bio geosciences*. 2014; 119: 1698-1716.

53. Xu X, Fu G, Zou X, Ge C and Zhao Y. Diurnal variations of carbon dioxide, methane, and nitrous oxide fluxes from invasive spartina alterniflora dominated coastal wetland in northern Jiangsu Province. *Acta Oceanologica Sinica*. 2017; 36: 105-113.

54. Jochen H. Stable isotope geochemistry. Seventh Edition, Springer Cham Heidelberg New York Dordrecht London © Springer International Publishing Switzerland. 2015; 402.

55. Peterson BJ and Fry B. Stable isotope in ecosystem studies. *Annual Reviews of Ecology and Systematics*. 2016; 18: 293-320.

56. Thompson HA, White JR, Pratt LM and Sauer PE. Spatial variation in flux, $\delta^{13}C$ and δ2H of methane in a small arctic lake with fringing wetland in western Greenland. *Biogeochemistry*. 2016; 131: 17-33.

57. Schaefer H, Fletcher SEM, Veidt C, Lassey KR, Brailsford GW, Bromley TM, et al. A 21st-century shift from fossil-fuel to biogenic methane emissions indicated by $^{13}CH_4$. *Science.* 2016; 352: 80-84.

58. Schwietzke S, Sherwood OA, Bruhwiler LM, Miller JB, Etiope G, Dlugokencky EJ, et al. Upward revision of global fossil fuel methane emissions based on isotope database. *Nature*. 2016; 538: 88-91.

59. Kirschke S, Bousquet P, Ciais P, Saunois M, Canadell JG, Dlugokencky EJ, et al. Three decades of global methane sources and sinks. *Nature Geoscience*. 2013; 6: 813-823.

60. Zhang G, Jing MA, Yang Y, Haiyang YU, Yaping S and Xu Hua. Variations of stable carbon isotopes of CH_4 emission from three typical rice fields in china. *Pedosphere.* 2017; 27: 52-64.

61. Atkins ML, Santos IR and Maher DT. Seasonal exports and drivers of dissolved inorganic and organic carbon, carbon dioxide, methane and $\delta^{13}C$ signatures in a subtropical river network. *Science of Total Environment*. 2017; 575: 545-563.

62. France, et al. Measurements of $\delta^{13}C$ in CH_4 and using particle dispersion modelling to characterize sources of arctic methane within an air mass. *Geophysical research*. 2016; 121: 14,257-214,270.

63. Holmes M, Elizabeth C, Jeffrey P, Tfaily MM and Ogram A. CO_2 and CH_4 isotope compositions and production pathways in a tropical peatland. *Global Biogeochemical Cycles*. 2015; 29: 1-18.

64. Marushchak ME, Friborg T, Biasi C, Herbst M, Johansson T, Kiepe I, et al. Methane dynamics in the subarctic tundra: Combining stable isotope analyses, plot- and ecosystem-scale flux measurements. *Bio geosciences*. 2016; 13: 597-608.

65. Vaughn LJ, Conrad ME, Bill M and Torn MS. Isotopic insights into methane production, oxidation, and emissions in arctic polygon tundra. *Global Change Biology*. 2016; 22: 3487-3502.

66. Wali SU. Methane emissions and soil carbon in floodplain wetlands of the Macquarie Marshes. MRes Thesis, Department of Environmental Science. Macquarie University, Sydney, Australia. 2017; 71.

67. Mitsch William J, Bernal Blanca Nahlik, Amanda M, Mander Ülo Zhang Li, Anderson Christopher J, Jørgensen Sven E, et al. Wetlands, carbon, and climate change. *Landscape Ecology*. 2012; 28: 583-597.

68. Berglund Ö and Berglund K. Influence of water table level and soil properties on emissions of greenhouse gases from cultivated peat soil. *Soil Biology and Biochemistry*. 2011; 43: 923-931.

69. Rietl AJ, Nyman JA, Lindau CW and Jackson CR. Wetland methane emissions altered by vegetation disturbance: An interaction between stem clipping and nutrient enrichment. A*quatic Botany*. 2017; 136: 205-211.

70. Olsson L, Ye S, Yu X, Wei M, Krauss KW and Brix H. Factors influencing CO_4 and CH_4 emissions from coastal wetlands in the Liaohe Delta, Northeast China. *Bio geosciences*. 2015; 12: 4965-4977.

71. Akumu CE, Pathirana S, Baban S and Bucher D. Modeling methane emission from wetlands in North-eastern New South Wales, Australia using Landsat Etm+. *Remote Sensing*. 2010; 2: 1378-1399.

72. Christensen TR, Ekberg A, Ström L, Mastepanov M, Panikov N, Öquist MS, et al. Factors controlling large scale variations in methane emissions from wetlands. *Geophysical Research Letters*. 2003; 30.

73. Ding W, Cai Z and Wang D. Preliminary budget of methane emissions from natural wetlands in china. *Atmospheric Environment*. 2004; 38: 751-759.

74. Sass RL, Fisher FM, Turner FT and Jund MF. Methane emission from rice fields as influenced by solar radiation, temperature, and straw incorporation. *Global Biogeochemical Cycles*. 1991; 5: 335-350.

75. Janne R, Terhi R, Mari P, Mika A, Sami H, Juha-Pekka T, et al. Annual cycle of methane emission from a boreal fen measured by the eddy covariance technique Department of Physical Sciences, PL 68, FI-00014, University of Helsinki, Finland. 2017; 37.

76. Bass AM, O'Grady D, Leblanc M, Tweed S, Nelson PN and Bird MI. Carbon dioxide and methane emissions from a wet-dry tropical floodplain in Northern Australia. *Wetlands*. 2014; 34: 619-627.

77. Call M, Maher DT, Santos IR, Ruiz-Halpern S, Mangion P, Sanders CJ, et al. Spatial and temporal variability of carbon dioxide and methane fluxes over semi-diurnal and spring–neap–spring timescales in a mangrove creek. *Geochimica et Cosmochimica Acta.* 2015; 150: 211-225.

78. Ginty EM. Carbon isotopic evidence that coal derived methane is altering the chemistry of the global atmosphere. Thesis: Submitted in partial fulfilment of the requirements for the degree of Bachelor of Environmental Science (Hons), School of Biological, Earth and Environmental Sciences, Faculty of Science, The University of New South Wales Australia. 2016.

79. Jackie Batson, Gregory B Noe, Cliff R Hupp, Ken W Krauss, Nancy B Rybicki and Edward R Schenk. Soil greenhouse gas emissions and carbon budgeting in a short-hydroperiod floodplain wetland. *Bio geosciences*. 2014; 120: 77-95.

80. Mikaloff F, E Sara Tans PP, Bruhwiler LM, Miller JB and Heimann M. CH_4 sources estimated from atmospheric observations of ch4and its $^{13}C/^{12}C$ isotopic ratios: 1. Inverse modeling of source processes. *Global Biogeochemical Cycles*. 2004; 18: 1-17.