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WETLANDS AND GREENHOUSE GAS FLUXES

Causes and Effects of Climate Change – A Meta-Analysis

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Thesis Completed in Partial Fulfillment of a Bachelor of Arts in Environmental Analysis, 2013-2014 Academic Year, Pomona College, Claremont, California

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ABSTRACT

Climate change is one of the largest problems facing this generation. Anthropogenically caused increases of greenhouse gas emissions is a significant culprit to this problem. Although the obvious problems such as cars, industry, and urbanism garnish a significant amount of the criticism, natural sources such as wetlands are also beginning to contribute to this issue. This is becoming increasingly significant as wetlands shift from being sinks of greenhouse gases to becoming sources as various anthropogenic impacts, including global warming itself, begin to affect the health of the wetlands. The aim of this project is to look at four common types of wetlands, being tropical mangroves, temperate coastal marshes, inland meadows, and subarctic peatlands, all located in different climactic areas of the world, and by doing a meta-analysis of available data of greenhouse gas production for each wetland type, observe how differences in their greenhouse gas production may contribute or be affected by climate change and global warming. Results of the meta-analysis revealed that the most significant production of the potent greenhouse gas nitrous oxide occurs in coastal wetlands such as tropical mangroves and coastal marshes, while the greenhouse gas methane is seen to be produced most in subarctic peatlands. These contributions of wetlands to global greenhouse gas production are not as significant as other anthropogenic contributions. However, subarctic wetlands contribute to more than half of the global methane emissions, and the most important aspect of wetland greenhouse gas production is that they are producing more greenhouse gases than they would normally be sequestering, contributing more than the basic greenhouse gas production data can display. Global climate changes such as temperature increase and sea level rise could also make these levels of greenhouse gas production become worse, although measures to decrease the effects of this such as regulations on anthropogenic nitrogen input, macrophyte presence, and prevention of peat burning.

INTRODUCTION

Recent changes in atmospheric concentrations of greenhouse gases (GHGs) are widely suggested to be responsible for Earth's global warming and the climate change crisis. Due to their ability to both absorb and release infrared radiation, GHGs are trapping the sun's warmth and keeping it in the Earth's atmosphere, essentially creating the "greenhouse effect" and acting as drivers of anthropogenic climate change (Pachauri and Reisinger, 2007). Although GHGs are necessary for maintaining Earth's atmosphere and habitable climate, the increasing levels associated with post-industrial society are contributing to the trend of higher temperatures and the breakdown of the "ozone layer" (Solomon *et al.*, 2007). The most dangerous GHGs in terms of global warming potentials observed in our atmosphere are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), with some other common contributors being water vapor and the anthropogenically produced hydrofluorocarbons and perfluorocarbons (Pachauri and Reisinger, 2007). Although carbon dioxide (CO_2) is often the GHG given the most attention in popular media due to the contributions of human activity to its rising levels, methane (CH_4) has about 25 times the potential for global warming per molecule and nitrous oxide (N_2O) is about 300 as dangerous per molecule than CO_2 (Solomon *et al.*, 2007; Ravishankara *et al.*, 2009).

This concept of "potential" for global warming is known as radiative forcing, a quantitative measure of the difference between radiant energy received by a molecule and the amount of that energy radiated back to space, which can also be affected by the residence time a given gas can have in the atmosphere (Solomon *et al.*, 2007). The higher this value is, the more incoming energy there is and the warmer the system becomes. However, since CO_2 is present in such larger concentrations in the atmosphere at about 400 ppm, while CH_4 and N_2O are at concentrations of around 2 and 0.3 ppm, respectively, CO_2 actually contributes to about 50% of the greenhouse effect, while CH_4 and N_2O contribute about 15% each (Moore, 2001). Based on these statistics, it is necessary that all forms of GHG emissions be mitigated, as all excess production of GHGs has the greatest potential for impact in changing atmospheric concentrations is the most concerning to the state of our global climate.

My interest in the topic of wetland GHG fluxes and global warming developed this past summer, working in the lab of Dr. Serena Moseman-Valtierra at the University of Rhode Island.

Her lab specializes in researching the different causes and factors that contribute to the effect of increased GHG fluxes, or outward flow of gases, from coastal marshes in Narragansett Bay, Rhode Island. These factors are hypothesized to increase nitrogen input resulting from anthropogenic activity in Narragansett Bay and higher temperatures resulting from global warming. These elevated levels of nitrogen from fertilizer and sewage run-off into estuaries and bays are relevant to GHG research because of the observed effects of nitrogen spikes on the increase of GHG emissions in coastal ecosystems (Moseman-Valtierra *et al.*, 2011).

Although this is a topic that has been more frequently studied in other types of wetlands (Zhang *et al.*, 2007; Song *et al.*, 2006), studies on nitrogen addition in coastal wetlands and salt marshes are relatively lacking. However, the effects of nitrogen addition to wetland ecosystems on the levels of N_2O fluxes from the area have been observed thus far in Plum Island Estuary, Massachusetts (Moseman-Valtierra *et al.*, 2011) and Narragansett Bay, Rhode Island (Davis *et al.*, 2004). This is an anomaly considering the previous established roles of these wetlands as GHG sinks, or areas that are able to remove GHGs from the atmosphere, rather than sources, areas that contribute GHGs to the atmosphere (Moseman-Valtierra *et al.*, 2011; Davis *et al.*, 2004). A major consequence of these coastal ecosystems becoming sources rather than sinks of GHGs is that these increased GHG fluxes are able to offset between one-third and half of the rates of carbon sequestration that normally occur in the wetlands (Moseman-Valtierra *et al.*, 2011). Similar problems with increased N_2O fluxes have also been seen in other nutrient loaded coastal areas in the United Kingdom, China, and Australia (Barnes and Upstill-Goddard, 2011; Zhang *et al.*, 2010; Allen *et al.*, 2011; Kreuzwieser *et al.*, 2003). However, there is a significant gap in the research as to whether GHG fluxes are bound to continually increase given anthropogenic nutrient loading and global warming, or if these wetlands have a near limitless capacity for GHG sequestration or storage.

The specific work I participated in with doctoral candidate Melanie Gárate focused on one potential contributor to the observed increases in N_2O emissions seen in coastal ecosystems, the common mussel (*Mytulis edulis*). By observing production rates of N_2O from mesocosms containing *Mytulis edulis* exposed to variations in temperature and nitrogen exposure, the conclusion I came to from my research with Dr. Moseman-Valtierra's lab was that increased nitrogen input and increased ocean temperatures are associated with significant increases in production rates of N_2O from the mussels, with greater increases being seen due to elevated temperature than elevated nitrogen levels (Figure 1). This is significant in showing that there is potentially a positive feedback mechanism in regards to *M. edulis*, in which warmer ocean temperatures caused by global warming may in turn cause an increased amount of N_2O production, which only exacerbates the potential for global warming further.

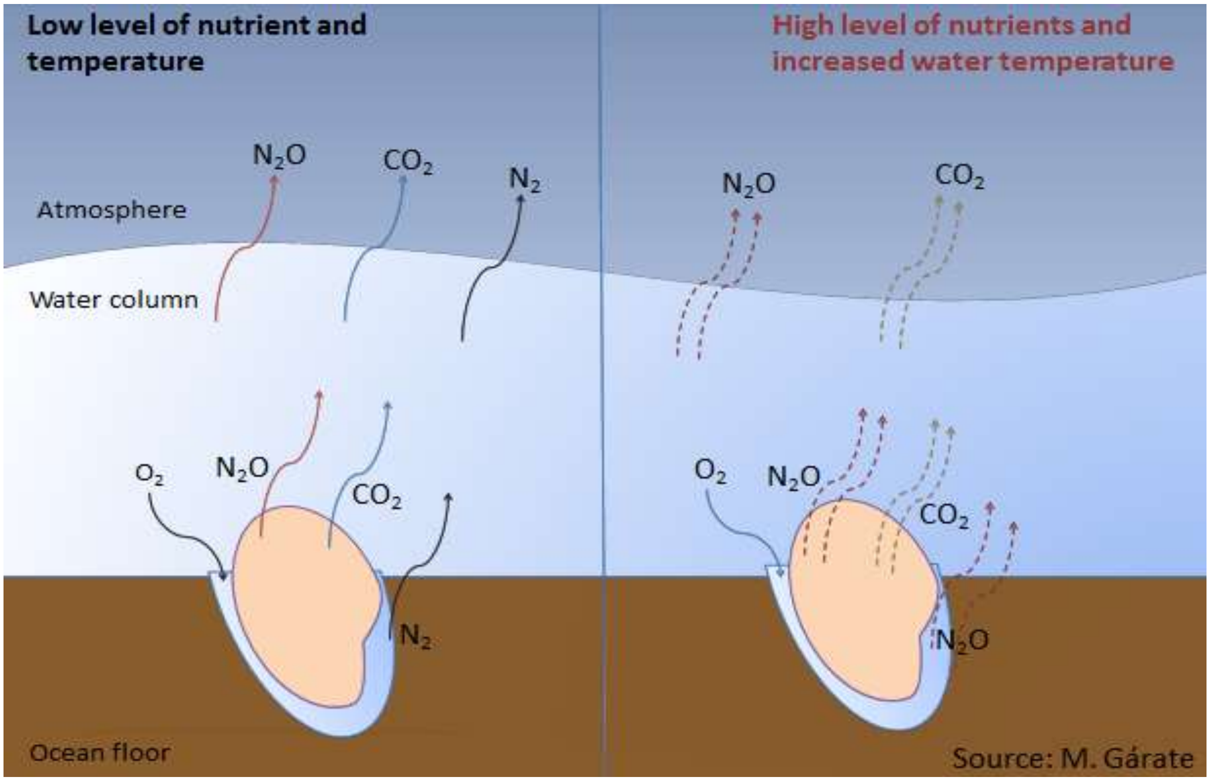

Figure 1. Effects of temperature and nutrient increase on production of GHGs by mussels.

My original plan for this current project was to use my knowledge of Rhode Island coastal marshes and N_2O fluxes and apply this knowledge to compare them to various GHG fluxes in Southern California coastal marshes, including the fluxes of CO_2 and CH_4 in addition to N_2O fluxes. However, due to significantly less research being done on GHG flux on Southern California coastal marshes as compared to New England coastal marshes, and a lack of academic researchers to collaborate with on field research and data collection, the scope of my inquiry had to be greatly expanded. Instead of looking at GHG fluxes from specifically coastal marshes in Southern California, my research purpose will be to compare multiple sources of wetland GHG flux data from four biomes, each containing a different type of wetland: tropical coasts and mangrove swamp wetlands; temperate coasts and salt marsh wetlands; temperate inland areas and non-tidal meadow wetlands; and subarctic inland areas and subarctic peatland wetlands. The reason that mostly wetlands found in temperate biomes will be analyzed is that most of the world's wetlands are located in these biomes, and the most data is available for coastal marshes located in the Northern Hemisphere, where the largest amount of researchers are investigating GHG fluxes of wetlands. In addition, wetlands in Northern arctic regions are the most at risk in terms of global warming, as the rapidly increasing temperatures in those areas are having major effects on the health of the ecosystems (Woo and Young, 2006). This is compared to wetlands found further south, such as arid wetlands, for which seemingly zero data exists on GHG fluxes, as the more pressing issue in those wetlands is them not drying out and affecting the general flora and fauna of the ecosystem.

In order to conduct this research, I will be using techniques of data meta-analysis, which is a specialized data analysis technique that compiles many sources of primary data in order to incorporate the many findings and come to a field-wide conclusion (Koricheva *et al.*, 2013). By gathering GHG data from varying research projects on these different types of wetlands and using the techniques of meta-analysis to equate and compare these findings, I hope to build a comparison of the differences between the wetlands of each biome, including explanations of why these differences exist and how they affect the storage residence times as well as the flux of GHG in each region. This information will then be placed in context of a global GHG significance scale in order to illustrate the importance of wetland GHG storage and flux in terms of global warming and the general climate crisis, and how further effects to the climate could possibly affect these delicate ecosystems.

WETLANDS: DEFINITIONS AND ORIGINS

A wetland is an ecological area known for its combination of aquatic and terrestrial, and this feature can vary a fair amount between Earth's different climatic regions. Given the issue of defining an area that is seen in varying degrees and intensities wherever water is found in proximity to land, it is not easy to come up with an all-encompassing definition for a wetland (Majumdar *et al.*, 1998). Especially due to the wide reaching significance of wetlands to a large amount of academic and professional fields, from all types of hard science to politics, law, and even economics, coming to a consensus on a universal definition of wetlands is a difficult task (Mitsch and Gosselink, 2007). A few commonalities appear in most definitions, especially those most importantly used in the scientific fields. These definitions are most relevant to this research focusing on biogeochemistry, and will be concentrated on in this project.

The essential definition of a wetland is as an intermediate area between ecosystems with chiefly land and chiefly water (Majumdar *et al.*, 1998; Mitsch and Gosselink, 2007; Van der Valk, Arnold G, 2012). The intersections of these environments leads to three main features found in all wetlands, which are used to define all wetlands concerned in scientific studies. These include consistently saturated soils (usually in shallow water), anaerobic soils, and distinct plant and animal life that are exclusive to the defined areas (Majumdar *et al.*, 1998; Van der Valk, Arnold G, 2012). The conventional level of water that is acceptable for defined wetlands is shallow enough to allow the rooting of flora over a large amount of its area, which sets it apart from a lake or a pond (Lewis, 2001). In addition to the amount of water, the flora and fauna that are found in wetlands have adapted specifically to thrive in these conditions and become associated with the definitions (Moore, 2001). The importance of the constant presence of water in these ecosystems cannot be stressed enough, owing to its ability to act as a solvent for all the nutrients, chemicals, and gases vital to the plant, animals, and gas flux and storage occurring there (Moore, 2001; Lewis, 2001). The flow and circulation of water in wetlands (hydrology) is also vital to the quality of the soil in terms of nutrient presence and saturation, which is the base for the survival of primary producers such as plants, microbes, and algae (Lewis, 2001). Not all wetlands have generally flowing water however; the water can be static and composed of any mixture of fresh and saline components (Mitsch and Gosselink, 2007). These differences in water salinity will become increasingly important when the specific comparison of tidal versus non-tidal wetlands is made, as well as serving as a basis for comparison of different types of wetlands in general.

CHEMICAL AND NUTRIENT CYCLING IN WETLANDS

The most relevant feature of wetlands to this research is their ability to sequester and store GHGs under usual and healthy conditions, meaning in ideal settings lacking anthropogenic influences such as nitrogen inputs, pollutants and other abnormalities such as climate change and global warming. One of the hypothesized contributors to rapidly rising carbon dioxide and general GHG levels is the continued loss of global wetlands; some of the largest carbon and nutrient sinks in the world, globally storing upwards of 44.6 teragrams per year (Moore, 2001; Chmura *et al.*, 2003). The delicate balance between the GHG sequestration abilities of wetlands, meaning the difference between a wetland being a sink of GHGs or being a source of GHGs, is easily compromised, and human negligence is often significant enough to tip this equilibrium in a detrimental direction (Moore, 2001).

The primary mechanisms in which wetlands work to soak up carbon and nitrogen, along with sulfur and phosphorous, are through various nutrient cycles. It is chiefly the flora, or macrophytes, that are rooted in wetlands which are responsible for absorbing the excess nutrients through saturated and anaerobic soils and releasing them as gases through organic processes (Van der Valk, Arnold G, 2012). These flora vary a good amount between the different types of wetlands, and these differences may also be a large factor in GHG flux and storage differences (Mitsch and Gosselink, 2007; Moseman-Valtierra,). Bacteria that rely on anaerobic processes to fix and convert the nutrients play a large role in these cycles as well, as they reside in the soil, or on and inside of plant tissues, and work with the macrophytes in order to drive the cycles (Van der Valk, Arnold G, 2012). In turn, macrophytes are the major players in wetland productivity, comprising the main physical structure of the wetlands, converting the most sunlight to organic matter, and participating in storage and direct cycling of the critical nutrients (Mitsch and Gosselink, 2007; Van der Valk, Arnold G, 2012). Since there is a great deal that is understood about the nutrient cycling processes that occur in wetlands, this knowledge can be used to explain how increases in GHG flux can occur in wetlands and supplement what is known about wetland GHG fluxes.

The nutrient cycle that is most significant to wetlands' biogeochemistry due to its contributions to biomass and interconnections with other nutrients and cycles is the carbon cycle;

a fairly intricate system that involves processes in the aerobic water column and upper sediment layer of the wetlands as well as the deeper anaerobic sediment (Moore, 2001; Mitsch and Gosselink, 2007; Van der Valk, Arnold G, 2012). The carbon cycle is chiefly responsible for converting atmospheric CO_2 to organic matter available for the plants and other lower level organisms present in the wetlands, a process called primary production (Mitsch and Gosselink, 2007). Primary production essentially moves carbon from the atmosphere into the wetlands, with at least some of the carbon going back into the atmosphere upon decay, as can be seen by the movement of CO2 via the plants and the wetland soils in Figure 1 (Moore, 2001). This is done via the well-known processes of photosynthesis, completed by the macrophytes and algae of wetlands. Primary production is often able to sequester more carbon at greater rates than respiration by other organisms in the wetland is able to remove it (via conversion it back to CO_2), which makes many wetlands net carbon sinks under normal conditions (Van der Valk, Arnold G, 2012; Figure 2). The main reason that wetlands are so efficient at performing this process is due to the high rates of primary productivity coupled with the fact that the anaerobic soils experience lower rates of decomposition than terrestrial habitats would undergo, due to a lack of oxygen and generally stable environmental conditions (Van der Valk, Arnold G, 2012; Callaway *et al.*, 2013). Judging thresholds for how to define a wetland as either a source or a sink of GHGs is challenging however, as although methods for measuring gas fluxes are usually fairly accurate, plenty of variables from climate to light levels and nutrient presence can affect the data. In addition, these gas fluxes are measured over the small areas that can be sampled for gas content, and there can be a high degree of estimation involved in order to determine what the GHG flux levels are for an entire wetland (Van der Valk, Arnold G, 2012). This concept is a significant limitation in establishing a completely accurate comparison between the many studies of different wetlands, and is an issue that will be discussed in further detail in the methods. Processes such as the burning of carbon-rich peat from wetlands and the land reclamation, or the draining of wetlands, work against the carbon cycle, allowing stored carbon to volatize back to atmospheric CO_2 (Moore, 2001).

Anaerobic carbon cycling takes place almost exclusively among the microbes and bacteria present in the water-saturated sediment that defines wetlands. In fact, many of the features that make wetland soils unique, such as their carbon sequestration ability and GHG production, are a result of anaerobic respiration. Much of the organic matter created from the aerobic processes are later used by the anaerobic processes to drive respiration and the nitrogen and sulfur cycles (Van der Valk, Arnold G, 2012). An example of this is seen by the respiratory processes of fermentation and glycolysis, undergone by anaerobic bacteria in the soil in order to produce small carbon chain compounds from larger carbohydrates, allowing for the formation of dissolved organic carbon, or DOC (Mitsch and Gosselink, 2007; Figure 2). Organisms such as sulfur-reducing bacteria and methane oxidizers known as methanotropic bacteria also contribute to DOC and the production of CO_2 (Mitsch and Gosselink, 2007). The carbon cycle begins to intersect with other aspects of GHG production with DOC, as it is highly important for all other anaerobic processes that take place in the soils of wetlands, including methanogenesis, or the creation of the GHG methane (CH_4) (Figure 3).

Methanogenesis is undertaken by the bacteria methanogens, which are exceedingly sensitive to free oxygen concentration. The more anaerobic the soil in a wetland is, the more the anaerobic process of methanogenesis outweighs CH_4 oxidation (Moseman-Valtierra, ; Fenchel and Blackburn, 1979; Morris and Whiting, 1986), which can continue to function in the presence of oxygen and produce CO_2 (Mitsch and Gosselink, 2007; Figure 3). The trade-off between CH_4 and CO_2 production involving methane is largely unstudied, however, and CH_4, despite being known to be a more potent GHG in terms of global warming potential, does not contribute as much to greenhouse warming due to generally lower concentrations (Moore, 2001). In addition, it is well-documented that CH_4 production increases in summer months and warmer climates, as well as with increased CO_2 concentrations, which means that a climate change crisis of higher temperature and more atmospheric CO_2 is causing a positive feedback, creating more GHG in CH_4 and contributing to the changing of wetlands from sinks to sources of GHGs (Moore, 2001; Mitsch and Gosselink, 2007). As mentioned above, sulfur and/or sulfate-reducing bacteria also undergo respiration similar to the methane oxidizers, and utilize DOC to reduce sulfur (Mitsch and Gosselink, 2007). This process occurs at much greater rates in saltwater wetlands, accounting for about half of all CO_2 production there, as sulfur-reducing bacteria are outcompeting methanogens for use of the organic matter (Moseman-Valtierra, ; Fenchel and Blackburn, 1979; Morris and Whiting, 1986). This stands much in contrast to freshwater wetlands, where it is the methanogens and methane oxidizer which contribute to about half of the carbon flux (via CH_4 and CO_2) (Mitsch and Gosselink, 2007).

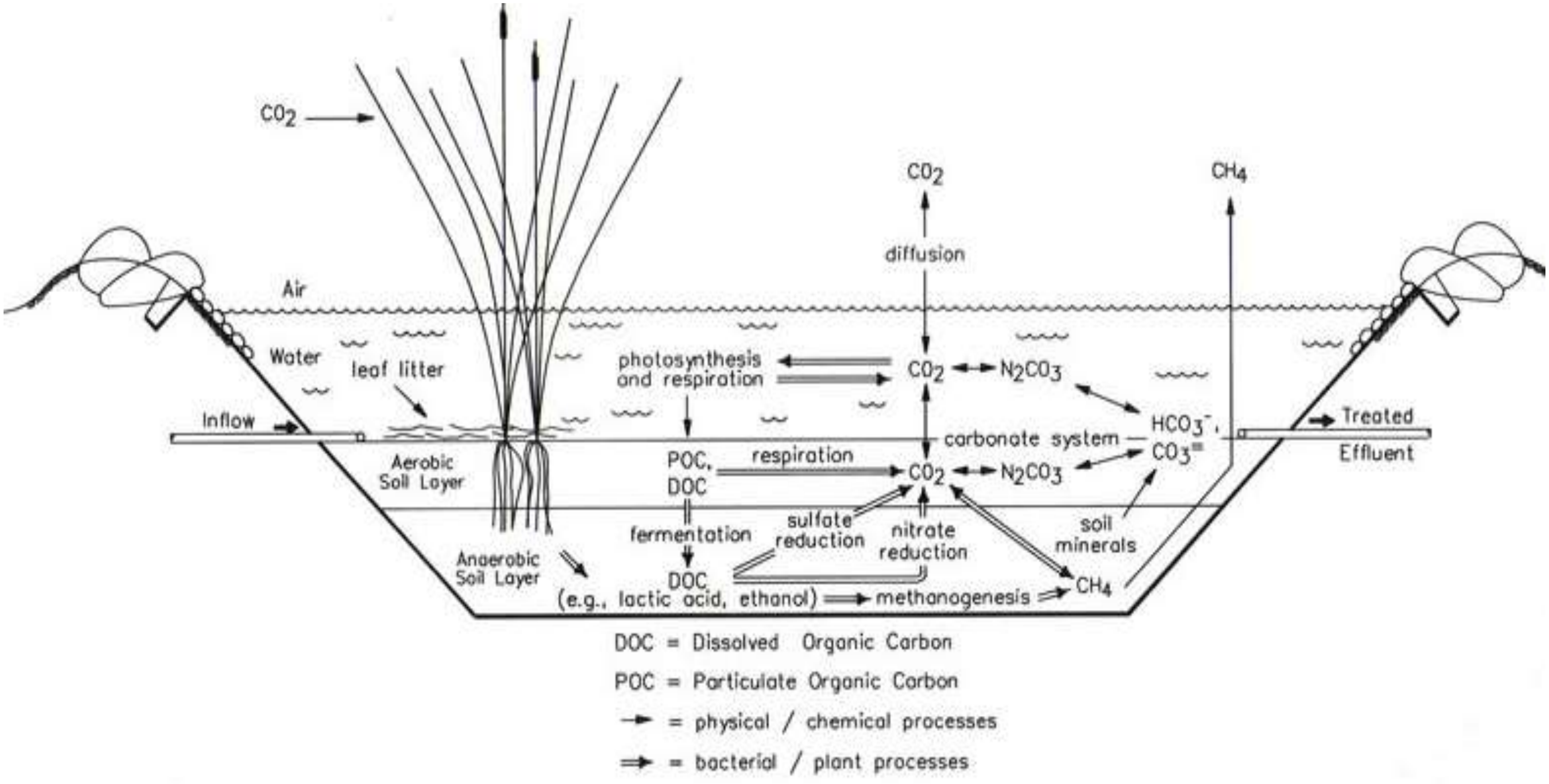

Figure 2. Taken from Campbell and Ogden, 1999, Figure 3.1. A schematic of the carbon cycle in wetlands, including the source of carbon (CO_2 intake used in photosynthesis by macrophytes), the creation of dissolved organic carbon (DOC) for methanogenesis, nitrate reduction, and sulfate reduction, and how these can produce CO_2 and CH_4.

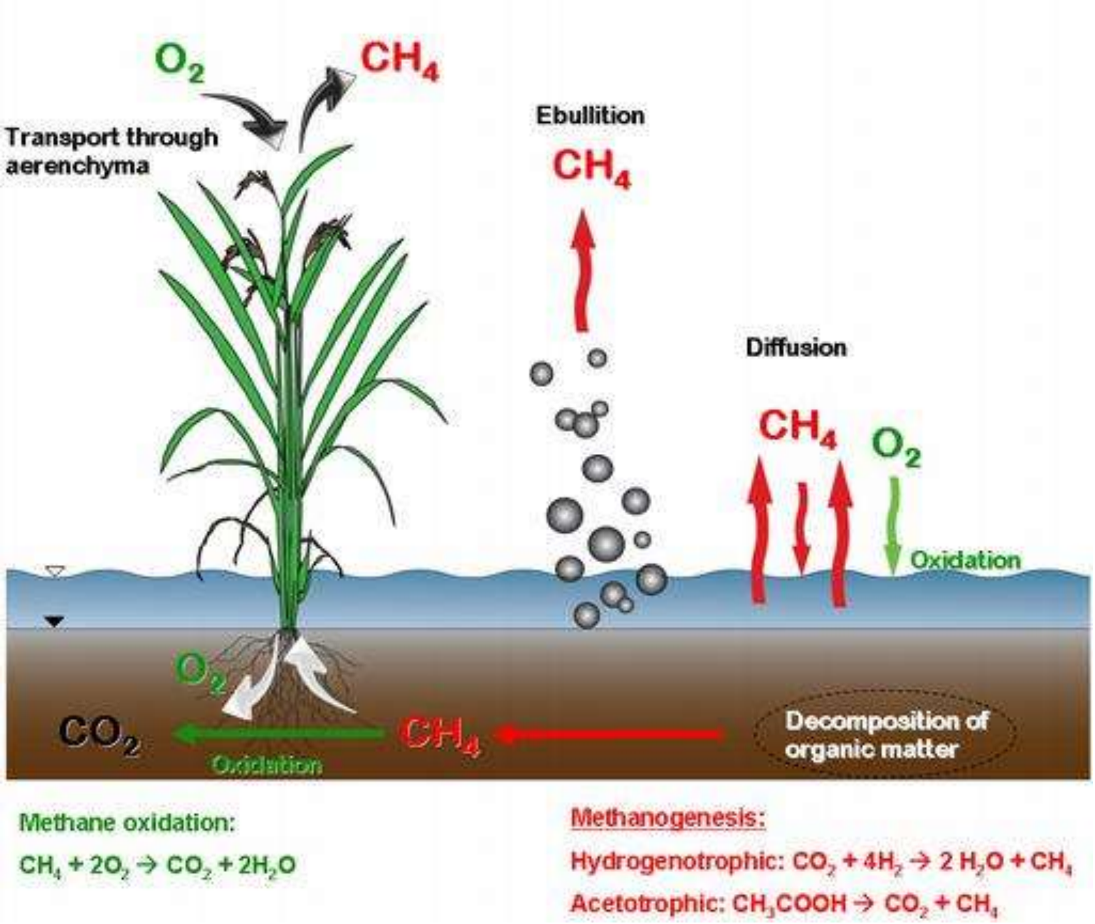

Figure 3. Taken from ETH Zurich. A diagram illustrating the process of methane production in wetlands, and the three ways that methane is produced by wetlands, which are ebullition, or release of large bubbles of methane from the soil, and diffusion, the slow gradual release of CH_4 from the soil. It is also shown at the bottom of the figure how CH_4 is oxidized to CO_2, which determines the tradeoff between CH_4 and CO_2 flux from wetlands

The last nutrient cycle essential to the creation of the last GHG not yet mentioned, N_2O, is the nitrogen cycle, which chiefly involves the transformation of nitrogen from nitrate (NO_3^-) to nitrogen gas (N_2). Although the carbon cycle plays a larger part relative to the other wetland GHG fluxes of CO_2 and CH_4, the nitrogen cycle is known to be one of most important processes taking place in swamps and marshes, as seen by the extensive research being done by the Moseman-Valtierra lab concentration on the nitrogen cycle and N_2O fluxes. This is because nitrogen is most often a limiting nutrient in saturated soils, and wetlands are the major contributors in the world scheme of transporting nitrogen from the water and earth into the atmosphere (Mitsch and Gosselink, 2007). This has become especially important in recent years, where increased fertilizer use and manufacture has greatly increased the amount of nitrogen that flows from rivers into the ocean, causing regions of eutrophication, or essentially over-fertilization that stimulates excessive and unhealthy blooms of algae to form, as well as the potential increase in N_2O flux discussed in the introduction (Moseman-Valtierra *et al.*, 2011; Diaz and Rosenberg, 2008). The nitrogen cycle "begins" with organic N if it is present from the soil or from anthropogenic input, and a process called ammonification, or nitrogen mineralization, which converts organic N to ammonium (NH_4^+) under either aerobic or anaerobic conditions (Van der Valk, Arnold G, 2012). With the presence of NH_4^+, a process called nitrification can begin, which converts NH_4^+ to the nitrate ion (NO_3^-). This can only occur in aerobic conditions, which is why the layer of aerobic sediment at the top of wetlands is important. Typically NH_4^+ cannot build up for long in the anaerobic layers (Van der Valk, Arnold G, 2012) without diffusing up to the aerobic layers to be nitrified (Mitsch and Gosselink, 2007). Once NO_3^- is present, it can take part in denitrification, which converts NO_3^- to N_2 gas in anaerobic sediment. This is one of the most important steps of the nitrogen cycle in terms of GHG fluxes of the wetland, as in situations of excess anthropogenic nitrogen input, it is more likely that the oxygen levels will not sufficient for the anaerobic conditions necessary to finalize the creation of the usual final product of N_2 from N_2O, and therefore more likely that N_2O is produced (Moseman-Valtierra *et al.*, 2011; Mitsch and Gosselink, 2007; Baumann *et al.*, 1996; Robertson *et al.*, 1988) (Figure 4). These processes are seen relative to the wetland and each other in Figure 4. N_2 gas can also undergo nitrogen fixation, which converts the gas to NH_4^- in anaerobic or aerobic bacteria and cyanobacteria, providing them with food and energy (Van der Valk, Arnold G, 2012). NH_4^- can then reenter the cycle, but more importantly, the energy the process provides to the bacteria is important for the health of the wetland soil (Van der Valk, Arnold G, 2012).

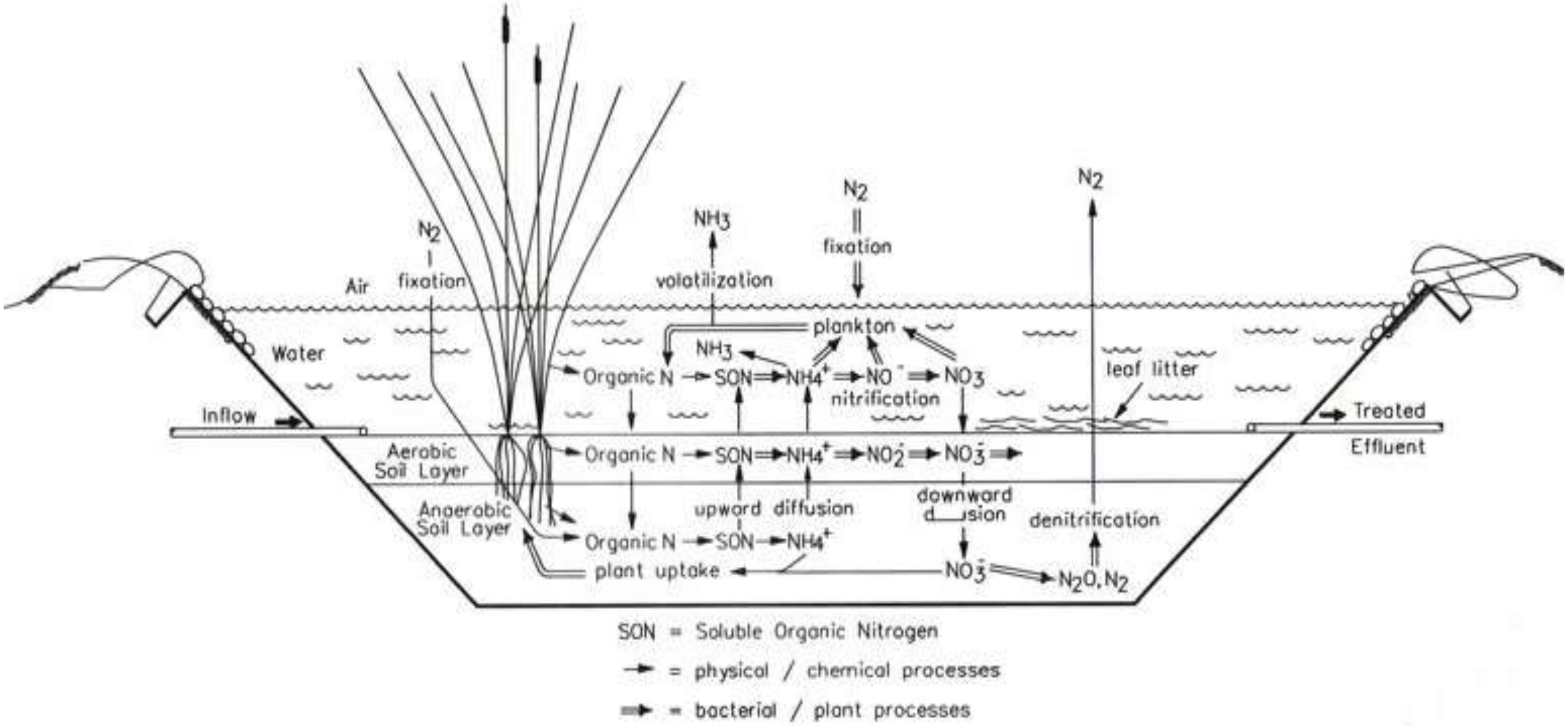

***Figure 3.2** Nitrogen cycle in wetlands.*

Figure 4. Taken from Campbell and Ogden, 1999, Figure 3.2. A schematic of the nitrogen cycle in wetlands, including the source of nitrogen (N_2 and organic N intake used in fixation by bacteria), the creation of NH_4^+ and NO_3^- via ammonification and nitrification, and how denitrification can produce the GHG N_2O and N_2.

FEATURES AND DESCRIPTIONS OF DIFFERENT WETLAND TYPES

With the basic biogeochemical workings of a wetland now thoroughly defined, the fundamental differences between wetland types that contribute to the variations in these cycles can be examined. The types of wetlands that will be analyzed in this project will be mangrove swamps, salt marshes, non-tidal meadows, and subarctic peatlands. Given the differences in GHG data availability for different types of wetlands, the wetlands outlined here are those that are most extensively studied for GHG flux and storage.

There are not a large amount of similarities between various types of wetlands, with essentially the only shared features being saturated, anaerobic soils, GHG production and storage, and distinct plant and animal life exclusive to wetlands. These differences can be analyzed to determine why differences in GHG flux levels vary between different types of wetlands. The differences between wetland types can begin with the flora and fauna, as that is one of the most striking differences that can be immediately observed. However, everything from the climate to the type of soil and the amount of water exchange and salinity factors into the variances between wetland types as well. These factors most often vary based on regional qualities, and climate, types of soil (due to local geology), and water exchange (based on proximity to the coastline) are all largely determined by the location of the wetland in the world (Lewis, 1995).

The first example of this can be seen in tropical coastal swamps, one of the wetland types that will be featured in this project. These swamps (wetlands dominated primarily by trees or shrubs (Mitsch and Gosselink, 2007)) are primarily characterized by the presence of mangrove trees, coral reef and seagrass macrophyte systems, and phytoplankton communities (Twilley *et al.*, 1992). In addition to these ecosystem features of tropical mangrove wetlands, five rudimentary necessities to defining an area as this type of wetland includes tropical climate, soft mud composed of fine silt and clay sediment abundant in organic matter, shores that lack excessively strong wave or tidal activity, a large gently sloping inclined area that can be extensively covered by tidal activity, and in a lesser way, salt water (Moore, 2001; Walsh, 1974). Mangrove trees are facultative halophytes that do not necessarily need salt water to survive, but are able to survive in brackish water where most flora cannot (Walsh, 1974).

High tides usually cover the sediment with about 15 cm to 1 m of water, meaning the water table changes with the tides, and the many different species of mangrove trees and shrubs have adapted to live in the brackish water associated with coastal areas (Moore, 2001; Mitsch and Gosselink, 2007). In fact, the characteristic long stilted roots of the mangrove trees have evolved as such in order to permit oxygen exchange in the constantly saturated environment of a wetland, as well as being able to bind and trap sediments to allow for further establishment of other flora and provide shelter and protection for the fauna that live in the mangroves (Moore, 2001). Mangrove swamps are estimated to cover upwards of 60 to 75% of all tropical coastlines, and though the size and extensiveness of these wetlands can vary, this majority of coastline coverage speaks to the significance that these habitats have for tropical coastlines all over the world (McGill, 1958).

Figure 5. Mangroves in West Bali National Park, Indonesia. The mangrove trees are seen to take root in the intertidal muddy sediment on the coastline.
Source: http://www.flickr.com/photos/ronrad/4081784387/

As for the other form of tidal wetland, the salt marsh, many similarities are shared with tropical mangrove swamps, as well as some key differences. Also known as coastal estuaries (wetlands dominated by herbaceous vegetation (Mitsch and Gosselink, 2007)), salt marshes are also often found on large, gentle slopes that allow for significant tidal penetration and do not encounter strong erosive actions such as excessive waves and tidal flow, as well as having moderate salinities and a changing water table due to the tidal flushing and constant fluvial influx of freshwater, similar to mangrove wetlands (Moore, 2001). The first main difference between mangrove swamps is their geographical location, only being found in temperate climates at higher latitudes and characterized by exposure to winter temperatures as a function of stress (Moore, 2001; Mitsch and Gosselink, 2007).

Salt marsh sediment can also vary due to geographical location, but is usually composed of similar fine sands and clays due to coastal proximity, similar to mangrove swamps (Odum, 1988). However, salt marshes often experience much lower productivity rates than mangrove swamps due to smaller amounts of biomass and less biodiversity (Moore, 2001), although in the grand scheme of ecosystems, still exhibit remarkable amounts of productivity despite the stress of constant water flushing (Mitsch and Gosselink, 2007; Moseman-Valtierra,). They are often characterized by much different flora than mangrove swamps, being grasses and vascular plants as opposed to trees. Like mangroves, the grasses found in salt marshes are also facultative halophytes evolved to tolerate saline conditions (Odum, 1988). These smaller plants can vary depending on the region of the salt marsh, however, with the stress of salinity selecting for plants that are able to grow in salt marshes, it is not often that there are more than ten species of plant growing in a regularly tidal flooded marsh (Odum, 1988). Because of this, most salt marshes around the world are colonized by glassworts (Salicornia spp.) and the cord grasses (Spartina), two genera of plants that occupy the intertidal "low" marsh region (Moore, 2001).

In east and Gulf coast salt marshes in the United States, which are conveniently the marshes that have the most GHG data associated with them, low marsh areas are universally dominated by only one species, being the smooth cord grass (*Spartina alternifora*) (Odum, 1988). The less frequently flooded "high" marsh is also only characterized by three or four species in most cases, despite less selectivity due to salinity (Mitsch and Gosselink, 2007; Odum, 1988). In recent years, the high marsh areas of east and Gulf coast marshes have become dominated by the invasive *Phragmites australis*, or common reed, a tall grass that is also halophytic (Chambers *et al.*, 1999; Nixon and Oviatt, 1973). Given that these types of wetlands are found all over the world in various temperate climates and with differing types of ecology, it is also understandable how they could feasibly play such a large role on GHG storage and flux, especially with the amount of productivity they experience.

Figure 6. Fox Hill Salt Marsh, Jamestown, RI. Large amount of the seagrass *Spartina alterniflora* can be seen, as well as the fine muddy sediment and water line. Source: Melanie Gárate.

As for non-tidal wetlands, the major difference from tidal wetlands is the lack of an oceanic tidal presence. In most cases, this means an absence of saline water, although there are inland saline wetlands that are noted for their low productivity, due to their potential to become overly saline with no water exchange, which also causes an increases probability for pollution and toxic build-up (Moore, 2001). However, non-tidal wetlands usually have comparable macrophytes and plants present as salt marshes, and although they do not have to be halophytic due to the presence of solely freshwater, many similar types of reeds and grasses are found in freshwater marshes (Moore, 2001). Examples of these include *Phragmites australis* (also found in many coastal marshes), cattails (*Typha* spp.), and sawgrass (*Cladium jamaicensis*) (Moore, 2001).

Inland freshwater wetlands also have fairly different sediment than their coastal counterparts, being composed of less sand and silt and more earthy souils (Mitsch and Gosselink, 2007). The upper layer of these types of marshes are usually composed of peat, or plant material in the process of decomposition, with lower layers being made up of further decomposed peat

(Mitsch and Gosselink, 2007). These soils, especially peats, are known for their higher capacity to hold water and as a result, their increased tendency to hold nutrients, meaning that they allow less nutrient availability to the surrounding areas for the biogeochemical cycles essential to GHG flux (Mitsch and Gosselink, 2007). As a result, it is thought that for these types of wetlands that they will contribute an overall less significant amount of GHGs to the environment than coastal marshes, which have much greater nutrient loading, due to the influx of river and tidal waters, both often containing more nutrients than groundwater (Nixon *et al.*, 1986).

The other major difference between tidal and non-tidal wetlands is the supply of water to the wetlands. Non-tidal wetlands can often be found on the edges of lakes or streams, in cooler and more temperate areas subject to a large amount of rainfall, and also in areas with access only to groundwater (Moore, 2001; Kennedy, 1981). This means that as opposed to coastal wetlands, which are often fed by estuaries and rivers that bring water including anything that had been deposited at any point along the river flow, as well as being fed by tidal flooding, inland freshwater wetlands are often supplied with fresh and unpolluted water directly from the ground, save for atmospheric deposition, acidic rain and toxic lakes or streams. Due to this and the aforementioned peat, nutrient excess is much less of a problem in these types of wetlands (Cherry).

Figure 7. An inland meadow, with many of the same grasses and flora as seen in coastal marshes. Source: http://geochange.er.usgs.gov/sw/impacts/hydrology/wetlands/inland.jpg

For the last example of wetlands being utilized in this project, subarctic peatlands, a few similarities are shared with freshwater meadows. The sediment composition for these two types of non-tidal wetlands are similar in terms of the formation of peat, however with not as much precipitation in these biomes, water is accumulated in wetlands more from thawing permafrost when the seasons change, and vegetation is dominated by more mosses and smaller grasses and shrubs (Moore, 2001; Mitsch and Gosselink, 2007). The major difference between the freshwater meadows and subarctic peatlands is the differences in climate, which is the main cause for the different seasons of ecological activity in these wetlands, as harsher and colder winters in the subarctic peatlands prevent a large amount of biogeochemical activity from occurring during frozen winters. However, the most significant climate changes due to global warming are expected to occur in these arctic biomes, with predictions of 5-8°C increases, and it is expected that increased GHG fluxes will occur as a result of warmer winters and more biogeochemical activity (Friborg *et al.*, 1997). In addition to this, precipitation levels are not expected to rise proportionally with the rise in temperature and therefore, an increase in evaporation from the wetlands is anticipated to occur, which will cause even larger amounts of GHG flux from these arctic peatlands, as is explained on page 30 (Roulet *et al.*, 1992).

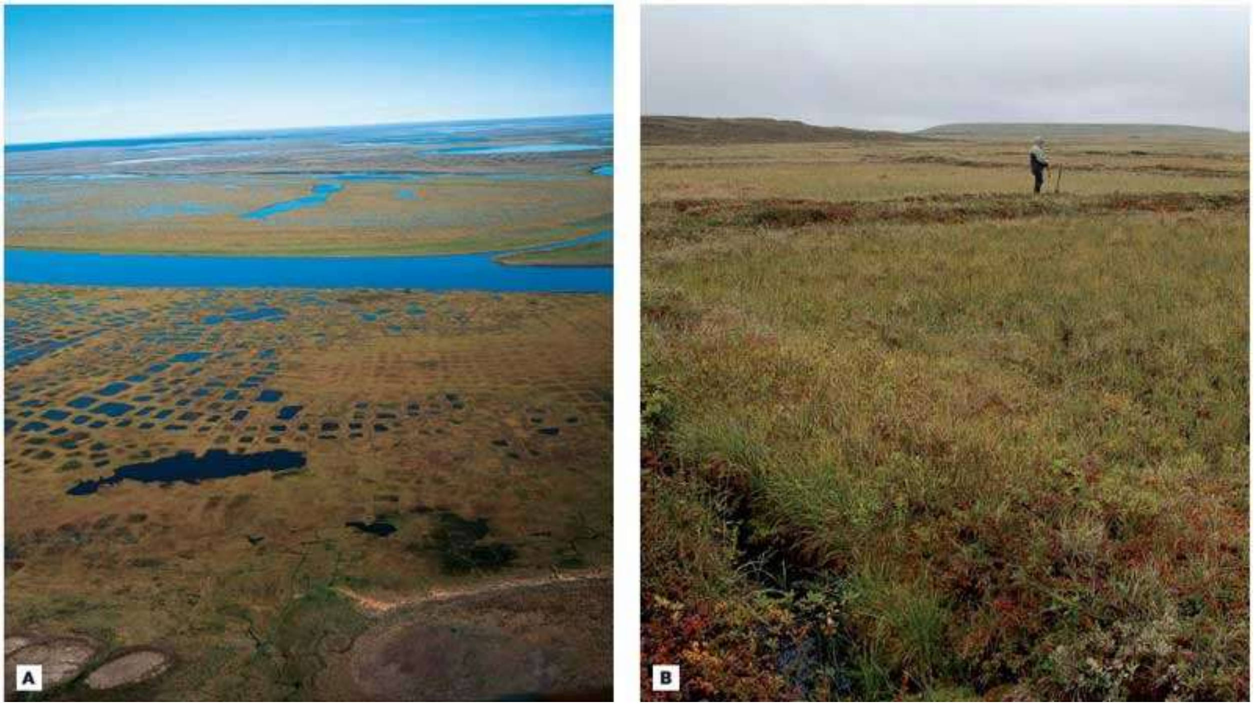

Figure 8. Arctic Peatlands, seen from an airial view (A) and from the ground leve (B), where the grasses and flora characteristic of these wetlands can be seen. Source: http://arcticbiodiversity.is/abt2010/images/stories/report/Ecosystems/14/3.jpg

With the differences and characteristics about each individual type of wetland being mentioned in this project now defined, the ways these features can contribute to the varied amount of GHG flux can be better understood and explained in later sections. GHG flux data from these types of wetlands will be compared and contrasted on pages 26-30, with the eventual goal of displaying which type of wetland can be expected to be the largest contributor to GHGs in the Earth's atmosphere in the coming years, especially as the inevitable future of climate change becomes a reality (discussed in further detail in on pages 31-32). Before this, however, the typical methods of how this GHG data is collected will be reviewed, as they affect the magnitude of fluxes that are reported in various experiments.

ANALYTICAL METHODS USED TO COLLECT GHG DATA

As mentioned in the introduction, my personal experience with assisting members of the Moseman-Valtierra lab at the University of Rhode Island this past summer, as well as scientists in a collaborative project at Waquoit Bay National Estuarine Research Reserve (http://wbnerrwetlandscarbon.net), has given me an introduction to the latest methods currently being utilized to collect GHG data on coastal wetlands and estuaries, using a laser-based GHG analysis known as cavity ring-down spectroscopy (CRDS). Although I am most familiar with the GHG data collection methods that I worked with firsthand this past summer, there are many other researchers that are still using less advanced methods in order to gather GHG data, and these include using GHG analysis known as flame ionization detector gas spectrometry (FID-GS). All of these methods however used sealed flux chambers over areas of the wetland in order to only collect the gases being emitted from the wetland, with no exchange from the atmosphere around the chamber. Having researched the literature about GHG data collection from various other research groups and publications, it seems that these new methods I was using this past summer will move towards being the most efficient methods of GHG data capture on wetlands, given their ability for real-time data feedback. However, given the recent development of these methods, most of the data used in this meta-analysis was collected using the traditional methods of wetland GHG data capture.

This original method of doing wetland GHG is known as the static (closed) chamber method, and the most common example of this usually involves measuring GHG emissions that accumulate in a sealed plot of the wetlands by capturing them in an airtight vial while in the field and inserting the sample (commonly referred to as a "headspace" sample) into an FID-GS later while in a laboratory and analyzing the content of GHGs in the sample (Moseman-Valtierra *et al.*, 2011; Chen *et al.*, 2010b; Freeman *et al.*, 1992; Moore and Knowles, 1989; Crill *et al.*, 1988; Allen *et al.*, 2007). Samples must be captured at certain time points throughout a sampling period in order to track changes in GHG flux.

A similar method to the static chamber method is the soil mesocosm method, which is not done in the field, but rather done by taking soil samples from a specific wetland, homogenizing the soil, and simulating tidal or other wetness conditions while storing the soil in a sealed chamber (mesocosm). GHG samples are collected in a vial similarly to the static chamber method, then analyzed using an FID-GS (Chauhan *et al.*, 2008; Chen *et al.*, 2011). Another method I used this summer that can be used with a FID-GS is done by extracting GHGs from samples of the water column, done by equilibrating water samples with helium (Zhang *et al.*, 2010; Crill *et al.*, 1988; Jahangir *et al.*, 2012). Adding helium to an airtight vial full of the water and shaking it causes all gases contained in the water column to come out with the helium, and by then expelling the water from the vial, the gases left behind can then be injected into an FID-GS and analyzed for GHG content. Mathematical formulas using the water temperature, salinity, GHG amounts and other factors are then used to convert the values of GHGs in water to comparable atmospheric GHG values (Zhang *et al.*, 2010; Kristensen *et al.*, 2008). In these methods that involve water column gas equilibration, it is incredibly important that no exposure to air outside of the mesocosm occurs in order to ensure that it is only the gases contained in the water column that are being accounted for during FID-GS analysis (Zhang *et al.*, 2010; Jahangir *et al.*, 2012). These methods utilizing FID-GS are fairly different in the GHG data collection process, however, the important aspect to keep in mind for the meta-analysis is that all of these methods eventually determine concentrations of GHG, which is going to be comparable no matter what the method. Although there are bound to be minor differences between researchers' individual methods, the specific concentration values can still be compared. One major difference between these methods and the new methods using CRDS is going to be the accuracy of real-time flux measurements, as the technology analyzing GHG concentrations in CRDS is much more advanced.

These new methods, based on CRDS and other laser-based analyses, were first used in coastal marshes within the last few years, and with support of a $1.3 million research grant being awarded to Moseman-Valtierra and her collaborators Dr. Kevin Kroeger (USGS) and Dr. Jianwu Tang (WHOI) in order to investigate nutrient addition to coastal wetlands (Emmett-Mattox, 2011). The major manufacturer of devices used in GHG data collection is Picarro™ (Santa Clara, CA), who recently created a field-ready analyzer made specifically for deployment in wetlands and gathering GHG data, a process I am thoroughly familiar with. These machines are real-time gas analyzers that pump air from a sealed area of a wetland and provide readings about levels of CO_2, CH_4 and N_2O every second. These readings are determined by measuring the amount of infrared absorption for each molecule. Sealed areas of wetland are often installed with collars that are left open at all times except for when they are being sampled from, upon which an airtight plastic container with tubing leading to and from the analyzer can be fit to the collar to form an airtight seal and sample the GHG concentrations inside of the container.

This allows for a trace over time to be made of changes in GHG levels, as well as instantaneous data capture when phenomena like nutrient spikes are added, a common variable utilized by the Moseman-Valtierra lab. CRDS has been tested and confirmed to be as accurate as flame ionization detector gas spectrometry (FID-GS) in analyzing greenhouse gases (Crosson, 2008). The specifics in how CRDS operates is that the optical absorbance of the gas sample pumped into the cavity is established by a wavelength monitor that determines the light dissipation rate of two lasers, and therefore the exponential decay of certain isotopes found in the GHGs CO_2, CH_4 and N_2O (Chen *et al.*, 2010a).

Dr. Lisa Marie Windham-Meyers of USGS and Dr. Tang are also working to utilize these real-time analyzers over long-term time scales in developing research. This is being done by employing eddy flux towers that use the same Picarro CRDS analyzer, with the analyzer being deployed continuously on the larger towers as opposed to being attached to different chambers and plots of land throughout the wetlands. These methods provide landscape-scale flux measurements for a wetland, often covering more area than the other CRDS methods explained above, however still smaller scale than other developing technologies for wetland GHG analyses.

Figure 9. Graphic from Piccaro™ illustrating how the analyzer is used in field GHG wetland data collections. Although this graphic is overly simplified, not including the external pump that draws air from the chamber into the machine, the computer monitor that shows the real-time readings, or the housing and/or cart that usually goes along with the machine into the field, it shows how an average sampling area would look, including the airtight sealed flux chamber used to collect GHG fluxes from a specific area of the wetland.
Source: http://www.picarro.com/sites/default/files/g2508.jpg

One final means of GHG data collection is a relatively new technology using satellites to collect remote sensing data of wetland regions. This data will work to combine local small-scale GHG data of wetlands and using modeling and the remote analysis of wetland area and coverage from space, develop a holistic set of data for GHG emission of a large area of wetlands (Varner, 2013; Fujiwara *et al.*,). Although at this point in time the technology has a lot of improvements and time that need to be put into it, the methods for these technologies have a great potential to improve, recognized by a $1.6 million dollar grant by NSF given to a group of researchers headed by Ruth Varner at the University of New Hampshire to develop remote sensing technologies to model methane emissions from arctic wetlands, with the hope to expand it to analyze wetlands of all climates (Varner, 2013; Aguirre, 2013).

Using these various methods of GHG flux analysis, there are multiple research groups around the world that have collected significant data about the various wetland types that were described above. By conducting a large-scale search and collection of the different data found in relevant publications, the next goal of this project is to determine which type of wetland featured here is liable to cause the most issues in terms of its ability to store GHGs and how different factors of climate change and global warming can affect the amounts of GHGs being released from these wetlands.

METHODS AND RESULTS OF DATA COMPARISON REVIEW

As mentioned in the introduction, the established method of data comparison that is used in this project is known as a meta-analysis, or a compilation of many sources of primary data. Although there are varying types and methods for doing a meta-analysis, for this GHG data, which is often presented in ranges due to the extreme variability that can occur with something as fickle as gas flux, most of the statistical tests done in other versions of meta-analyses will be foregone, especially given the large standard deviations that are often seen with GHG flux data. The general methods of this meta-analysis are based off of those outlined in (Koricheva *et al.*, 2013), and as a previously existing meta-analysis of GHG flux data such as this could not be found, adjustments were made as necessary in order to suit the needs of this project.

These adjustments include choosing averages that represented the largest amount of area used in each study, as well as being consistent in choosing the largest average values in order to represent worst-case scenarios for each type of wetland, considering the final goal of this project to analyze wetland GHG contributions in terms of global warming and climate change. I also converted each of the values to a universal unit of grams per square meter per year to illustrate the amount of GHG flux over a year, again in order to analyze significance to global warming, as other data from other aspects of GHG production, such as urban and anthropogenic contributions, are often published in units per year. Unfortunately, an issue with doing these calculations is that GHG fluxes are rarely constant over the course of a year, due to seasonal change and water table levels, among other varying factors. However, almost all values were originally published in units per hour or day, and these values can also be observed in order to gain a more accurate perspective of these wetland GHG contributions in comparing them to each other. Using this compilation of the data, the differences between wetland types in GHG flux and contribution to global emissions of GHGs, as well as climate change and global warming, can begin to be analyzed in order to attempt to form a cohesive opinion about the issue.

Table 1. Results of wetland GHG meta-analysis

Publication	**Wetland Type**	**Area of World**	**Method Used**	**Range Average CO$_2$ Flux**	**Calculated (g m^{-2} yr^{-1})**	**Range Average CH$_4$ Flux**	**Calculated (g m^{-2} yr^{-1})**	**Range Average N$_2$O Flux**	**Calculated (g m^{-2} yr^{-1})**
Allen et al. (2006)	Tropical Mangrove	Australia	Static	----	----	.072-417.8 mg m^{-2} h^{-1}	0.63-3560	-4 to 65 µg m^{-2} h^{-1}	-0.035-0.57
Allen et al. (2010)	Tropical Mangrove	Australia	Soil Mesocosm	----	----	5.93-37.67 mg m^{-2} h^{-1}	52-330	28.3-201.9 µg m^{-2} h^{-1}	.25-1.77
Chauhan et al. (2008)	Tropical Mangrove	East India	Soil Mesocosm	----	----	2.2-77.52 mg m^{-2} d^{-1}	.8-28.3	9-208 µg m^{-2} h^{-1}	.08-1.8
Chen et al. (2010)	Tropical Mangrove	South China	Static	5.23-10.78 mmol m^{-2} h^{-1}	2015-4155	11.91-274.43 µmol m^{-2} h^{-1}	1.67-38.5	0.14-23.83 µmol m^{-2} h^{-1}	.05-9.2
Kreuzwieser et al. (2003)	Tropical Mangrove	Australia	Static/Soil Mesocosm	-----	-----	350 ug m^{-2} h^{-1}	3.1	6-13 ug m^{-2} h^{-1}	.05-.11
Krithika et al. (2008)	Tropical Mangrove	South India	Static	-----	-----	18.69-37.47 mg m^{-2} d^{-1}	6.8-13.7	0.43-0.81 mg m^{-2} d^{-1}	.16-.30
Livesley and Andrusiak (2012)	Tropical Mangrove	Australia	Static	-----	-----	25-375 ug m^{-2} h^{-1}	.219-3.29	3 ug m^{-2} h^{-1}	0.03
Bubier et al. (1995)	Arctic peatland	North Canada	Static	-----	-----	92.3-380 mg m^{-2} d^{-1}	34-139	-----	-----
Crill et al. (1988)	Subarctic peatland	Minnesota, USA	Static	-----	-----	11-866 mg m^{-2} d^{-1}	4-316	-----	-----
Freeman et al. (1993)	Subarctic peatland	Wales, UK	Soil Mesocosm	646 mg m^{-2} h^{-1}	236	230 mg m^{-2} d^{-1}	84	0.11 -1.14 mg m^{-2} d^{-1}	.04-.42
Friborg et al. (1997)	Subarctic peatland	NW Sweden	Static	1.8 g m^{-2} h^{-1}	657	2.6-22.5 mg m^{-2} d^{-1}	.95-8.2	-----	-----
Moore and Knowles (1989)	Subarctic peatland	Canada	Soil Mesocosm	6.6-9.4 g m^{-2} d^{-1}	2409-3431	12-28 mg m^{-2} d^{-1}	4.4-10.2	-----	-----

Moseman-Valtierra et al. (2011)	Coastal salt marsh	Massachusetts, USA	Static	-----	-----	139-836 umol m^{-2} d^{-1}	.8-4.9	0-42 umol m^{-2} d^{-1}	0-0.68
Davis et al. (2004)	Coastal salt marsh	Rhode Island, USA	Dissolved Gas Mesocosm	-----	-----	-----	-----	negligible	-----
Bartlett et al. (1987)	Coastal salt marsh	Virginia, USA	Static	-----	-----	5.6-22 g m^{-2} yr^{-1}	5.6-22	-----	-----
Ford et al. (2012)	Coastal salt marsh	Wales, UK	Static	333-420 mg m^{-2} h^{-1}	2917-3679	9 mg m^{-2} h^{-1}	78.8	-----	-----
Chmura et al. (2011)	Coastal salt marsh	Canada	Static	398-485 mg m^{-2} h^{-1}	3486-4249	19-35 ug m^{-2} h^{-1}	.17-.31	2-13 ug m^{-2} h^{-1}	.01-.11
Yang et al. (2013)	Freshwater inland marsh	NE China	Static	360.2 mg m^{-2} h^{-1}	3155	7.43 mg m^{-2} h^{-1}	65	12.84 ug m^{-2} h^{-1}	0.11
Sha et al. (2011)	Freshwater inland marsh	Ohio, USA	Static	-----	-----	0.33 to 85.7 mg m^{-2} h^{-1}	2.9-751	-----	-----
Song et al. (2006)	Freshwater inland marsh	NE China	Static	779.33-965.40 mg m^{-2} h^{-1}	6827-8457	19.19-30.52 mg m^{-2} h^{-1}	168-267	0.072-0.15 mg m^{-2} h^{-1}	.63-1.3
Pennock et al. (2010)	Freshwater inland marsh	Canada	Static	-----	-----	3.33-110 g m^{-2} h^{-1}	3.33-110	0-0.4 kg ha^{-1} yr^{-1}	0-0.4
Zhang et al. (2007)	Freshwater inland marsh	NE China	Static	898.59-2,154.17 mg m^{-2} h^{-1}	7872-18869	6.05-13.56 mg m^{-2} h^{-1}	53-119	-----	-----

Tropical Mangroves
- CO_2 – 2015-4155* g m^{-2} yr^{-1}
- N_2O – 0.03-9.2 g m^{-2} yr^{-1}
- CH_4 – 2-330 g m^{-2} yr^{-1}

Coastal Salt Marshes
- CO_2 – 3000-4300 g m^{-2} yr^{-1}
- N_2O – 0-0.68 g m^{-2} yr^{-1}
- CH_4 – 0.17-80 g m^{-2} yr^{-1}

Inland Meadows
- CO_2 – 6800-19000 g m^{-2} yr^{-1}
- N_2O – 0.11-1.3 g m^{-2} yr^{-1}
- CH_4 – 3-270 g m^{-2} yr^{-1}

Artic peatlands
- CO_2 – 240-3400 g m^{-2} yr^{-1}
- N_2O – 0.04-0.42 g m^{-2} yr^{-1}
- CH_4 – 10-316 g m^{-2} yr^{-1}

Figure 10. A condensed graph illustrating the most significant results from Table 1

Given the data compiled in Table 1, it appeared that for the most part, salt marshes had a negligent amount of N_2O flux, although this is affected by the fact that very few papers have examined N_2O in wetlands relative to studies done looking at of CO_2 and CH_4. In addition, the Moseman-Valtierra lab is currently one of the few sources of investigation for how potential nitrogen loading can increase N_2O flux in these coastal salt marshes, something that is not considered in many of the papers where negligible N_2O fluxes are reported (Moseman-Valtierra *et al.*, 2011; Addy *et al.*, 2013; Ford *et al.*, 2012). N_2O fluxes in mangroves seems to be more frequently studied (Table 1), and N_2O fluxes seem to be more significant in mangroves than the other wetlands, although since N_2O is not usually the focus of study in the other types of wetlands, it is tough to determine. The study of N_2O in these coastal wetlands is potentially due to anthropogenic contributions of nitrogen, something that would not be seen in the inland wetlands, as nitrogen loading often occurs only in estuaries due to riverine inputs (Calabretta and Oviatt, 2008). Based on the fact that most researchers studying mangroves look at N_2O, and most researchers that study subarctic peatlands concentrate on CH_4, it most likely that these are the most important factors of each type of wetland as determined to be by the scientific community, unless researchers that have been doing work with N_2O fluxes have not been getting significant results and are facing problems with their research being published.

During review of these publications, many mentions of seasonal fluxes of GHGs were seen, as well as comments on the variation of the water table and nitrogen input. For example, the greatest amounts of CH_4 fluxes were seen in the autumn in mangroves, with an intermediate amount in the summer and the least in the winter, while an opposite trend was seen in subarctic peatlands, as the thawing of the lands in the spring and summer correlated with greater CH_4 fluxes, while freezing in the autumn and winter correlated with smaller CH_4 fluxes (Song *et al.*, 2006; Barnes and Upstill-Goddard, 2011; Allen *et al.*, 2011; Friborg *et al.*, 1997; Chen *et al.*, 2010b). N_2O tended to be much more variable in terms of seasonal variation in mangroves (Allen *et al.*, 2011), but seasonal variations have been seen in coastal salt marshes, as well as noted in the literature by those comparing mangrove seasonal N_2O variations to those of other wetlands (Liikanen *et al.*, 2006).

As for water table effects on GHG emissions, those have been seen and studied most often in the inland nontidal marshes, but most significantly in subarctic peatlands (Freeman *et al.*, 1992; Moore and Knowles, 1989; Yang *et al.*, 2013). In both temperate freshwater marshes and subarctic peatlands, the higher the water table was observed to be, the lower the GHG emissions tended to be (Freeman *et al.*, 1992; Moore and Knowles, 1989; Yang *et al.*, 2013). This is a very important feature to keep an eye on, as climate alterations in the near future could cause droughts and other abnormal water table variations that could function as a positive feedback towards global warming, as increased GHG emissions lead to increased climate change and so on.

Nitrogen addition is something that is mostly studied in the tidal estuarine wetlands such as mangroves as salt marshes, due to the previously mentioned riverine fluxes that often bring nitrogen from fertilizers from inland farms and flush out in estuaries, also known as non-point source pollution (Moseman-Valtierra *et al.*, 2011; Song *et al.*, 2006; Zhang *et al.*, 2010; Chen *et al.*, 2011; Addy *et al.*, 2013). This anthropogenic effect is most likely the reason that N_2O fluxes tend to be most significant in these types of wetlands, as with nitrogen being the limiting factor in the nitrogen cycle and denitrification, responsible for these wetlands' productions of N_2O, additional nitrogen increases the amount of N_2O that can be produced (Moseman-Valtierra *et al.*, 2011).

COMPARISON OF WETLAND GHG FLUXES TO GLOBAL FLUXES

The best estimates for wetland cover unfortunately are not broken into the four types of wetland biomes concentrated on in this project, however estimates that do exist show that wetlands cover a fair amount of Earth's land. The area of wetlands in subarctic areas have been made at 2.7 million km^2, with temperate wetlands thought to cover an estimated 0.7 million km^2 and subtropical and tropical wetlands covering 1.9 million km^2 (Matthews and Fung, 1987). Considering the land area of the earth is about 145 million km^2, these amounts of wetlands are clearly significant to the global contribution of GHG fluxes.

Although this GHG data does not make it a simple task to compare the estimated ranges of GHG flux data to the data provided by the EPA about global GHG contributions, which are still estimated but not in ranges, calculations can be done to observe how much some of the wetlands studied in this study contribute to the projected yearly GHG production. As an example of how just one of the wetland types can contribute, and since subarctic wetland area is the only value given that is for a specific biome covered in this project, an estimate for the amount of GHG contributed by subarctic peatlands can be determined.

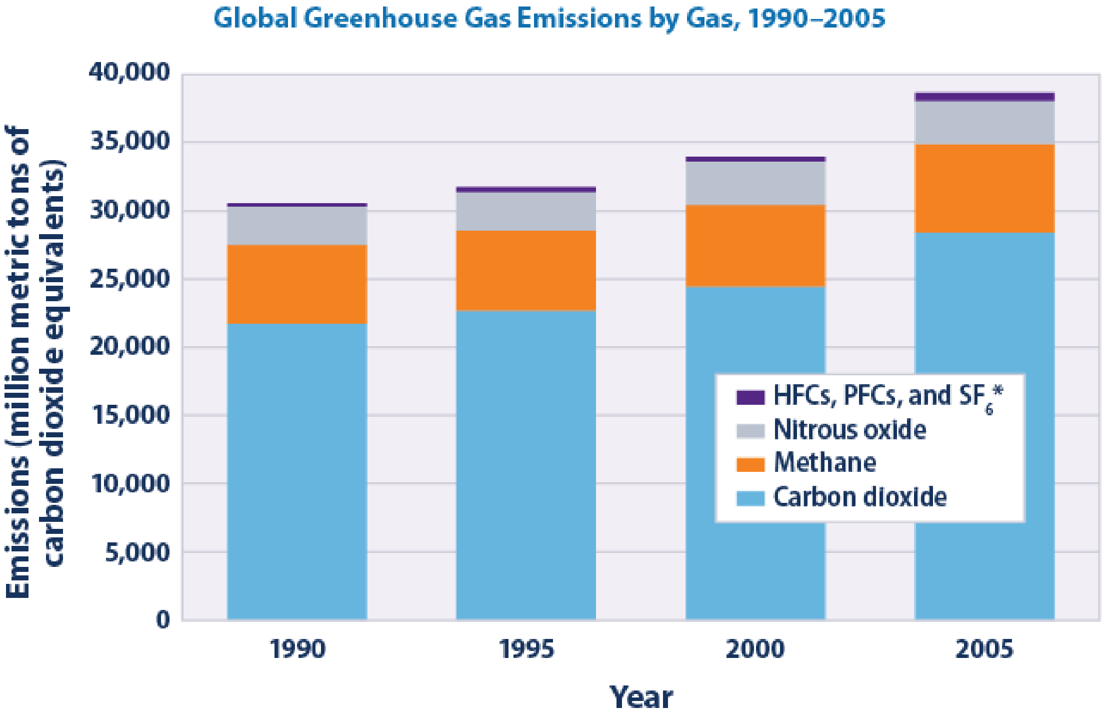

Figure 11. Graph illustrating the increase of global carbon dioxide equivalents emissions. Source: IPCC, Pachauri R, Reisinger A (2007)

It is stated in the IPCC 2007 report that in 2005, 39 billion metric tons of CO_2 equivalent were emitted (Figure 11). Looking at the values in Figure 10 for yearly GHG fluxes from arctic wetlands, and multiplying these values by 2,700 since there were multiplied by 2.7 trillion (for the 2.7 trillion m^2 in 2.7 million km^2), and divided by 1 million (since there are 1 million grams in a metric ton), and also multiplying these values by their CO_2 equivalents for CH_4 and N_2O, which are 25 and 300 as stated on page 4, estimated calculations for yearly contributions by arctic wetlands could be made. Ranges for these values were 648,000 - 9.18 million metric tons of CO_2, 32,400 - 340,200 metric tons of CO_2 equivalent from N_2O, and 675,000 - 21.3 million metric tons of CO_2 equivalent from CH_4, coming to a total range of 1.34 - 30.8 million metric tons estimated to be contributed by arctic wetlands alone.

An estimated 52 to 58% of global CH_4 emissions per year are a result of arctic and subarctic wetlands, which does not include the 6 to 20% of CH_4 emissions that comes from rice paddies (Bloom *et al.*, 2010). As temperature changes in these regions begin to take effect as climate change becomes a greater reality, this potent GHG will continue to exhibit a positive feedback effect (Moore, 2001; Wright and Reddy, 2009). In order for these wetlands to be considered GHG sinks, they would have to sequester 25 times as much CO_2 as CH_4, due to the difference in radiative forcing, but research has suggested that this is not the case with these increased CH_4 emissions (Wright and Reddy, 2009; Mitsch *et al.*, 2012).

Although it seems that in direct values of CO_2, wetlands do not have that large of an effect on the global flux, this is mostly because other anthropogenic activities tend to have such an extreme effect on actual emissions. However, it is simply the fact that wetlands are producing GHGs which has a significant impact on their contribution to global warming, since these increased GHG fluxes function to detract from the role of wetlands as universal stores of carbon (Mitsch *et al.*, 2012).

CLIMATE CHANGE AND PROJECTED CHANGES TO WETLANDS

Although some have already been mentioned in the previous section, there are multiple changes predicted to occur with the onset of climate change and global warming that will contribute to the importance of wetlands in the grand scheme of GHG flux. One major problem that has to be accounted for is the significant temperature increase expected in arctic regions, as GHG production from subarctic wetlands is bound to increase given the likely lowering of the water table (discussed on page 30) and the health of methanogenic bacteria, which tend to produce more CH_4 under abnormally warmer climates (Moore, 2001; Wright and Reddy, 2009).

The other major problem that is likely to affect all coastal wetlands is the problem of sea level rise, with the global average sea level rise of 1-2 millimeters per year estimated to increase by two to five fold during the next 100 years (Reddy and DeLaune, 2004). The issue with sea level rise in terms of wetlands is the submerging of coastal wetlands, as if the accretion of sediment in wetlands cannot keep up with the rise of the sea level, and wetlands cannot migrate further inland, they will no longer exist and be able to serve their carbon sequestration purposes (Reddy and DeLaune, 2004).

FUTURE PREVENTATIVE ACTIONS

The issue of global warming is not one that can be easily handled, but there are a few methods that have been researched as to how GHG contributions from wetlands can be alleviated. One of these methods that have received a lot of attention in recent years is the blue carbon movement. Since it is understood that healthy coastal habitats are able to absorb and sequester significant amounts of carbon and help to alleviate the global increase in CO_2 levels, a lot of research and awareness has been funded about blue carbon and how wetlands are able to sequester carbon at much higher rates than trees and oceans. Based on this knowledge, a lot of effort is being put into the use of constructed wetlands, or artificially made wetlands that function in the same ways that natural wetlands function in terms of biogeochemistry and carbon sequestering ability. Another method of artificially adjustment of wetlands that can help with GHG fluxes is the use of vascular plants in subarctic peatlands to decrease CH_4 emissions (Joabsson *et al.*, 1999). Since peatlands are usually composed of mostly peat and very few vascular plants, this research has seen that if these peatlands are instead filled with vascular plants, processes that are associated with transport, production and consumption of methane are often improved (Joabsson *et al.*, 1999).

In terms of legislative actions and acts of human awareness that can be improved in order to assist with wetland GHG flux issues, one problems that is contributing a significant amount of GHGs to the atmosphere while also releasing and negating stores of sequestered carbon is peat burning and mining that occurs in Southeast Asia (Page *et al.*, 2009). It is estimated that during fires in 1997, between 190 and 230 million metric tons of CO_2 were released to the atmosphere, more than ten times the amount of carbon estimated to be released by arctic wetlands on page 32 (Page *et al.*, 2009). In addition, improvements on wastewater treatment and control of fertilizer release to rivers are something that can help improve nitrogen loading to coastal wetlands. As mentioned on page 5, increased nitrogen often contributes greatly to increased N_2O fluxes, and with less nitrogen being released to coastal wetlands, it is expected that less N_2O will be produced.

CONCLUSION

Even though climate change and global warming is a multi-faceted and complex issue, with many contributions that can be tough to understand, the GHG fluxes of wetlands is one topic that has the potential to be well understood. As techniques to measure GHG fluxes in wetlands continue to improve and our knowledge of how their carbon sequestration abilities are affected by some of our actions accumulates, hopefully our understanding of their role in the grand scheme of global warming becomes significant enough to use them to our advantage. This is something that will become increasingly important as the crisis of climate change becomes worse in coming years, and it is expected that the idea of wetlands as a major component of climate change research will become more widely known and accepted with the help of research such as the studies featured in this project.

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