Global Climate Change and Greenhouse Gases Emissions in Terrestrial Ecosystems

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Contents

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Abstract

Global climate change has significantly influenced soil greenhouse gases (GHG, i.e., carbon dioxide – CO_2, methane – CH_4, and nitrous oxide – N_2O) emissions that feedback to climate change. Terrestrial ecosystems are important sources and sinks of these GHG that are produced and consumed through biological processes including decomposition, methane oxidation, photosynthesis, methanogenesis, nitrification, and denitrification. In this chapter, we synthesize publications related to global climate change and soil GHG emissions and provide case studies of the impacts of global climate change on soil GHG emissions. The chapter starts with a brief introduction, followed by a description of GHG and soil emission processes. The common methods of GHG emission measurements and research approaches of the global change study are described. Case studies using laboratory incubation, field experiment, meta-analysis, and ecosystem modeling are provided. We focus on the impacts of global warming, precipitation change, atmospheric CO_2 concentration, and nitrogen deposition on soil GHG emissions in terrestrial ecosystems. Some recommendations for future studies are provided.

Keywords

Carbon dioxide · Climate change · Elevated CO_2 · Temperature · Experiment · Global change · Methane · Meta-analysis · Microbial processes · Model · Nitrogen deposition · Precipitation · Respiration · Temperature · Global warming · CO_2 · CH_4 · N_2O · Greenhouse gas emission

Introduction: Global Climate Change and Soil Greenhouse Gases Emissions

Global climate change is one of the great challenges we are facing today. The increasing concentration of greenhouse gases (GHG) in the atmosphere has been verified as the most important cause of global warming (IPCC 2013; Oertel et al. 2016; Wu et al. 2020). Due to fossil fuel burning, land use change such as deforestation, crop production and agricultural practices, emissions of GHG including carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) have increased dramatically. The average global CO_2 concentration has been increased from 280 ppm at preindustrial revolution to higher than 400 ppm presently (Pearson and Palmer 2000; Saban et al. 2019). The annual increases in CO_2, CH_4, and N_2O concentrations are about 0.5%, 0.8%, and 0.3%, respectively (Baggs and Blum

2004). As a result, mean global surface temperature is increasing at an unprecedented rate and will increase by 1.4–5.8 °C over 100 years (Luo et al. 2001; Beier et al. 2004). A more vigorous hydrological cycle is also expected with more severe droughts and floods. These changes are likely to have significant effects on soil GHG emissions, and further feedback to global climate change. Understanding the impacts of the climate change (e.g., global warming, elevated CO_2, precipitation change, and nitrogen (N) deposition) on soil GHG emissions is crucial for predicting future global climate change.

To understand the effects of global climate change on soil GHG emissions, great efforts have been made to quantify soil GHG emissions in grasslands, forests, wetlands, and agricultural systems using laboratory incubations, field measurements, meta-analyses, and ecosystem modeling (Cao and Woodward 1998; Lei et al. 2007; Deng et al. 2016; Tian et al. 2019; Luo and Schuur 2020). Measurements of CO_2 concentration started in the nineteenth century (Russell and Appleyard 1915; Oertel et al. 2016), and measurements of N_2O and CH_4 concentrations started in the 1950s and 1980s, respectively (Arnold 1954; Holzapfel-Pschorn et al. 1985). Different approaches and instruments have been developed and implemented for soil GHG concentration measurements. With these measurements, many laboratory and field experiments have been conducted to better understand the potential effects of global warming, elevated atmospheric CO_2, N deposition (the input of ammonia, nitrate and other forms of N from the atmosphere to the biosphere both as gases, dry deposition and in precipitation as wet deposition), and changing precipitation regimes on soil GHG emissions (Fig. 1). Temperature has been found to be a key factor that regulates terrestrial biogeochemical processes, such as soil carbon (C) decomposition, litter decomposition, N mineralization, nitrification, denitrification, and CH_4 emission and uptake (Rustad et al. 2001; Deng et al. 2016; Tian et al. 2019). Precipitation changes such as drought and flood also have significant impacts of soil CO_2 emission, soil N_2O emission, and soil CH_4 update and release (Li et al. 1996; Deng et al. 2016). As a limiting factor for plant growth and substrate for soil N_2O emission, N deposition or fertilization strongly enhances soil N_2O emission, and has different impacts on soil CO_2 and CH_4 emissions. Climate impacts on soil CH_4 emission also vary with different ecosystems, particularly between dryland and wetland ecosystems (Liu and Greaver 2009; Yan et al. 2018). The interactive effects of global change on soil GHG emissions are more complex and remain uncertain. Future research and tool development are needed to further understand the interactions between climate change and GHG emissions.

In this chapter, we review the existing knowledge of the impacts of global climate change on soil GHG emissions in terrestrial ecosystems. This is an updated version of our previous chapter Impacts of Climatic Changes on Biogeochemical Cycling in Terrestrial Ecosystems (Hui et al. 2012), but with a focus on three GHG instead of C cycling. We first briefly describe the impacts of GHG on climate change and biological processes related to soil GHG emissions, then summarize the commonly used methods to quantify soil GHG emissions. Next, we introduce research approaches applied in global change studies, such as laboratory incubation, field experiment, meta-analysis, and ecosystem modeling. After that, we provide

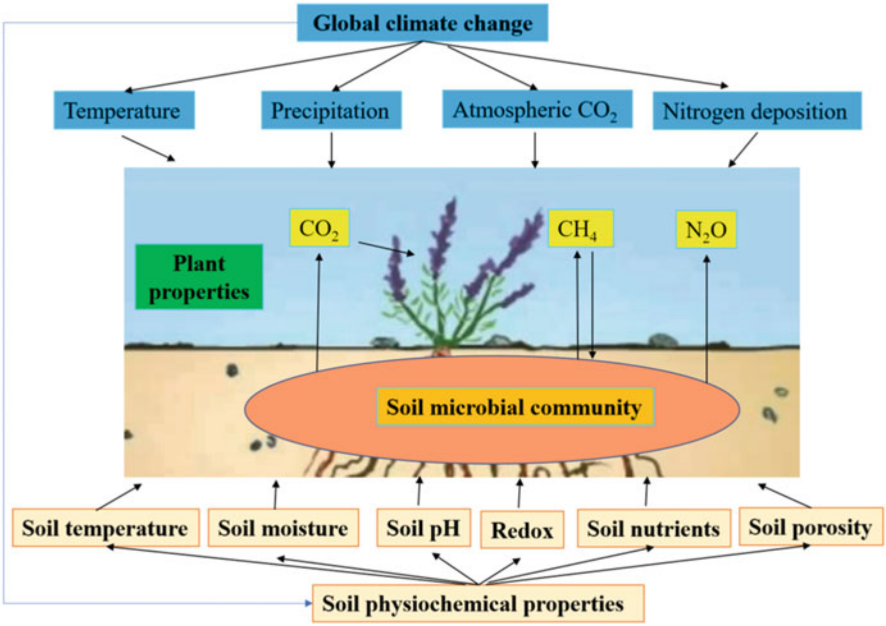

Fig. 1 Conceptual diagram of the impacts of climate change on soil greenhouse gas (CO_2, CH_4, and N_2O) emission through changes in plants, soil properties, and soil microorganisms. Climate changes influence plant physiological processes and soil physical and chemical properties, thus substrates in soil and soil microbial composition and activities, resulting in changes in greenhouse gas emissions

examples of some recent results in global climate change research, particularly using meta-analysis and ecosystem modeling. This chapter is not a comprehensive review of the impacts of global change on soil GHG emissions, rather some case studies in global climate change published from earlier years to most recently. Lastly, we identify some of the remaining questions warranting further research.

Soil Greenhouse Gases Emissions

Atmospheric concentrations of GHG (i.e., CO_2, CH_4, and N_2O) have increased dramatically since the beginning of the industrial revolution largely due to human activities such as fossil fuel combustion and land-use change (IPCC 2013; Rustad et al. 2001; Tian et al. 2016). The absorption and reflection of GHG to infrared radiations generate a warming effect and increase the earth's surface temperature, and further change precipitation intensity and pattern around the world.

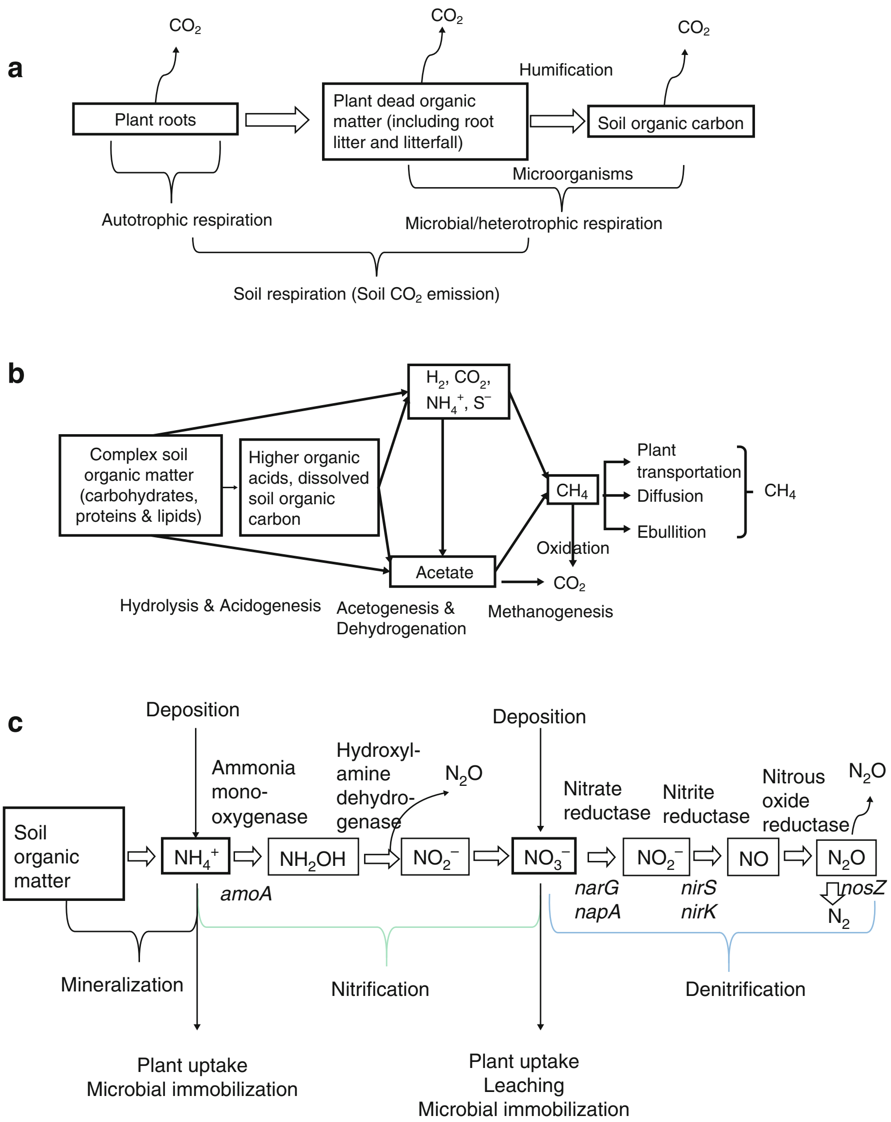

Fig. 2 Biological processes involved in soil greenhouse gas (CO_2, CH_4, and N_2O) emissions. (**a**) Soil CO_2 emission. (**b**) Soil CH_4 emission and uptake. (**c**) Soil N_2O emission

Soil CO_2 Emission

Carbon dioxide is the most important GHG that contributes significantly to global warming and the contemporary climate change (IPCC 2013). Soil CO_2 emission (Rs, also called soil respiration or soil CO_2 efflux) includes heterotrophic respiration of soil microbes (Rh) and autotrophic respiration of plant roots (Ra) (Fig. 2a). Root respiration contributes about 50% of the total Rs, but ranges from 10% to 95% depends on ecosystems and seasons (Hanson et al. 2000). Soil CO_2 emission is an important component of the global C cycle in terrestrial ecosystems and contributes 68–100 Pg C year^{-1} to the atmosphere, roughly nine times of the annual anthropogenic CO_2 emission (Bond-Lamberty and Thomson 2010). More C is expected to be respired from soils as atmospheric CO_2 concentration and global temperature continuously increase.

Soil CO_2 production results from respiration of living roots and microbial decomposition of litter and soil organic matter (Fang and Moncrieff 1999; Hui and Luo 2004). Soil CO_2 transport to the atmosphere is controlled by the rate of CO_2 production in the soil, the CO_2 concentration gradient between the soil and the atmosphere, soil physical properties, and environmental conditions (Fang and Moncrieff 1999). Thus, measured soil CO_2 emission from the soil surface is the result of these complex processes influenced by a number of factors.

Both biotic and abiotic factors can influence soil CO_2 emission. Temperature is considered as the most important factor and Rs rates mostly increase exponentially with temperature. Temperature sensitivity of Rs rates is commonly quantified using the Q_{10} index (the proportional increase in respiration per 10 °C rise, normally is around 2). Besides temperature, soil CO_2 emission is often positively related to root biomass, mycorrhizal associations, and the quality and size of soil C pools (Melillo et al. 2011; Liu et al. 2020).

Soil CH_4 Emission

Methane is the second most important anthropogenic GHG after CO_2 and accounts for at least 20% of the anthropogenic radiative forcing of warming agents since preindustrial era (Tian et al. 2010; Etminan et al. 2016; Ganesan et al. 2019). The greenhouse effect of CH_4 is 28 times that of CO_2 in 100 years (IPCC 2013; Tian et al. 2016). CH_4 is also a chemical precursor to tropospheric ozone (O_3) formation and causes air quality issues for human health (Ganesan et al. 2019). Global mean CH_4 concentration is 1860 ppb in 2018, up from approximately 710 ppb in preindustrial era (Etheridge et al. 1998; Rubino et al. 2019). It has been estimated that 270 Tg CH_4 year^{-1} are emitted from natural sources globally (Zhuang et al. 2007), with annual CH_4 emissions of 550–740 Tg year^{-1}, approximately 50–60% coming from anthropogenic sources (Saunois et al. 2016; Ganesan et al. 2019). More than 40% of the global total methane emission come from wetlands, hydrothermal vents, and oceans (Bridgham et al. 2013; Ma et al. 2017; Zhang et al. 2020).

CH_4 is produced by methanogens through the process of methanogenesis under anaerobic conditions (Liu and Whitman 2008). Methanogens harbor three major pathways of methanogenesis (i.e., hydrogenotrophic, acetoclastic, and methylotrophic pathways) (Bridgham et al. 2013) (Fig. 2b). All the pathways have a common enzyme, methyl-coenzyme M reductase (MCR), for the final step of CH_4 synthesis (Liu and Whitman 2008). The *mcrA* gene is a commonly used gene marker for surveying diversity of methanogens (Yang et al. 2014; Zhang et al. 2020). Depending on its initial complexity, this processing may involve several steps. It starts with degradation of complex polymers by microbial exoenzymes followed by subsequent degradation steps by fermenting bacteria (Bridgham et al. 2013). In freshwater ecosystems, it is generally assumed that the sole fermentation products utilized by methanogens are H_2, which is oxidized to CH_4 using CO_2 as an electron acceptor in the process of hydrogenotrophic methanogenesis, and acetate which is split to form CO_2 and CH_4 in the process of acetoclastic methanogenesis (Bridgham et al. 2013).

Methane can leave a wetland via diffusion, ebullition, and/or plant-mediated transport, and the relative importance of these various routes is an important control on wetland CH_4 emissions (Bridgham et al. 2013; Ma et al. 2017). When CH_4 exits a system through diffusion, chemoautotrophic methanotrophs can oxidize it to CO_2 (Ma et al. 2017). Aerobic methanotrophy can dominate wetland CH_4 cycling, and the global wetland CH_4 oxidation sink has been estimated to be between 40% and 70% of gross CH_4 production (Dorodnikov et al. 2011; Ma et al. 2017). Flux of CH_4 through plant aerenchyma can also be an important component of net CH_4 flux from wetlands, and the contribution of plant-mediated CH_4 flux varies dramatically between ecosystems and ranges from ca. 30% to 100% of total CH_4 flux (Dorodnikov et al. 2011). Ebullition also allows CH_4 leaving a wetland to bypass zones of aerobic oxidation. Historically, ebullition has been thought to be primarily episodic following supersaturation of pore water with CH_4. Goodrich et al. (2015) suggested that ebullition can occur not just as rare releases of accumulated CH_4, but as a regular transport pathway of CH_4 as typical as diffusion and plant transport.

Generally, soils are a trivial atmospheric CH_4 sink, accounting for 4–7% of the global sink (Saunois et al. 2016; Feng et al. 2020). Wetlands and rice fields are the major source of CH_4 emission. When soil conditions change, due to drought, flooding, warming, natural disturbances, and forest management, forest soils may become larger sinks or even turn to sources of CH_4 (O'Connell et al. 2018; Feng et al. 2020).

Soil N_2O Emission

Nitrous oxide is the third important long-lived GHG next to CO_2 and CH_4 and has a direct global warming potential 298 times higher than that of CO_2 on a 100-year span (Tian et al. 2016). In addition to its contribution to global warming, N_2O plays an important role in stratospheric O_3 depletion through O (^{1}D) oxidation (Ravishankara et al. 2009). The atmospheric N_2O concentration has increased to 330 ppb in recent years from 270 ppb in the preindustrial period. It increases with an average of 0.73 ppb year^{-1} in recent decades (Ciais et al. 2014; Tian et al. 2019). A significant amount of N_2O is released into the atmosphere from agricultural soils, due to the large amounts of N fertilizer applied to maintain high crop yield (Smith et al. 2008; Deng et al. 2016). The total release of N_2O from agricultural soils accounts for up to 80% of anthropogenic N_2O emissions (Mosier et al. 1998; Kroeze et al. 1999).

Nitrous oxide emitted from the soil is produced by microbial (i.e., archaea, bacteria, and fungi) transformations of inorganic N, through a series of processes usually involving nitrification (oxidation of NH_4^+ to NO_3^- via NO_2^-) under aerobic conditions and denitrification (reduction of NO_3^- to N_2O and N_2) under anaerobic conditions (Davidson and Swank 1986; Ussiri et al. 2009) (Fig. 2c). Nitrification by ammonia oxidizers is the primary starting process of N_2O production (Inatomi et al. 2019). Different soil microbes, such as ammonia-oxidizing archaea and bacteria, have contributed to nitrification. There are denitrifiers with both of the genes encoding nitrite reductase, *nirK* and *nirS*, and denitrification correlates more strongly with the occurrence of *nirS*- type denitrifiers (Voigt et al. 2020). Denitrification is also the most important N_2O consumption process. The ratio between the genes encoding nitrite reductase and nitrous oxide reductase, which consumes N_2O and is encoded by *nosZ*, has been connected to net N_2O emissions from permafrost-affected soils (Voigt et al. 2020). Thus, higher nitrite reductase to nitrous oxide reductase gene ratios ((*nirK* + *nirS*)/*nosZ*) are typically indicative of increased soil N_2O emissions (Voigt et al. 2020).

These processes are subject to complex regulation involving the interaction among numerous factors, such as the amounts of N fertilizer applied, precipitation or soil moisture content, availability of dissolvable organic C (DOC), and air/soil temperature, pH, and oxygen supply (Li et al. 2000; Dobbie and Smith 2003; Deng et al. 2016; Tian et al. 2019). Field data- and model-based emission estimates show the highest emissions of N_2O in moist tropical areas and lower emissions at high latitudes (Joos et al. 2020). High soil moisture increases N_2O fluxes due to increased denitrification activity in response to reduced oxygen diffusion into the soil (Arias-Navarro et al. 2017; Joos et al. 2020). Warming treatment increases N_2O emissions in boreal peatlands (Dijkstra et al. 2012). Positive or neutral responses in N_2O emissions have been found in field experiments under elevated CO_2 in temperate and boreal forests and grasslands (Dijkstra et al. 2012; Zhong et al. 2018). Nitrogen addition by mineral and organic fertilizer usually enhances soil N_2O emissions (Joos et al. 2020).

Based on the soil GHG emissions, the global warming potential (GWP) is often estimated. GWP is defined as the ratio of the time-integrated radiative forcing from the instantaneous release of 1 kg of a trace gas relative to that of 1 kg of a reference gas. It can be calculated as:

$$\mathrm{GWP} = \mathrm{F}_{\mathrm{CO_2-C}} \times 44/12 + \mathrm{F}_{\mathrm{CH_4-C}} \times 16/12 * \mathrm{RF}_{\mathrm{CH_4}} + \mathrm{F}_{\mathrm{N_2O\_N}} \times 44/28 \times \mathrm{RF}_{\mathrm{N_2O}}$$

where $\mathrm{F}_{\mathrm{CO_2-C}}$, $\mathrm{F}_{\mathrm{CH_4-C}}$, and $\mathrm{F}_{\mathrm{N_2O\_N}}$ are the fluxes of soil CO_2, CH_4, and N_2O based on mass of C and N, respectively. $\mathrm{RF}_{\mathrm{CH_4}}$ and $\mathrm{RF}_{\mathrm{N_2O}}$ are constants indicating radiative forcing of CH_4 and N_2O in terms of a CO_2 equivalent unit, and are 25 and 298, respectively, at 100-year time horizon. Negative GWP indicates GHG uptake from the atmosphere and a potential climate cooling effect while positive GWP indicates GHG release to the atmosphere and a potential climate warming effect (Tian et al. 2016).

Measurement Methods of Soil Greenhouse Gases Emissions

Soil CO_2 emission was first measured in laboratories in the nineteenth century, as soil CO_2 emission or soil respiration is an indicator of soil fertility (Russell and Appleyard 1915; Oertel et al. 2016). The chamber method for soil CO_2 emission was introduced in the fields at the beginning of the twentieth century (Lundegardh 1927). N_2O and CH_4 measurements started in the 1950s and 1980s, respectively, as gas analyzers became available (Arnold 1954; Holzapfel-Pschorn et al. 1985). Now soil CO_2, CH_4 and N_2O emissions have been routinely measured in different terrestrial ecosystems around the world. Soil GHG concentration are being directly measured in both laboratory and field using chamber methods and micrometeorological technique, and simulated using empirical and process-based ecosystem/biogeochemical models (Luo et al. 2001; Huang et al. 2014; Tian et al. 2019; Oertel et al. 2016; Hui et al. 2020). GHG emissions in the atmosphere can also be measured using remote sensing and airplane gas gradient measurements. Here we briefly describe different methods commonly used in the global change research, with a focus on chamber methods.

Soil Chamber Methods

Chamber-based soil GHG emission measurement is widely used in laboratory and field studies of CO_2, CH_4, and N_2O (Kitzler et al. 2006; Zhou et al. 2014a; Oertel et al. 2016; Hui et al. 2020) (Fig. 3). Both closed and open chambers are used for soil GHG emissions measurements. For closed chambers, there are closed static chamber and closed dynamic chamber as described below (Kutzbach et al. 2007; Rochette et al. 1997). To derive the relationships between GHG emissions and environmental factors, sensors for air temperature, soil moisture, air pressure and relative humidity are often monitored with soil GHG emission measurements. Clough et al. (2020) reviewed and synthesized literature on chamber designs and associated factors that affect soil N_2O flux measurement. They discussed the details on the materials, insulation, sealing, venting, depth of placement, and the need to maintain plant growth and activity. Similar considerations can be applied to all GHG emission measurements. Here we briefly describe different chamber methods used for the measurements.

Closed Static Chamber

The closed static chamber is often made of polyvinyl chloride (PVC) material and consisted of two parts: a permanent base (bottom part) and a removable lid with a rubber septum for gas sampling, both made of polycarbonate engineering plastics (Fig. 3a–c; Deng et al. 2015b; Tan et al. 2018). The base is inserted 10 cm into the soil. For manual sampling, the lid can be placed on top during sampling and removed afterwards. A fan is usually installed on the top wall of the cover to create gentle turbulent mixing when the chamber is closed. A typical measurement often lasts for about 30–45 min. Gas samples are generally collected at three time-intervals using syringes (Deng et al. 2015b). All gas samples can be analyzed for CO_2, CH_4 and N_2O concentrations using a gas chromatograph (GC, e.g., Model GC-2014, Shimadzu Scientific Instruments, Columbia, MD). Instantaneous soil GHG emission is calculated based on the rate of change in GHG concentration within the chamber, which is estimated as the slope of linear regression between concentration and time or using an exponential model (Deng et al. 2015b).

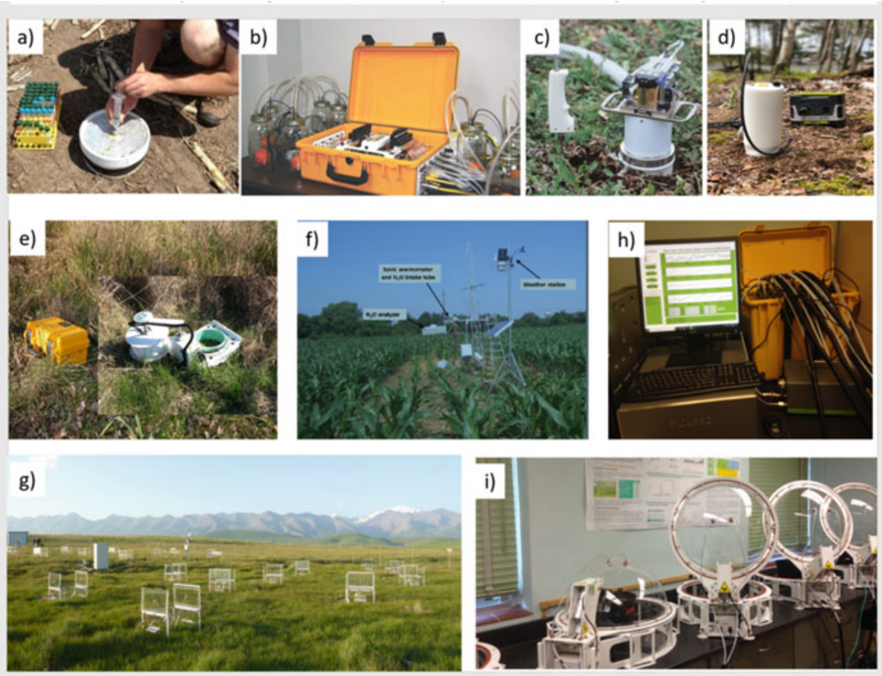

Fig. 3 Examples of measurement methods of greenhouse gas (CO_2, CH_4, and N_2O) concentration. (**a**) Close static chamber method by sampling gas and measuring CO_2, CH_4, and N_2O concentrations using GC. (Photo credit: Eosense). (**b**) Lab incubation. (Photo credit: Eosense). (**c**) Soil respiration measurements using Li-6400 with soil chamber. (Photo credit: Li-Cor, Inc.). (**d**) Continuously measurements of soil respiration using Eosense eosFD soil respiration auto-chamber. (Photo credit: Eosense). (**e**) Continuously measurements of soil respiration using Li-8100 A. (Photo credit: Li-Cor, Inc.). (**f**) Eddy covariance measurements of CO_2 and N_2O exchanges. (Photo credit: Junming Wang). (**g**) Evaluation of feedback of CO_2 and CH_4 fluxes to global change using a 20-channel automated chamber system at a wetland ecosystem on the Tibetan Plateau (leading by J. He of Peking University). (Photo credit: Naishen Liang). (**h**, **i**) Measurements of soil CO_2, CH_4, and N_2O emissions using the Picarro G2508 and Eosense smart chambers. (Photo credit: D. Hui)

For automatic measurements, the chambers usually have a moveable lid which enables gas exchange (Edwards and Riggs 2003; Pape et al. 2009). Chambers are commonly used with a pump, flow controller, and sustained power supply to maintain equilibrium between atmospheric air and soil-source gases (Risk et al. 2011). A vent is usually included to prevent pressure gradients during chamber deployment (Bain et al. 2005). Emitted gases accumulate in its chamber headspace. The change of mixing ratio can be analyzed with various gas sensors, such as gas chromatography (GC) and Cavity-Ring-Down spectrometry (Oertel et al. 2016).

Closed Dynamic Chamber

Closed dynamic chambers are also widely used in field studies. In closed dynamic chamber systems, gases accumulating in the chamber are analyzed either externally and pumped back into the chamber (Rochette et al. 1997; Heinemeyer and McNamara 2011) or are being analyzed inside the chamber with a compact sensor that continuously monitors the atmospheric CO_2 concentrations (Oertel et al. 2016). For example, Li-Cor Li-8100 Automatic Soil CO_2 Efflux System is an example of closed dynamic chamber system. Soil N_2O and CH_4 emissions can be analyzed as well with closed dynamic chambers (Cowan et al. 2015; Deng et al. 2015b), as illustrated in Fig. 3h, i. In this system, the CO_2, CH_4, and N_2O gas concentrations are analyzed with a cavity ring-down spectrometer (Picarro G2508, Picarro, Inc., Santa Clara, CA) with a recirculation pump. Additional components are six Eosense Smart chambers connected through a recirculating multiplexer.

Open Dynamic Chamber

Open dynamic chamber has two openings draw in ambient air and generate a continuous gas flow (Kutzbach et al. 2007; Oertel et al. 2016). Gas concentrations are analyzed at the air inlet and outlet of the chamber. The gas flux is calculated by the difference of the concentrations at both ends. Consequently, there are no accumulation times needed, since the flux is analyzed continuously. One example of automated chambers is forced diffusion (FD) chamber such as the eosFD soil CO_2 emission auto-chamber (Eosense, Dartmouth, Nova Scotia, Canada; Fig. 3e, g, i) (Kim et al. 2016; Oikawa et al. 2017). The FD technique is functionally similar to dynamic steady-state chamber systems but uses a diffusive membrane to regulate the flow of gases rather than a pump (Risk et al. 2011; Kim et al. 2016). While accuracy and precision of eosFD chambers are comparable to other techniques, measurements of Rs with eosFD offer several benefits created by the lack of external moving parts, including reduced power consumption, simplified calibration, great flexibility, and the ability to function in harsh environments (Risk et al. 2011; Kim et al. 2016).

Closed chamber systems require longer accumulations times under dry and hot conditions, leading to temperature and pressure gradients (Balogh et al. 2007; Oertel et al. 2016). Open chambers are suitable such conditions in summer with low gas exchange rates. But open dynamic chambers are more sophisticated and more expensive compared to closed systems (Oertel et al. 2016).

Micrometeorological Methods – Eddy Covariance Technique

The eddy covariance (EC) method is a direct micrometeorological approach (Goulden et al. 1996; Baldocchi 2003; Huang et al. 2014; Oertel et al. 2016). It uses vertical turbulences to analyze the turbulent heat and gas exchange between soil surface and atmosphere (Launiainen et al. 2005). Fluxes of gases are calculated based on the covariance between fluctuating component of the vertical wind and the fluctuating component of gas concentrations. The EC method is applied initially to quantify ecosystem CO_2 exchange between terrestrial ecosystems and the atmosphere (Baldocchi

2003). Later, CH_4 measurements are integrated with CO_2 measurement. EC method requires rapid, simultaneous measurements of gas concentration and wind velocity at the same point in space. With the developments of fast-response (i.e., 10 Hz or higher) N_2O analyzers in recent years, the EC method has been used to monitor ecosystem N_2O emissions (Mammarella et al. 2010; Jones et al. 2011; Huang et al. 2014).

Typical eddy covariance system includes a 3D sonic anemometer (CSAT3-A, Campbell Sci, Logan, UT) measures three-dimensional wind velocities and virtual air temperatures at a sampling rate of 10 Hz, and gas analyzers. CO_2 and CH_4 concentrations can be measured using Li-7500 Gas Analyzer. N_2O concentrations are measured using a QCL spectrometer. The N_2O analyzer needs to be housed in a place with a stable working temperature (20–30 °C). The analyzer provides 10 Hz measurements of N_2O and water vapor (H_2O) concentrations. A Campbell Scientific CR3000 data logger can be used to record all the data collected at 10 Hz. The EddyPro can be used to process and correct the N_2O flux (Huang et al. 2014).

Advantages and Disadvantages of the Measurement Methods

Significant efforts have been invested in developing reliable tools for measuring CO_2, CH_4, and N_2O emissions from the soil into the atmosphere. The two major measurement methods widely used are the chamber method and the eddy covariance method (Molodovskaya et al. 2011; Huang et al. 2014). The static chambers are the traditional tools that are widely used in different ecosystems (Arnold et al. 2005; Klemedtsson et al. 1996). It is simple, costs less, and can be applied in different ecosystems and environments. But it can only cover a small area, may disturb soil environment, and has a low sampling frequency (Molodovskaya et al. 2011; Denmead 2008). The automatic sampling chambers can monitor soil GHG emissions continuously, but are usually expensive. The EC method calculates the mean fluxes of an ecosystem within its footprint (Denmead 2008), does not disturb the soil and crop ecosystem, and provides a continuous and real-time flux measurements (Huang et al. 2014), but it is difficult to apply to small treatment plots with climate change factors due to footprint requirement. The EC system is also more expensive, particularly when N_2O emission is considered. The chamber methods will continue to be the major one for soil GHG emission measurements (Oertel et al. 2016).

Research Approaches of Global Climate Change and Soil Greenhouse Gases Emissions: Laboratory Incubation, Field Experiment, Meta-Analysis, and Ecosystem Modelling

Laboratory Incubation

Laboratory incubations are very useful when the impact of a single climatic factor such as soil temperature or nutrient availability on soil GHG emissions is evaluated (Zhou et al. 2014b; Oertel et al. 2016; Hui et al. 2020) (Fig. 4a, b). It is relatively easy to control one factor and kept others constant in a laboratory setting. Climate

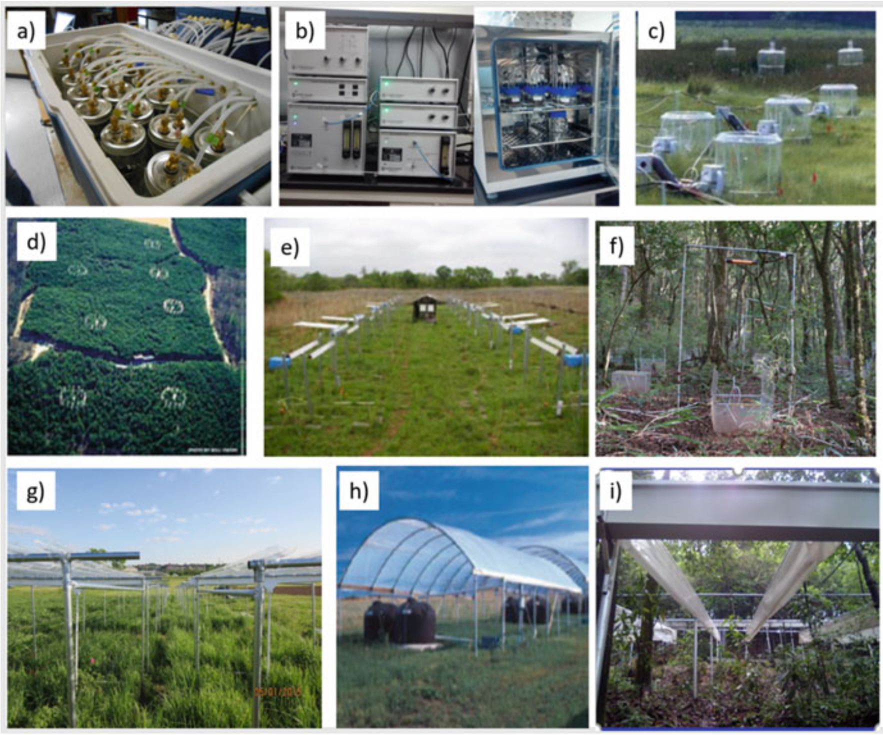

Fig. 4 Examples of laboratory incubation and field experimental facility in global change research. (**a**) Lab incubation in the Permafrost carbon network. (Photo credit: R. Bracho). (**b**) Laboratory incubation under controlled temperature and moisture in the incubator at DOE ORNL, soil CO_2 and CH_4 emissions are continuously measured using the Micro-Oxymax respirometer. (Photo credit: Melanie Mayes). (**c**) Open-top chambers (OTCs) on a Florida Scrub-oak Ecosystem. (Photo credit: Bert Drake). (**d**) FACE (Free Air CO_2 Enrichment) experiment in Duke loblolly pine forest. (Photo credit: Will Owens). (**e**) Global warming experiment in a tall grass prairie near Norman, Oklahoma. (Photo credit: Yiqi Luo). (**f**) Global warming experiment in a subtropical forest in southwestern China (leading by Y. Zhang). (Photo credit: Naishen Liang). (**g**) Precipitation experiment in a switchgrass field in Nashville, Tennessee. (Photo credit: Dafeng Hui). (**h**) Precipitation shelter in a tallgrass prairie at the Konza Prairie Research Natural Area, Kansas. (Photo credit: Philip A. Fay). (**i**) Precipitation experiment in a subtropical forest in southern China. (Photo credit: Weijun Shen)

chambers or incubators allow full control of temperature and humidity for laboratory experiments (Oertel et al. 2016). Either sieved and homogenized soil or undisturbed soil cores can be used. Undisturbed soil cores can keep soil structure intact and do not disturb microorganisms (Petersen et al. 2013; Yao et al. 2010; Oertel et al. 2016), but there is large heterogeneity among soil cores and a larger sample size is often required (Gritsch et al. 2015). As a result, homogenized soil material is more commonly used in laboratory incubations (Zhou et al. 2014b; Oertel et al. 2016). As soil structure is mostly destroyed during soil processing such as air-drying or sieving, it creates a disturbance and may influence soil microbial activity (Schaufler et al. 2010; Yao et al. 2010; Oertel et al. 2016).

One advantage for laboratory incubation is that multiple levels of treatments can be applied. For example, Zhou et al. (2014b) investigated soil moisture and temperature on soil Rh, and set 5 soil moisture levels: 20%, 40%, 60%, 80%, and 100% water holding capacity (WHC) and 5 temperature levels: 10 °C, 17 °C, 24 °C, 31 °C, and 38 °C. Five soil samples of six replications for each treatment were used and resulted in a total of 600 samples/flasks in the study. As in most soil incubation studies, they used air-dried soil sample (equivalent to 50 g of oven-dried soil) in each triangle flask. Soil water content was adjusted to the corresponding soil moisture level by adding deionized water. Soil temperature was controlled using incubators. The incubation experiment lasted for 90 days. They measured soil Rh rates using a Li-6262 Infrared Gas Analyzer (Li-Cor Inc., Lincoln, NE) on days 1, 2, 3, 4, 6, 7, 13, 18, 27, 34, 41, 53, 62, 74, and 90. Before Rh measuring, each triangle flask was ventilated for 3 min to minimize gas accumulation in the headspace. After ventilation, another type of rubber stoppers with two plastic tubes for gas inlet and outlet was used to seal the flask and the tubes were connected to Li-6262 for measuring headspace CO_2 concentration. The CO_2 concentration in the headspace was recorded every second for 2 min and Rh rate was calculated using the linear portion of the response curve of CO_2 concentration versus time.

For laboratory incubations, different methods can be used to monitor soil GHG emissions. For example, gas analyzers such as the Micro-Oxymax Respirometer system can be used for CO_2 and CH_4 measurements, and Picarro G2508 for soil CO_2, CH_4, and N_2O emissions (Figs. 3h, 4b). When connected to the multiplexer, multiple flasks/chambers can be used to measure different chambers/flasks. GHG concentration changes in each chamber can be measured, and soil GHG emission rates and cumulative soil GHG emissions can be calculated.

Field Experiment

Many manipulated experiments of global climate change have been conducted under field conditions (Luo et al. 2001; Fay et al. 2003; Ren et al. 2017; Song et al. 2019) (Fig. 4c–i). An international research coordination network, Terrestrial Ecosystem Response to Atmospheric and Climatic Change (TERRAC), includes 135 field experimental sites in 25 countries. In a comprehensive review, Song et al. (2019) collected data from 1119 manipulative experiments in 2230 peer-reviewed studies. These studies include single, two and multiple climatic factors experiments and consider atmospheric CO_2 concentration, temperature, precipitation, and N addition. So far, global climate change studies have covered all different ecosystems such as tropical forest, deciduous forest, grassland, wetland, and desert, and different climatic factors. But the coverage of ecosystems and climatic factors are not uniform. There are more studies focused on global warming and precipitation studies and in grassland ecosystems.

For field experiments, most studies use a perturbation approach that creates different treatment levels (i.e., magnitudes in changes of treatment factors) that are large enough to generate detectable ecosystem responses (Luo and Reynolds 1999). To evaluate atmospheric CO_2 concentration effects, ecologists usually increase CO_2 concentration to a much higher levels, such as 200 and 400 ppm above current level, using growth chambers, open-top chambers, and Free Air CO_2 Enrichment (FACE) facilities (Norby and Luo 2004; Ainsworth and Long 2005) (Fig. 4c, d). The responses of terrestrial ecosystem processes such as soil GHG emissions can be monitored and investigated.

Since soil GHG emissions are closed related to global warming, many studies have been conducted with different experimental facilities. The common methods used to manipulate temperature include soil warming using heating cables, infrared heaters/lamps, passive heating using doom/shelters, and open-top chambers with heated air (Bergh and Linder 1999; Harte et al. 1995; Luo et al. 2001; Beier et al. 2004) (Fig. 3e, f) Most studies raise soil/air temperature by 1–4 °C higher in treatment plots than that in the control. Melillo et al. (2011) compared different warming facilities and discuss the pros and cons of them.

Precipitation change is another factor significantly influence soil CO_2, CH_4 and N_2O emissions, and has been widely studies in grasslands, forests, and other ecosystems (Fay et al. 2003; Zhou et al. 2016b; Deng et al. 2017). Most studies change the intensity of the precipitation, particularly the reduced precipitation (drought), and a few studies investigate the precipitation pattern impacts. For precipitation studies, rain-out shelter and precipitation interception-redistribution system are commonly used in field experiments (Fig. 4g–i).

Multiple factor global change experiments are still logistically and financially challenging. Only a small number of experiments considered multiple climatic factors in the same study (Zhou et al. 2016a; Song et al. 2019). But these studies quantify not only the main effects of climatic factors but also the interactive effects of these factors. Results from some experimental and modeling studies also demonstrate interactive responses to combinations of treatments and underscore the need for multi-factor experiments at different ecosystems and over long term (Zhou et al. 2008; Randerson et al. 2009).

The major advantages of experimental study include: (1) to reveal true ecosystem responses to climate change and help understand mechanisms underlying these changes. As other factors are controlled and kept at relative same levels, the results are considered as the direct effects of climatic change. Since only one or a few climatic factors are manipulated, it is relative to track the influences of climatic factors on ecological processes and components; (2) as treatment levels can be set at different levels, the effects of climate change factors in the past, current and future can be evaluated. The disadvantages of field experiments include: (1) short terms and small spatial scales. Majority of the experiments last less than 5 years and cover a small spatial area or ecosystem. The results from these studies may be transient, and both magnitude and direction of responses may change over time (Fay et al. 2003; Luo 2007). (2) Artifacts caused by experimental facilities. Experimental manipulation methods may have some unintended or undesirable changes such as disturbance to soil and plants (i.e., heating cables of warming) and changes in light and wind conditions (i.e., plastic roofs in precipitation facilities) (Beier et al. 2004). (3) Climatic factors change simultaneously, but most experiments can only consider one or a few climatic factors and set two or a few treatment levels.

Meta-Analysis

Meta-analysis is a quantitative review that synthesizes results from multiple independent studies to address a common question or to test a common hypothesis (Curtis 1996; Curtis and Wang 1998; Rustad et al. 2001; Luo et al. 2006; Lei et al. 2007). Since the early 1990s, the use of meta-analysis in the field of ecology and global climate research has increased exponentially. In 2007, Lei et al. (2007) reviewed the applications of meta-analysis in global change research. Recently, the number of meta-analysis increases dramatically due to a large amount of experimental data have been accumulated over the last several decades. Here we provide a brief overview of the procedure of the method and discuss the advantages and disadvantages. The detailed methods of meta-analysis can be obtained in Curtis and Wang (1998), Hedges et al. (1999), Luo et al. (2006), and Lei et al. (2007).

There are several steps in meta-analysis: formulating a research question, collecting and coding data, analyzing data, and interpreting the results (Lei et al. 2007). One important step in meta-analysis is to identify a knowledge gap in the research areas and generate a research question to be addressed. After that, data collection from relevant individual studies are a critical step. A well conducted meta-analysis should be able to collect most available data. Criteria for inclusion of studies should be explicitly documented. After quality control (e.g., delete these with missing data), data need to be organized and coded. To analyze data, effect size metrics and analysis models need to be selected. In most of meta-analysis, response ratio (RR, the ratio of means for a measured variable between the treatment group and the control group) is used as an index of the estimated magnitude of the treatment effect (Hedges et al. 1999; Luo et al. 2006). The significance of RR can be statistically tested to determine whether a response variable of the treatment group is different from that of the control group. The heterogeneity of RR is often calculated to examine whether all studies share a common magnitude of the treatment effect. Finally, the RR is grouped according to independent variables (e.g., vegetation type and time after treatment) for the purpose of detecting the differences in RRs among groups. Sometimes, the bias of data selection and publication bias need to be tested.

Meta-analysis allows a more objective assessment of many individual research results and provides a more precise overall estimate of a treatment effect. It is often more powerful to detect true effects and can explain heterogeneity between the results of individual studies. Meta-analysis often does not provide much novel information. There are still some debates over the meta-analysis such as mixing experiments with different background information, biased estimates of effects due to publication bias (e.g., negative results are often not published), and pooling of heterogeneous studies with different qualities. As demonstrated by Hungate et al. (2009), results from different meta-analyses are not always conclusive neither, due to different inclusions of field experimental studies and data. But when applied adequately, meta-analysis can generate quantitative conclusions on some controversial issues and provide some new insights and research directions (Luo et al. 2006; Lei et al. 2007).

Biogeochemical and Ecosystem Modeling

Biogeochemical and ecosystem models are powerful tools examining the impacts of global climate change on soil GHG emissions in terrestrial ecosystems (Luo et al. 2008; Xu et al. 2012a; Bridgham et al. 2013; Deng et al. 2016; Ito et al. 2018). Many process-based biogeochemical models have been developed to simulate the processes responsible for production and transport of CO_2, CH_4, or N_2O (Li et al. 1992; Del Grosso et al. 2006; Tian et al. 2019). Some models only consider soil CO_2 emission, while other considered two or three GHG. These models can not only investigate multiple interacting factors of global change, but also scale experimental results up in time and space to quantify regional and global GHG emission and forecast GHG emissions in the future (Cao and Woodward 1998; Tian et al. 2019).

Several ecosystem models have been developed at regional and global scales (Potter et al. 1993; Cao and Woodward 1998; Zhuang et al. 2007; Riley et al. 2011; Tian et al. 2016; Ma et al. 2017). The Carnegie-Ames-Stanford Approach (CASA) biosphere model (Potter et al. 1993) was designed to study C-climate feedback and able to estimate C and N trace gas emissions at the global scale. The daily version of the CENTURY model (DAYCENT) (Parton et al. 1998) and the Denitrification Decomposition Model (DNDC; Li et al. 1992) were developed to study the impacts of various agricultural practices and global change on soil CO_2, CH_4, and N_2O emissions. TECO is a biochemical and ecophysiological model that use daily meteorological data to simulate ecosystem C dynamics (Luo et al. 2008). The Dynamic Land Ecosystem Model (DLEM) considered the biotic and abiotic processes that regulate ecosystem CO_2, CH_4, and N_2O fluxes in natural and managed soils (Tian et al. 2019). Other models, such as the Organizing Carbon and Hydrology in Dynamic Ecosystems (ORCHIDEE) with N cycle (O-CN; Zaehle et al. 2011) and Community Land Model with prognostic C and N (CLMCN)-N_2O (Saikawa et al. 2014), have been developed to simulate ecosystem C and N and energy balances including soil CO_2 and N_2O emissions from terrestrial ecosystems. Those models have different model structures and various complexities, but the major biological processes are included in these models. The forcing factors are also different for different models, but temperature and precipitation are the most important factors that included in all models. Here we used the DNDC and DLEM as examples to illustrate model structure, processes involved and model functions.

The DNDC Model

The DeNitrification-DeComposition (DNDC) model is one of the process-based models and has been successfully applied to both small plots and regional studies with different crop types in many places around the world (Li et al.; Beheydt et al. 2007; Deng et al. 2016). The DNDC model includes all important C and N processes influencing CO_2, CH_4, and N_2O production and transportation, making it possible to assess and predict the impacts of different agricultural management practices and climate change on soil CO_2, CH_4, and N_2O emissions (Fig. 5a). The detailed description of the model can be found in Li et al. (1992, 1996).

The DNDC model version 95 (http://www.dndc.sr.unh.edu) can be used to simulate and evaluate soil CO_2, CH_4, and N_2O emissions from the different terrestrial ecosystems such as different croplands, grasslands, and forests. The DNDC model is a biogeochemical model, originally developed for estimating N_2O emissions from agricultural fields (Li et al. 1992). It has now been extended to estimating C and N processes such as CO_2, NO, CH_4 and NH_3 emissions, soil organic C (SOC) dynamics and crop yields (Li et al. 2000; Deng et al. 2016). The DNDC model contains six sub-models: soil climate, crop growth, decomposition, nitrification, denitrification and fermentation. The functional equations of the six sub-models are primarily derived from basic physical, chemical and biological theories or from empirical relationships based on observed data. Thus, the simulated CO_2, CH_4, and N_2O emissions by the DNDC model are primarily regulated by soil environmental variables (e.g., soil temperature and WFPS) and substrates (e.g., DOC and inorganic N). The DNDC model can be used to simulate CO_2, CH_4, and N_2O emissions at the site or at a regional scale, and the simulations with the site mode enable comparison against field observations.

The model requires the following input data: (1) local meteorological data (e.g., daily air temperatures and precipitation); (2) initial soil physical and chemical properties [e.g., SOC, soil inorganic N, soil bulk density, pH, and soil texture]; (3) agricultural management information (e.g., crop parameters, tillage, fertilization and irrigation). Outputs of the model included soil C and N pools and fluxes, crop leaf, stem and root biomass C and N, and crop yield.

Dynamic Land Ecosystem Model (DLEM)

DLEM is a process-based ecosystem model that simulates the fluxes and storage of C, N and water among/within the terrestrial ecosystem components with consideration of multiple natural and anthropogenic perturbations (e.g., climate change, CO_2 concentration, atmospheric composition, land use and management practices), working at multiple scales in time from daily to yearly and space from meters to kilometers, from region to globe.

The DLEM includes five core components (Fig. 5b): (1) biophysics, (2) plant physiology, (3) soil biogeochemistry, (4) dynamic vegetation, and (5) disturbance, land use and management. Briefly, the biophysics component simulates the instantaneous fluxes of energy, water, and momentum within land ecosystems and their exchanges with the surrounding environment. The plant physiology component

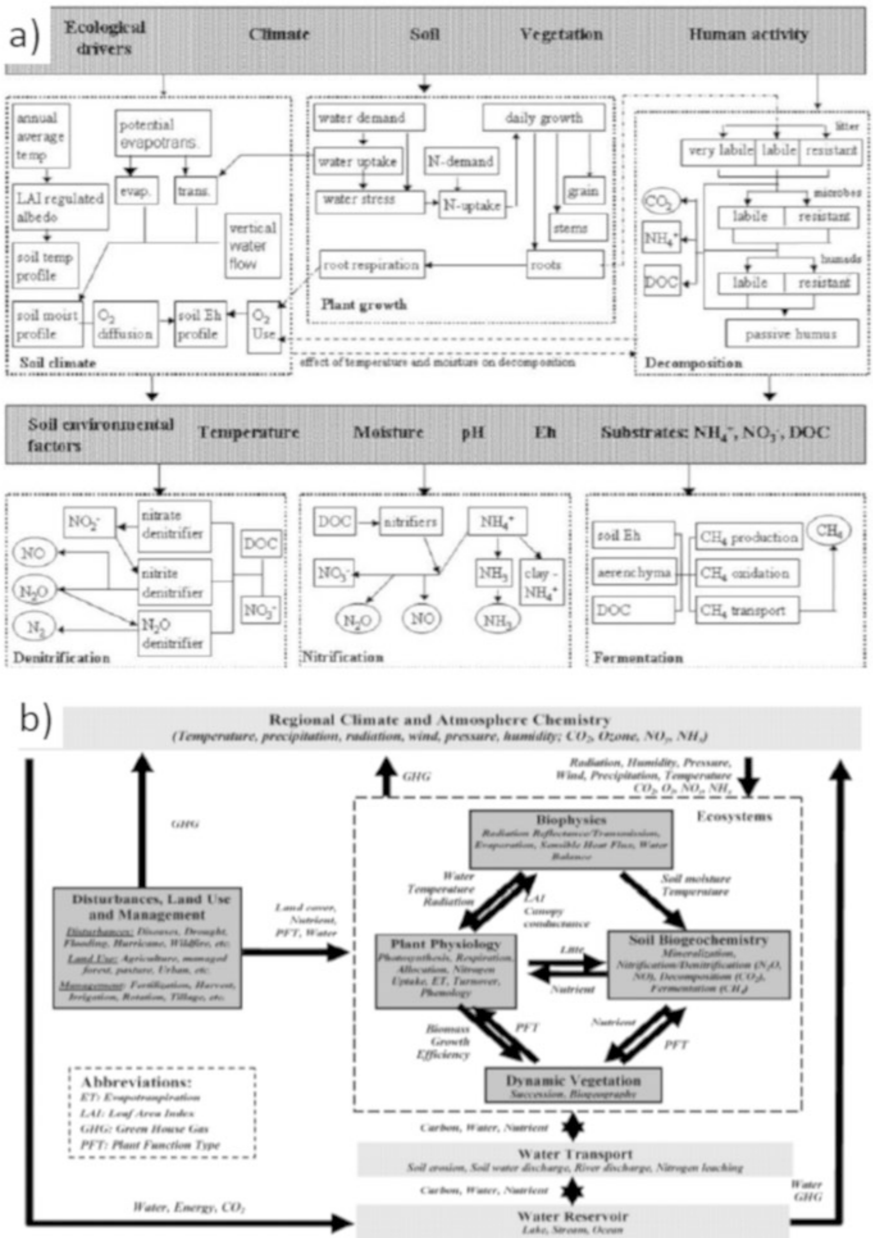

Fig. 5 Framework of the Decomposition and Denitrification (DNDC) (**a**). (Adapted from Li et al. 1996) and Dynamic Land Ecosystem Model (DLEM) (**b**). (Adapted from Tian et al. 2010). Similar mechanisms are applied for greenhouse gas emissions in both models

simulates major physiological processes, such as plant phenology, C and N assimilation, respiration, allocation, and turnover. The soil biogeochemistry component simulates the dynamics of nutrient compositions and major microbial processes. The biogeochemical processes, including the mineralization/immobilization, nitrification/denitrification, decomposition, and methane production/oxidation are considered in this component. The dynamic vegetation component simulates the structural dynamics of vegetation caused by natural and human disturbances. Two processes are considered: the biogeography redistribution when climate change occurs, and the recovery and succession of vegetation after disturbances. Like most dynamic global vegetation models, the DLEM builds on the concept of plant functional types (PFT) to describe vegetation distributions. It has been extensively used to study the terrestrial C, water and N cycles around the world in response to global change, and the detailed assumptions and processes are well documented in previous work (Tian et al. 2016, 2019).

Impacts of Global Climate Change on Soil Greenhouse Gases Emissions: Case Studies

Laboratory Incubation and Field Experiment

Laboratory Incubation

Laboratory incubations have been widely used in global change studies to investigate changes in climatic factor on soil GHG emissions (Oertel et al. 2016). Most of these studies tested the impacts of temperature, soil moisture, and C and nutrients additions on soil CO_2, CH_4, and N_2O emissions. Soils are mostly collected from grasslands, forests, wetlands, and croplands. As there is no live root in soil incubation studies, soil CO_2 emission measured is Rh. More studies have been done on soil CO_2 emission (or soil respiration, Rs) than soil CH_4 and N_2O emissions. For example, Reichstein et al. (2000) conducted a laboratory incubation of soil samples at three temperature levels (5 °C, 15 °C, and 25 °C) over 104 days, and displayed typical soil CO_2 response curves for both soil types (gully: A-horizon; ridge: Oe-/Oa-layer) (Fig. 6). In both soils, the cumulative Rs increases and Rs decreases substantially with incubation time that is typical responses for most soil incubation studies. Rs can be well described by a first-order kinetic two-compartment model and a functional temperature dependence of the rate constants. Rs is higher under high temperature than low temperature. Some incubation studies also reported an initial peak of gas emission, particularly when soil has high SOC or labile C is added (Hui et al. 2020). By incubating soils collected from the Giessen Free Air CO_2 Enrichment (FACE) study, Abbasi and Müller (2011) found that soil CO_2 fluxes are approximately 20% higher under elevated CO_2 than soil from ambient with large variations. CH_4 oxidation rates are increased by 49%. These changes could be caused by more carbon substrates inputs into soil. Elevated CO_2 does not have any significant effect on nitrification enzyme activity while total denitrification is increases by 36%.

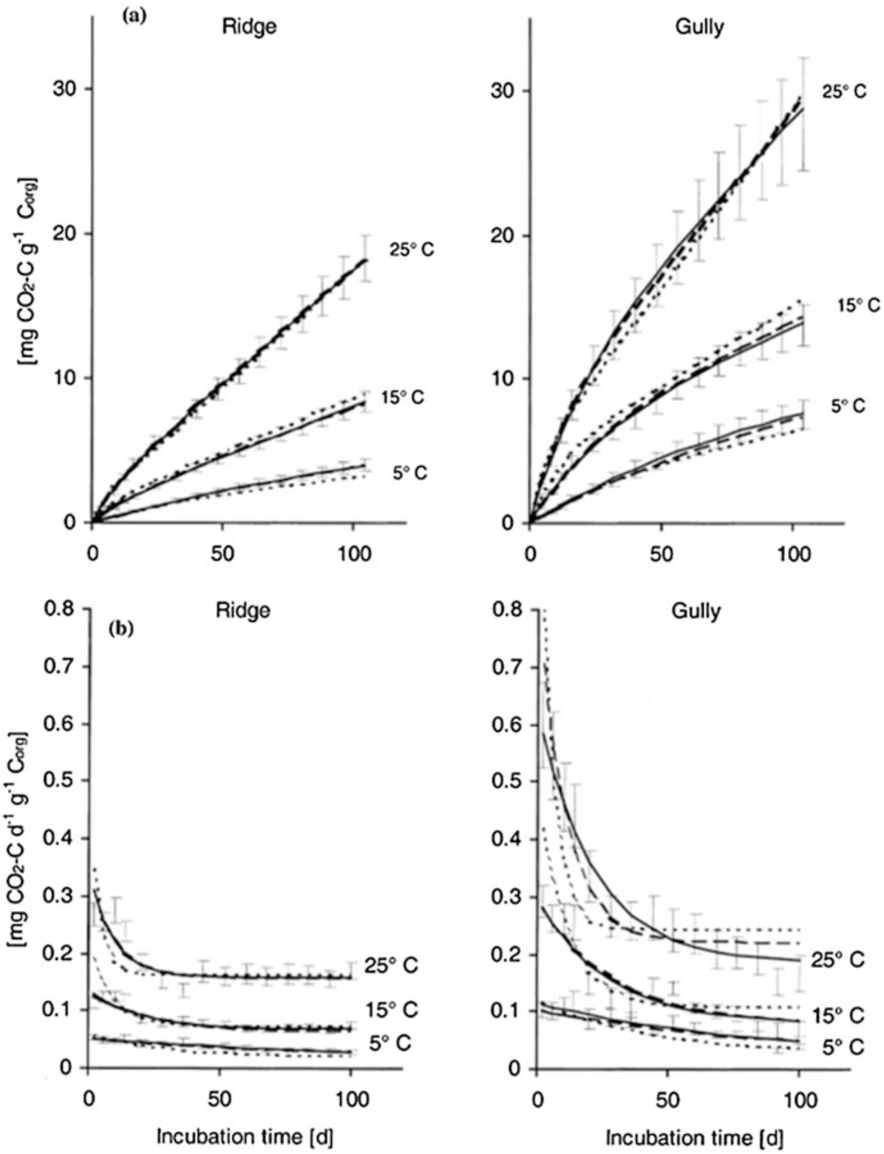

Fig. 6 Measured and modelled cumulative soil CO_2 emission (**a**), and soil respiration rates (**b**) of Gully-Ah- and Ridge-organic-layer soil samples incubated at 5 °C, 15 °C and 25 °C. (Adapted from Reichstein et al. 2000)

Zhou et al. (2014b) conducted a 90-day laboratory incubation experiment using a subtropical forest soil with a full factorial combination of 5 moisture levels (20%, 40%, 60%, 80%, and 100% water holding capacity – WHC) and 5 temperature levels

(10 °C, 17 °C, 24 °C, 31 °C, and 38 °C). Microbial biomass C (MBC), microbial community structure and soil nutrients and soil CO_2 emission were measured. Results showed that soil CO_2 emission increases with soil temperature, following an exponential model, as in most studies. Q_{10} is calculated based on the exponential model for each soil moisture level and it is significantly lower at lower moisture levels (60%, 40% and 20% WHC) than at higher moisture level (80% WHC) during the early stage of the incubation. Soil CO_2 emission has the highest value at 60% WHC and the lowest at 20% WHC. They found that variations of Q_{10} are significantly associated with MBC during the early stages of incubation, but with the fungi-to-bacteria ratio during the later stages, suggesting that changes in microbial biomass and community structure contribute to the changes in Q_{10}. By synthesizing laboratory incubation data from soils acquired at 73 sites, Xu et al. (2016) showed that incubation temperatures significantly affect soil C decomposition rates of the active and the slow C pools, following the quadratic polynomial models (Fig. 7a). After normalizing the decomposition rate with temperature, decomposition rates significantly decline with water holding capacity (Fig. 7b). They considered that the impacts of WHC are significantly related to clay content of soil, as the higher the percentage of clay content, the higher WHC. With increasing WHC and clay content, soils hold more water and less soil O_2 supply, and lower soil CO_2 emission.

As different soil preparation methods are used in soil incubation studies, Herbst et al. (2016) quantified the influence of air-drying and sieving on the soil Rh response to soil water content, and found that the incubation of sieved and intact soils reveals distinct differences in the response of soil CO_2 emission to soil water content. The sieved soils have a threshold-type pattern, whereas the undisturbed soils show a quadratic increase with increasing effective soil water saturation. They concluded that the destruction of soil structure by sieving hampers the transferability of measured soil moisture response of soil CO_2 emission to real-world conditions. Future studies need to consider the impacts of soil preparation on soil GHG emissions when designed the laboratory incubation experiments.

Field Experiment

The effects of global climate change factors such as global warming, precipitation, atmospheric CO_2 concentration change and soil CO_2, CH_4, and N_2O emissions have been conducted using field manipulative facilities over the past several decades (Rustad et al. 2001; Luo et al. 2001; Zhou et al. 2014a; Yan et al. 2018; Song et al. 2019; Liu et al. 2020). Most of the elevated CO_2 and precipitation experiments focused on ecosystem C processes such as plant growth, biomass and productivity, litter decomposition, and soil and ecosystem C sequestration, while global warming studies focused more on CO_2 emissions. In wetland and rice fields, soil CH_4 emission has been widely studied while soil N_2O emission is the focus of the croplands with N fertilization. Emissions of GHG from soils vary among different ecosystems (Oertel et al. 2016) (Fig. 8). Wetlands have the highest mean absolute GHG emission rates; higher than forests, grasslands, and barren soils. But there are large variations of GHG emissions in all ecosystems, due to differences in climate

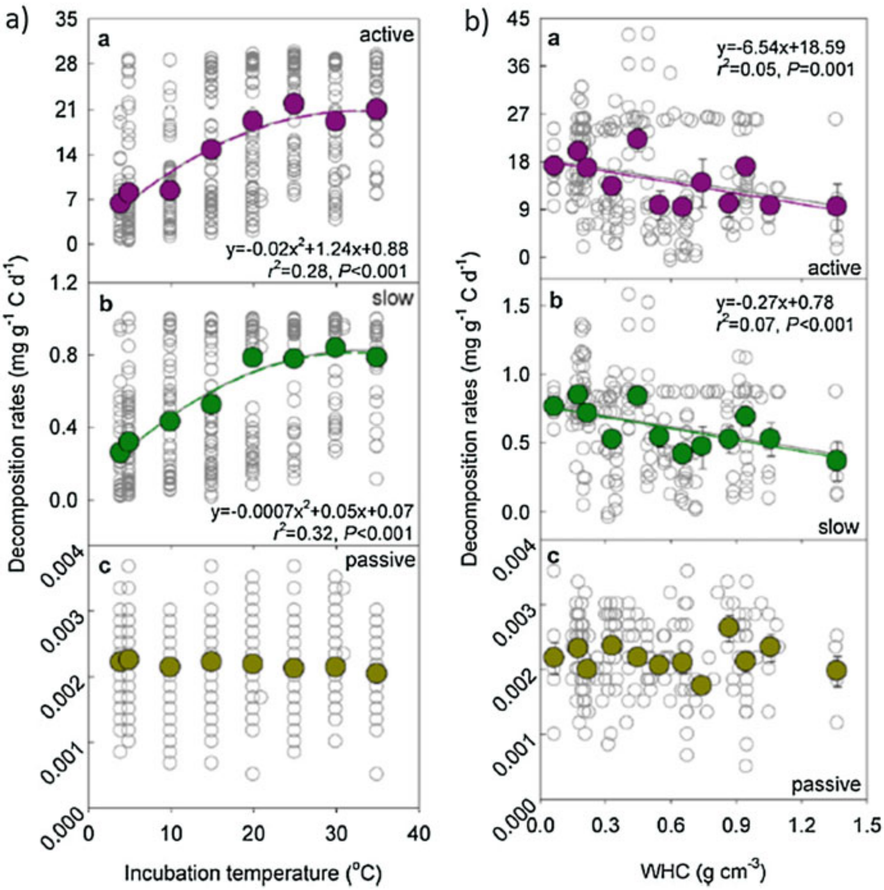

Fig. 7 Relationships of incubation temperatures (n = 376, left panel) and water holding capacity (WHC, n = 197, right panel) with carbon decomposition rates of the active (**a**), slow (**b**), and passive (**c**) soil organic carbon pools. (Adapted from Xu et al. 2016)

zone and land management-related conditions. Mean GHG emission is about 300 mg CO_2 equivalents m^{-2} h^{-1} for all ecosystems.

Elevated CO_2 mostly enhances soil CO_2 emission and soil CH_4 emission and has different impacts on soil N_2O emission. For example, Zak et al. (2000) reported that soil CO_2 emission in grassland ecosystems is increased by 51% under elevated CO_2 compared to ambient CO_2. Smith et al. (2010) reported that soil CO_2 emission in an arable soil is increased by 15–50% under elevated CO_2. Cheng et al. (2006) reported a 58% increase in CH_4 flux from rice fields under elevated CO_2 and attributed the increase to greater root exudates and numbers of tillers (Abbasi and Müller 2011). Bhattacharyya et al. (2013) found that elevated CO_2 (500 ppm, open-top chamber – OTC) enhances soil CH_4 and N_2O emission in tropical rice by 26.0% and 24.6%, as total organic C in root exudates is enhanced and labile soil C and N, and readily

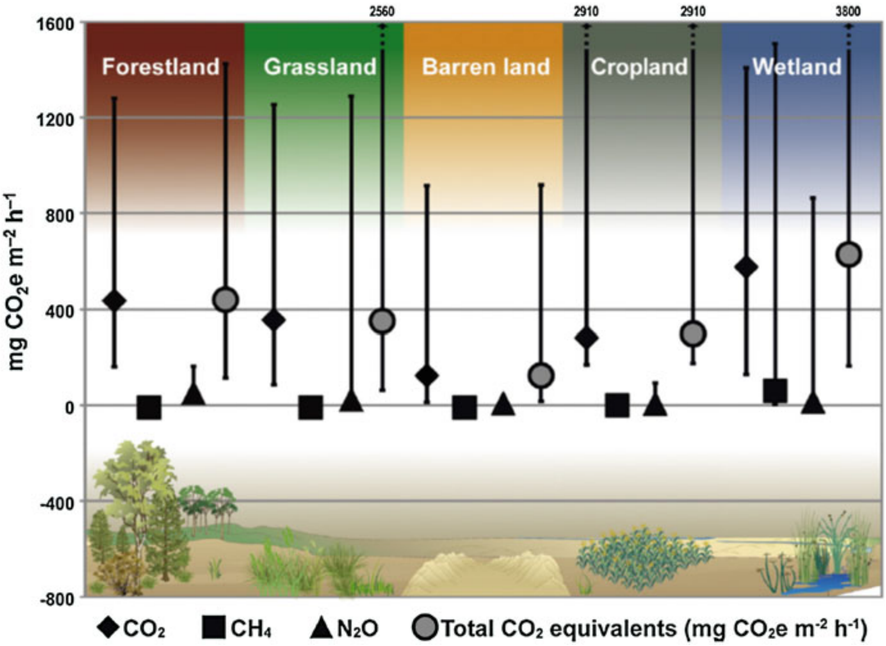

Fig. 8 Greenhouse gas emissions (CO_2-eq) of CO_2, N_2O and CH_4 from soils with different land cover: grassland ($n = 47$), forestland ($n = 22$), barren land ($n = 17$), cropland ($n = 41$), and wetland ($n = 67$). (Adapted from Oertel et al. 2016). Large variations are found for CO_2 emission and total emission in all ecosystems and N_2O emission in grasslands and wetlands

mineralizable C are enhanced under the elevated CO_2. Elevated CO_2 also increases methanogens and denitrifier population thus soil N_2O emission. However, no responses or negative responses have also been reported in different ecosystems. For example, Carter et al. (2011) found that elevated CO_2 reduces CH_4 uptake in a temperate heathland but does not influence soil N_2O emission. Ineson et al. (1998) measured fluxes of CO_2, CH_4, and N_2O from soils of *Lolium perenne* under ambient and elevated CO_2 at the Swiss FACE experiment plots and reported that elevated CO_2 increases N_2O emissions by 27% but inhibits CH_4 oxidation (Abbasi and Müller 2011). Mosier et al. (2002) conducted an OTC CO_2 enrichment study in the Colorado shortgrass steppe and reported that none of the trace gas fluxes are significantly altered by CO_2 enrichment over the 43 months period of observation.

The experimental warming studies published to date have suggested significant increases in soil CO_2 and N_2O emission but showed different effects on soil CH_4 uptake and emission. As temperature increases, Rs generally increases (Luo et al. 2001), because warming generally directly increases both Rs and Rh. For example, in a warming experiment using buried heating cables, Melillo et al. (2011) reported that 5 °C temperature increase has resulted in a cumulative net loss of C (1.3×10^7 kg C ha^{-1}) from a New England forest relative to a control area over the 7-year study. Warming-induced CO_2 emissions are likely linked to higher microbial activities, root biomass, and enhanced plant C input (Luo et al. 2001; Zhou et al. 2019; Wu et al. 2020). In addition, warming causes a shift in the soil microbial community toward more fungi, which are more tolerant to high soil temperature and dry environments than bacteria. Soil warming may suppress soil CO_2 emission as high temperature induces moisture stress and thus decreases SOM mineralization (Liu et al. 2020; Wu et al. 2020). Global warming effects on soil CH_4 may vary with different ecosystems. Warming may increase CH_4 uptake under semiarid soil environments (Wu et al. 2020) as soil drought could enhance oxygen diffusivity and stimulate the oxidation of soil CH_4 (Dijkstra et al. 2012; Wu et al. 2020). But warming may also enhance net CH_4 emission in wetlands and rice fields due to enhanced aerobic decomposition and increased root biomass production (Dijkstra et al. 2012; Wu et al. 2020). For soil N_2O emission, global warming can stimulate the microbial nitrifiers and denitrifiers activities, thus enhance it; but warming may also reduce soil water content, and decrease soil N_2O emission (Tu and Li 2017; Wu et al. 2020).

Precipitation change has the potential to greatly influence soil GHG emissions, as the soil moisture content is the key driver for optimal microbial activity (Petrakis et al. 2017; Wu et al. 2020). Soil moisture influences the SOC decomposition and mineralization, soil aeration, the substrate availabilities, thus the microbial processes of GHG production and consumption (Dijkstra et al. 2012; Wen et al. 2020; Wu et al. 2020). Precipitation manipulation experiments generally show increased soil CO_2 emission following experimental water additions and decreased soil CO_2 emission following rain exclusion (Deng et al. 2017; Wu et al. 2020). But excessive precipitation could reduce soil CO_2 emission (Liu et al. 2020; Wen et al. 2020). For soil CH_4, its uptake is often negatively correlated to soil moisture, as higher soil moisture content decreases soil CH_4 diffusivity for oxidation by methanotrophs, and soil moisture enhances production of CH_4 by methanogens (Del Grosso et al. 2006; Wu et al. 2020). High soil moisture typically enhances soil N_2O emissions due to enhanced denitrification (Dobbie and Smith 2003; Deng et al. 2016; Wu et al. 2020). But anaerobic conditions may increase the reduction of N_2O to N_2, and enhance N_2O consumption, resulting a reduced soil N_2O emission (Wen et al. 2020; Wu et al. 2020).

Atmospheric N deposition and N fertilization have significant and more impacts on soil N_2O emission than CO_2 and CH_4 emissions. Many studies in croplands and nature ecosystems have reported significant increases of soil N_2O emission (Tian et al. 2016, 2019). For example, Castro et al. (1994) measured soil GHG emissions in a mature slash pine plantation under control and urea-N fertilization and found that fertilization significantly increase soil N_2O emission and lowers the atmospheric CH_4 uptake, but does not change soil CO_2 emission. Fertilization causes a shift of the relative activities of the CH_4 oxidizing bacteria from those dominated by methanotrophs in the control soils to those dominated by nitrifying bacteria in the fertilized soils.

Some studies also considered more than one climatic factor in the field experiments. For example, Wu et al. (2020) tested the effects of increased soil temperature (+4 °C, soil heating cables) and increased precipitation (+30% of ambient precipitation) on GHG emissions. They found that soil warming significantly promotes cumulative N_2O and CO_2 emissions by 49% and 39%, respectively. Soil N_2O and CO_2 emissions are also enhanced by 54% and 14% under increased precipitation, respectively. Soil warming increases soil CH_4 uptake by 293%, but CH_4 flux is not influenced by increased precipitation. Overall, soil warming and increased precipitation significantly enhance the GHG budget by 39% and 16%, respectively. Similar results were found in winter wheat-soybean cropping system, as Hu et al. (2019) found that elevated temperature (+2 °C) increases soil CO_2 and N_2O emissions, but reduced precipitation (−30%) decreases soil CO_2 emission, not soil N_2O emission.

Despite these recent advances, only a small number of manipulative long-term field studies have directly assessed the combined effects of multiple climatic factors on soil CO_2, CH_4 and N_2O emissions (Martins et al. 2017; Wu et al. 2020). Evaluating multifactor interactions in influencing soil GHG emission in terrestrial ecosystems is still critical to understanding their response to global change in the real world. Indeed, when interactive effects dominate over the main effects of individual factors, results from single-factor experiments become less useful for understanding ecosystem changes. Only in the case that interactive effects are minor relative to main effects, results from single-factor experiments may become useful in informing us of potential changes of ecosystems in response to multifactor global change. Ecosystem models may be used to address the long-term impacts of multiple factors on soil greenhouse gas emissions.

Meta-Analysis

Many meta-analyses have been conducted in ecology including global change ecology since early 1990s (Curtis 1996; Rustad et al. 2001; Luo et al. 2006; Deng et al. 2015a; Song et al. 2019; Table 1). For example, Curtis (1996) and Curtis and Wang (1998) started to apply meta-analysis on the effects of elevated CO_2 on plant physiology, growth, biomass and C cycling. Arft et al. (1999) and Rustad et al. (2001) synthesized warming effects on tundra plants and Rs. The impacts of precipitation changes have also been synthesized by many studies (Wu et al. 2011; Zhou et al. 2016b). Here we summarized the results of some of these studies related to soil GHG emissions.

For the effects of CO_2 on soil CO_2 emission processes, Dieleman et al. (2010) conducted a meta-analysis of CO_2 effects using FACE and OTC on plant and soil properties in forests (32 sites) and found that soil CO_2 efflux and soil Rh are increased by 19% and 37% respectively. Van Groenigen et al. (2011) synthesized 73, 21, and 24 observations for N_2O, CH_4 in rice paddies and wetlands, respectively, and found that elevated CO_2 stimulates N_2O emissions by 18.8%, CH_4 emissions by 13.2% in wetlands, and by 43.4% in rice paddies. In upland ecosystems increased CO_2 causes a small and insignificant net uptake of CH_4.

For the effects of warming on GHG emissions, most studied focused on soil CO_2 emission. Rustad et al. (2001) synthesized results from 32 study sites and found that

Table 1 Meta-analyses on global climate change and soil greenhouse gas emissions in terrestrial ecosystems

Reference	Sample size	Treatment factor	Effective size	Greenhouse gas emission	Ecosystem or scale
Dieleman et al. (2010)	51 sites	CO_2	Response ratio	CO_2: +19%	Terrestrial ecosystems
Song et al. (2019)	1119 studies	CO_2 (E), temperature (W), increased precipitation (IP), reduced precipitation (DP), nitrogen (N)	Response ratio	CO_2: E, +28%; W, +18%; IP, +21%; DP, −22%; N, 0%	Terrestrial ecosystems
Van Groenigen et al. (2011)	73 (N_2O) and 45 (CH_4) sites	CO_2	Response ratio	CH_4: +13.2% (Wetlands); +43.4% (Rice paddies) N_2O: +18.4%	Terrestrial ecosystems
Xiao et al. (2018)	133 sites	CO_2, temperature, increased precipitation, reduced precipitation, nitrogen	Response ratio	EEA	Terrestrial ecosystems
Zhou et al. (2016a)	150 studies	CO_2, temperature, irrigation, nitrogen	Response ratio	CO_2: E, +28.6%; W, +7.1%; IP, +9.7%; N, +8.8%	Terrestrial ecosystems
Garcia-Palacios et al. (2015)	330 studies (75 papers)	CO_2, temperature, nitrogen	Hedges' d index	Microbe, C and N cycles	Terrestrial ecosystems
Bai et al. (2013)	51 papers	Temperature	Hedges' d	N_2O: not significant	Terrestrial ecosystems
Chen et al. (2018)	56 studies	Temperature	Response ratio	CO_2: +15.8%	Terrestrial ecosystems
Chen et al. (2020)	65 papers	Temperature	Response ratio	CO_2: +14.3%	Alpine grasslands in Tibetan Plateau
Dai et al. (2020)	134 publications	Temperature	Response ratio	N_2O: about +80%	Terrestrial ecosystems

(continued)

Table 1 (continued)

Reference	Sample size	Treatment factor	Effective size	Greenhouse gas emission	Ecosystem or scale
Liu et al. (2020)	164 publications	Temperature	Response ratio	CO_2: +12.9%; CH_4, emission, 23.4–37.5% Uptake, +13.8%; N_2O: +35.2%	Terrestrial ecosystems
Lu et al. (2013)	66 sites	Temperature	Response ratio	CO_2: +8.98%	Terrestrial ecosystems
Romero-Olivares et al. (2017)	19 studies (25 sites)	Temperature	Response ratio	CO_2: +48% (declined with time)	Terrestrial ecosystems
Rustad et al. (2001)	32 sites	Temperature	Hedges' d index	CO_2: +20%	Terrestrial ecosystems
Wang et al. (2014)	202 sites	Temperature	Response ratio	CO_2: +12%	Terrestrial ecosystems
Wang et al. (2019)	70 sites	Temperature	Response ratio	CO_2: +15.4%	Grasslands
Yan et al. (2020)	115 studies	Temperature	Response ratio	CO_2: +14.54%	Terrestrial ecosystems
Li et al. (2020)	46 studies	Temperature, precipitation	Response ratio	N_2O: W, +33%; IP, +55%; DP, −31%	Terrestrial ecosystems
Wu et al. (2011)	27 papers	Temperature, precipitation	Hedges' d	CO_2: W, +12.0%	Terrestrial ecosystems
Zhou et al. (2019)	184 studies (48 papers)	Grazing, temperature, precipitation, nitrogen	Response ratio	CO_2: W, +2.12%; P, +13.44%; D, −21.0%; N, +5.49%	Grasslands
Homyak et al. (2017)	37 studies	Precipitation (rain-out shelters, DP)	Hedge's d	N_2O: DP, −0.76	Terrestrial ecosystems
Liu et al. (2016)	113 (IP), 91 (DP), and 14 (long drought) studies	Precipitation	Response ratio	CO_2: IP, +16%; DP, −17%; long drought, −6%	Terrestrial ecosystems

Ren et al. (2017)	70 studies	Precipitation	Response ratio	CO_2: IP, +16.68%; DP, 0%	Terrestrial ecosystems
Ren et al. (2018)	45 papers (114 studies)	Precipitation	Response ratio	Microbe	Terrestrial ecosystems
Yan et al. (2018)	84 studies	Precipitation	Response ratio	CO_2: IP, +112.2%, DP, −8.6%; CH_4 uptake, IP, −41.4%; DP, +32.4%; N_2O: IP, +154.0%; DP, −64.7%.	Terrestrial ecosystems
Zhou et al. (2016b)	179 papers	Precipitation	Response ratio	CO_2:IP, +21.9%; DP, −15.6%	Terrestrial ecosystems
Aronson and Allison (2012)	33 sites	Nitrogen	Mean study index	N_2O: −55%	Non-agricultural lands
Bejarano-Castillo et al. (2015)	30 studies	Nitrogen	Response ratio	CO_2: 0%; N_2O: ~8%	Tropical forests
Eagle et al. (2017)	27 studies	Nitrogen	Regression analysis	N_2O increase with N addition	Croplands (Corn)
Liu and Greaver (2009)	109 studies	Nitrogen	Response ratio	CO_2:NEE, +5% forests CH_4 emission: +97% CH_4 uptake: −38% N_2O: +216%	Terrestrial ecosystems
Liu and Greaver (2010)	68 studies	Nitrogen	Response ratio	CO_2: no change	Terrestrial ecosystems
Lu et al. (2011)	257 studies	Nitrogen	Response ratio	CO_2: +16.13% (agriculture), +2.17% (non-agriculture)	Terrestrial ecosystems
Yue et al. (2016)	133 studies	Nitrogen	Response ratio	CO_2: 0%	Terrestrial ecosystems

warming significantly increases Rs by 20%, but with different effective sizes from individual studies (Fig. 9a). The response of Rs to warming is larger in forests compared to tundra and grasslands. Lu et al. (2013) synthesized 66 studies of global warming on Rs and found that Rs is increased by 8.98% by warming, a rate lower than Rustad et al. (2001), with a 7.5% increase in Ra and a 7.5% increase in Rh (Fig. 9b). Soil CO_2 emission is enhanced in all experimental facilities except those with reflective curtain. They reported that MBC is also increased by warming. Yan et al. (2020) recently collected 1131 observations from 115 studies and found that warming stimulates Rs by 14.5%, a value between Rustad et al. (2001) and Lu et al. (2013), and stimulates Rh by 8.42% (Fig. 9c, d). Rs is also enhanced by warming. Responses of Ra and Rh are sometimes different (Fig. 9d). The changes in soil CO_2 emission and soil MBC are correlated. Wang et al. (2014) synthesized 202 datasets from 50 ecosystem and found that warming by 2 °C increases Rs by 12% during the early warming years, but warming-induced drought partially offset this effect. Rh and Ra also show distinct responses. Warming does not change Ra, but increases Rh by an average of 21%, and this stimulation remains stable over the warming duration. Wang et al. (2019) analyzed 622 observations from 70 sites and found that experimental warming stimulated Rs by 9.5%. The stimulatory effect of warming on Rs observed in short-term studies ($\leq$3 years) disappears in longer-term experiments (>3 years). Similarly, Romero-Olivares et al. (2017) reviewed 25 field experiments that examined warming effects on Rs and microbial biomass, and reported that warming initially increases Rs, but the magnitude of this effect declines significantly. After 10 years of warming, Rs in the warmed plots is similar to the controls.

In a meta-analysis of warming effects on soil N_2O emission, Bai et al. (2013) collected 528 observations from 51 papers and investigated experimental warming effects on variables related to N dynamics and pools in terrestrial ecosystems. They found that a non-significant mean effect size of 0.128 of soil N_2O emission by warming, using 26 studies reported soil N_2O emission. Dai et al. (2020) synthesized 1270 observations from 134 publications and found that warming increases soil nitrification and denitrification rates, leading to an increase in N_2O emissions (up to 227%), with or without plants, and for all temperature and moisture conditions, in all ecosystems except others, and for all experimental lengths (Fig. 10a). Li et al. (2020) synthesized 46 published studies of N_2O fluxes and found that temperature increases N_2O emissions by 33% (Fig. 10b). The effects of warming vary with different biomes, and warming methods. OTC has the highest and significant effects. Liu et al. (2020) synthesized 1845 measurements from 164 publications related to the warming effects on all three GHG emissions, and found that warming significantly enhances all of them, with the largest increases in soil CH_4 release in wetlands, and soil N_2O emission (Fig. 11a, b). About 1.5 °C of experimental warming significantly stimulates Rs by 12.9%, CH_4 by 23.4% in rice paddies and 37.5% in wetlands, and N_2O emissions by 35.2%. Warming increases CH_4 uptake by 13.8% in upland soils. Warming facility (soil warming vs. air canopy warming) has significant influences on GHG emissions. Warming also increases inorganic N and SOC.

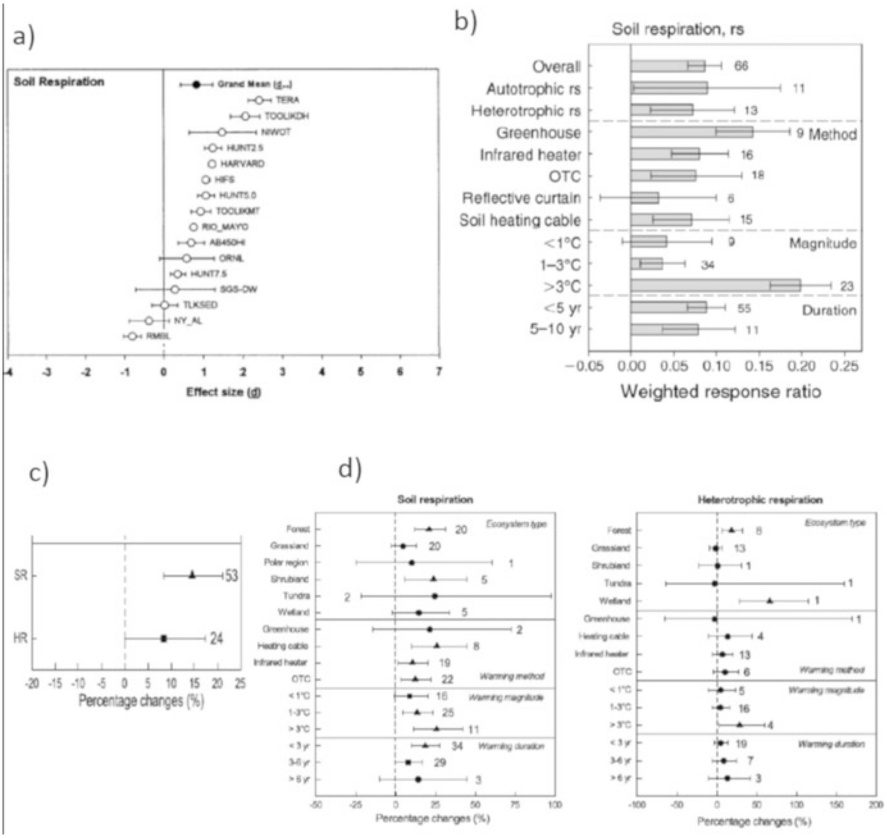

Fig. 9 Examples of warming effects on soil CO_2 emission with meta-analysis. (**a**) Effect size from individual studies. (Adapted from Rustad et al. 2001). (**b**) Response ratio of different components, methods, warming magnitudes and durations. (Adapted from Lu et al. 2013). (**c**) Percentage change of respiration components. (Adapted from Yan et al. 2020). (**d**) Percentage change of soil respiration (left panel) and heterotrophic respiration (right panel) of different ecosystems, warming methods, magnitudes, and durations. (Adapted from Yan et al. 2020). SR and rs, soil respiration; HR, heterotrophic respiration; OTC, open-top chamber

The impacts of precipitation on soil GHG emissions have been well documented using meta-analysis. Liu et al. (2016) conducted a global synthesis of Rs data by collecting data from 113 increased precipitation treatments, 91 decreased precipitation treatments, and 14 prolonged drought treatments. They found that increased precipitation increases Rs by 16% with the greatest increases in arid areas, but decreased precipitation and prolonged drought reduce Rs by −17% and −6%, respectively. They concluded that soil Rs tends to be more sensitive to increased precipitation in more arid areas and more responsive to decreased precipitation in more humid areas. Zhou et al. (2016b) synthesized 179 papers of ecosystem processes under drought and irrigation treatments and found that drought decrease

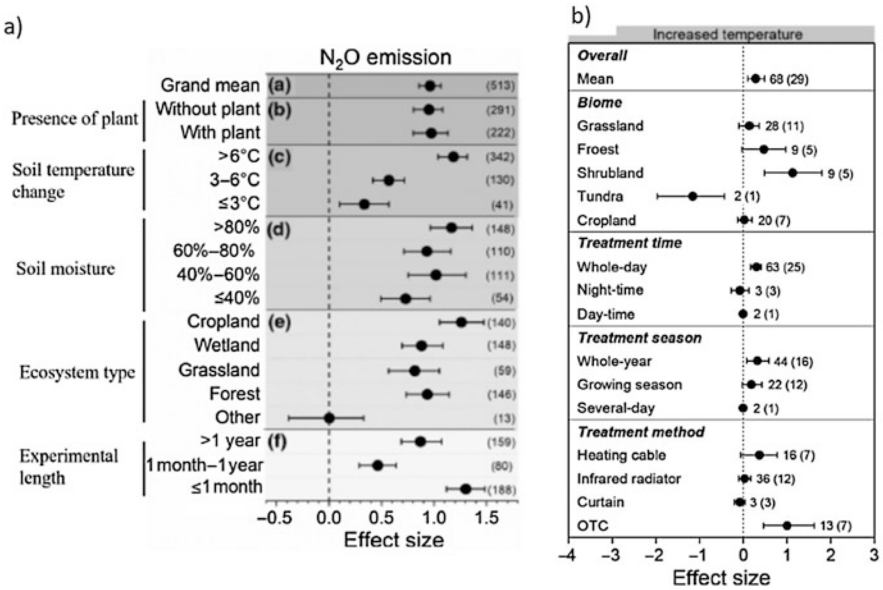

Fig. 10 Examples of warming effects on soil N_2O emission with meta-analysis. (**a**) Effect size of soil N_2O emission of different environmental conditions and ecosystems. (Adapted from Dai et al. 2020). (**b**) Effect size of soil N_2O emission of different biomes, treatment time, season, and method. (Adapted from Li et al. 2020)

Rs and Rh by 15.7% and 34.7%, respectively, and irrigation enhances them by 21.9% and 23.0%, respectively. Yan et al. (2018) performed a meta-analysis with 84 published studies and examined the impacts of altered precipitation on soil GHG emissions (Fig. 12). They found that increased precipitation significantly increases N_2O emissions (+154.0%) and CO_2 fluxes (+112.2%) but decreases CH_4 uptake (−41.4%). Decreased precipitation significantly decreases N_2O emissions (−64.7%) and CO_2 fluxes (−8.6%) but increases CH_4 uptake (+32.4%). Homyak et al. (2017) investigated the effects of reduced precipitation on soil N_2O emissions and found that reducing precipitation significantly lowers soil N_2O emissions across studies, suggesting that denitrification is more sensitive to drought than processes controlling N supply.

The impacts of N addition on soil GHG emission were also synthesized. For example, Liu and Greaver (2009) conducted a meta-analysis of 313 observations across 109 studies and evaluated the effect of N addition on the flux of three major GHGs: CO_2, CH_4 and N_2O. They reported that N addition increases CH_4 emission by 97%, reduces CH_4 uptake by 38% and increases N_2O emission by 216%. Although N addition increases the global terrestrial C sink, the CO_2 reduction is largely offset (53–76%) by N stimulation of global CH_4 and N_2O emission in multiple terrestrial ecosystems. Liu and Greaver (2010) also synthesized 410 observations from 111 papers in multiple terrestrial ecosystems and quantified the responses of belowground C cycling under N addition. They found that N addition

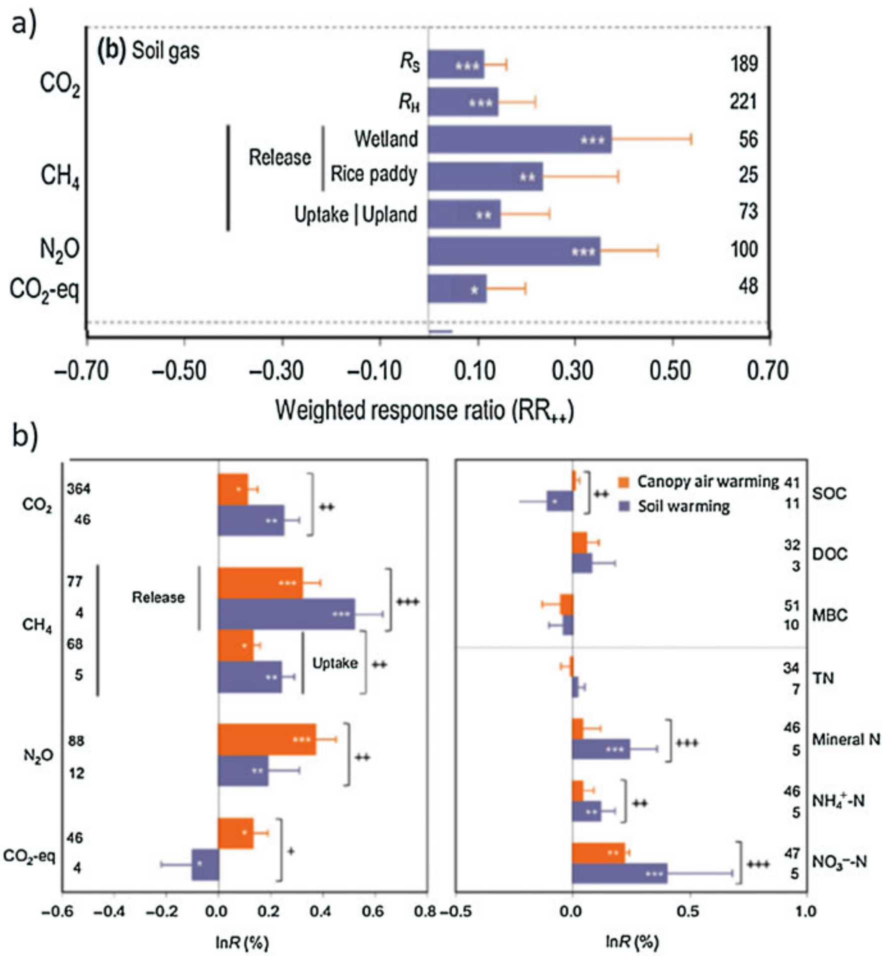

Fig. 11 Global warming effects on soil CO_2, CH_4, and N_2O emission using meta-analysis. (**a**) Overall response ratio. (**b**) Response ratio using different warming method. (Adapted from Liu et al. 2020)

reduces Rh (−8%) and MBC (−20%), but does not change Rs (Fig. 13a). The decreases of soil MBC occur in all ecosystems except the tropical forests. Soil Rh is reduced in temperate conifer forests. Different N application forms and amounts also influence soil MBC, but not soil Rh. Rs is not changed by N addition in all ecosystems (Fig. 13a). Yue et al. (2016) synthesized data from 133 studies from 198 publications and showed that N addition did not significantly change Rs and Rh and in all ecosystems and N forms, and intensities except a reduction of Rs when N application is low (Fig. 13b). N addition also has no significant influence on litter decomposition. Bejarano-Castillo et al. (2015) investigated N addition on C and N cycles in tropical forests and found that N_2O emission is increased in tropical

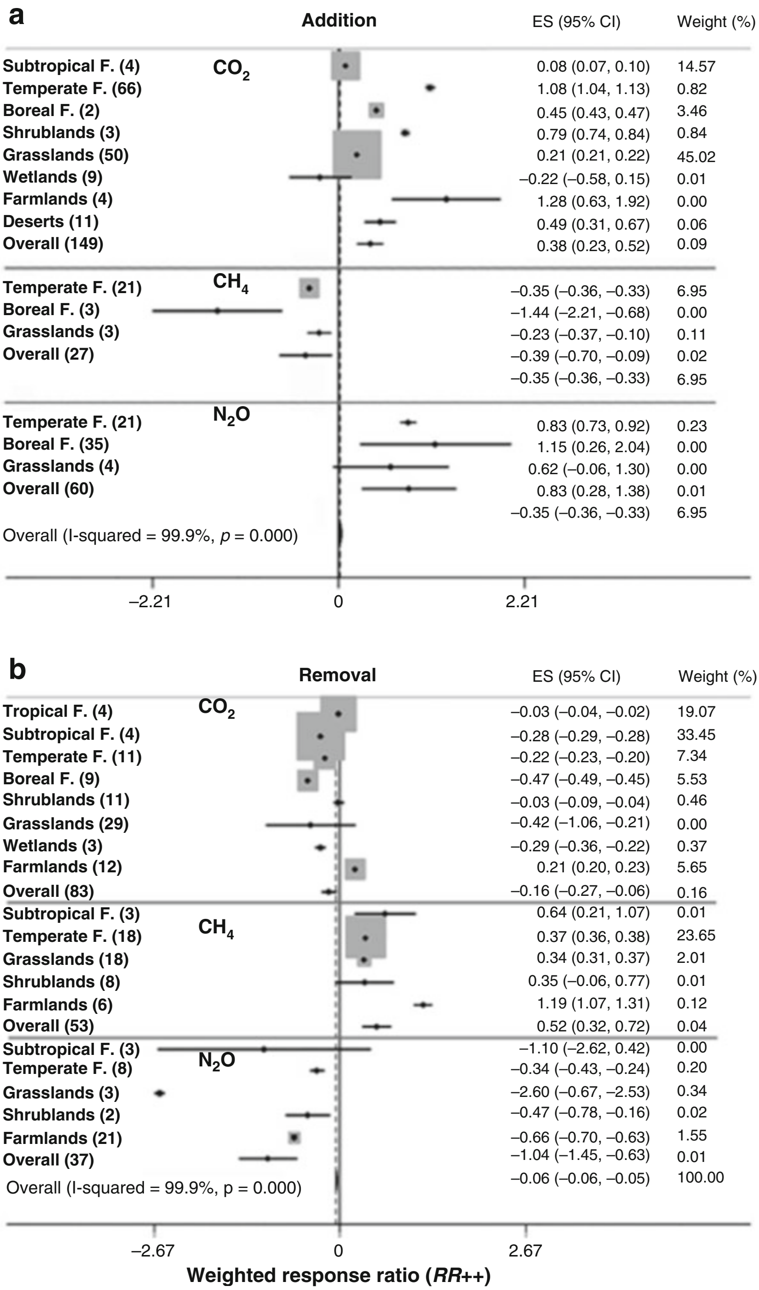

Fig. 12 Precipitation changes on soil CO_2, CH_4, and N_2O emission using meta-analysis. (**a**) Precipitation addition. (**b**) Precipitation removal. (Adapted from Yan et al. 2018)

montane and lowland forests based on 64 studies. Soil CO_2 emission is not influenced by N addition in both montane and lowland forests. Lu et al. (2011) reported a significant increase in soil N_2O emission by N addition, with 16.13% increase in non-agricultural lands (26 studies) and 2.17% increase in agricultural lands (124 studies). Aronson and Allison (2012) found that of the 99 comparisons from 33 different studies, 94 showed a larger N_2O release in the N addition plots relative to control. The effects vary by ecosystem and season of study. Shrubland has highest effect, significantly higher than grasslands and deciduous forests.

Similar to field experimental studies, several meta-analysis studies considered the effects of two and more climatic factors on soil GHG emissions. Sillen and Dieleman (2012) used data from 77 publications with elevated CO_2 and/or N application treatments and found that moderate N additions promotes C decomposition in elevated CO_2. Soil microbial biomass is reduced under N addition, but enhanced under the elevated CO_2 and elevated CO_2 plus N addition. Zhou et al. (2016a) performed a meta-analysis of 150 multiple-factor studies derived from 65 publications to examine the effects of global change factors on Rs. They found that elevated CO_2, N addition, irrigation, and warming induced significant increases in Rs by 28.6%, 8.8%, 9.7%, and 7.1%, respectively. They found that the interactions of elevated CO_2 and warming have opposite effects on Ra and Rh. Zhou et al. (2019) also conducted a meta-analysis on grazing and global change factors on soil and ecosystem respiration in grasslands, and found that grazing and drought significantly decrease Rs by 12.35% and 20.95%, respectively, while warming, N addition, and increased precipitation stimulate it by 2.12%, 5.49%, and 13.44% respectively. Song et al. (2019) conducted a meta-analysis of 1119 manipulative experiments from 2230 peer-reviewed studies on terrestrial C processes including Rs to global change, and found that warming, increased precipitation, and elevated CO_2 enhance Rs by 15%, 21%, and 30%, respectively, while decreased precipitation and N addition reduce Rs by 22% and 5%, respectively. Soil CO_2 emission is enhanced in all ecosystems by increased precipitation and elevated CO_2. Only the high N addition level significantly reduces Rs.

Manty studies have also investigated the impacts of global climate changes on enzyme activities and soil functional genes involved in soil GHG emissions (Fig. 2). Kelley et al. (2011) conducted a meta-analysis of 34 studies that examined responses in microbial enzyme activity to elevated CO_2. Among 10 enzymes examined including those degrading starch, beta-glucan, cellulose, xylan/hemicellulose, lignin, organic P, and organic N, only the activity of one enzyme N-acetyl-glucosaminidase is increased consistently at elevated CO_2 by an average of 12.6%. Chen et al. (2018) synthesized 56 studies on the effects of global warming on soil cellulase and ligninase activities, and found that warming significantly enhanced ligninase activity by 21.4% but had no effect on cellulase activity. Ren et al. (2017) synthesized the responses of C-degrading extracellular enzyme activities (EEAs) to altered precipitation from 70 published studies. They found that increased precipitation significantly enhances soil oxidative C-degrading EEAs by 6.58%, but has no effects on hydrolytic C-degrading EEAs. Decreased precipitation increases hydrolytic EEAs by 25.79%. Xiao et al. (2018) synthesized data from 132 peer-review publications of

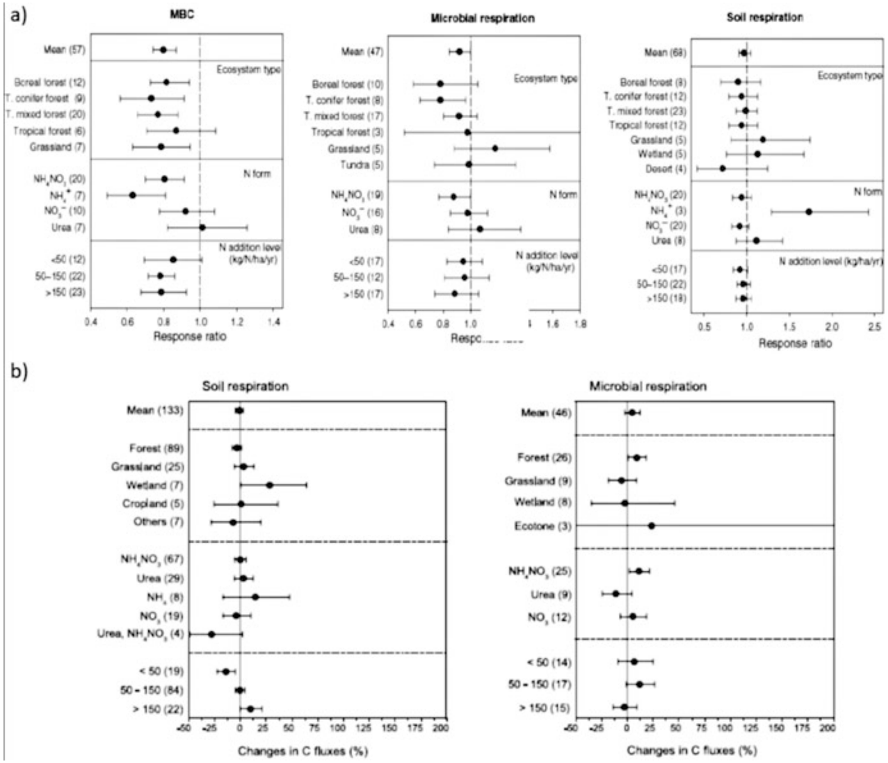

Fig. 13 Effects of nitrogen addition on soil microbial biomass carbon, soil respiration and microbial respiration. (**a**) Response ratio of microbial biomass carbon (MBC, left panel), microbial respiration (middle panel), and soil respiration (right panel). (Adapted from Liu and Greaver 2010). (**b**) Respiration ratio of soil respiration (left panel) and microbial respiration (right panel). (Adapted from Yue et al. 2016)

soil EEAs involved in C, N and phosphorus (P) acquisition in response to seven global change factors. They found that elevated CO_2 has no significant effects on soil EEAs. Nitrogen addition stimulates C-acquisition (9.1%) and P acquisition (9.9%) EEAs, but suppresses oxidase activity (−6.8%) (Table 2). Decrease in precipitation dramatically suppresses oxidase activity (−47.2%), and increase in precipitation marginally stimulates N-acquisition EEA (16.7%), while warming significantly decreases oxidase activity (−10.9%) and has minor positive effect on hydrolytic enzymes. They concluded that EEAs are generally more sensitive to nutrient addition than to atmospheric and climate change.

Li et al. (2019) collected 72 case studies from 46 papers and that reported the effects of temperature and/or precipitation on soil functional genes (i.e., archaeal *amoA*, bacterial *amoA*, *nosZ*, *narG*, *nirK*, and *nirS*). They found that increased temperature does not significantly affect abundance of archaeal *amoA*, bacterial *amoA* and *nosZ*, but significantly decreases the abundances of *nirK* and *nirS* by 26% and 31%, respectively (Fig. 14a). No significant differences of temperature effects are found between biome, treatment season, and method groups. Decreased precipitation has few effects on abundances of archaeal *amoA*, bacterial *amoA*, *nirK* and *nosZ* but shows positive effects on abundances of *nirS*. Increased precipitation has little effect on abundances of archaeal *amoA*, *nirK*, *nirS*, and *nosZ* while showing negative effects on abundances of bacterial *amoA*. Dai et al. (2020) synthesized 1270 observations from 134 publications and found elevated temperature increases the abundances of the *nirS* gene with plants and *nosZ* genes without plants. There is no effect on the abundances of the ammonia-oxidizing archaea *amoA* gene, ammonia-

Table 2 Summary of the relative change of microbial enzymes under different global change factors. Bold numbers represented significant changes when the 95% confidence interval (CI) of the effect size did not overlap with zero. Abbreviations of the enzymes and soil properties are: β-1,4-glucosidase (BG), β-Dcellobiohydrolase (CB), β-1,4-N-acetyl-glucosaminnidase (NAG), leucine amino peptidase (LAP), urease, acid phosphatase (AP), phenol oxidase (POX), and peroxidase (PER). (Adapted from Li et al. 2018)

	+CO_2 (%)	+Temperature (%)	+Precipitation (%)	−Precipitation (%)	+Nitrogen (%)
BG	4.2	0.1	−1.0	−4.6	**10.8**
CB	11.1	11.7	14.8	NA	5.4
NAG	12.9	10.9	**25.2**	29.5	2.0
LAP	NA	1.4	**24.5**	NA	5.1
Urease	1.5	2.4	NA	**−30.6**	−2.8
AP	−1.8	7.1	8.5	−5.1	**9.9**
POX	−10.7	−8.4	42.8	**−47.2**	**−9.3**
PER	NA	**−8.6**	NA	NA	−4.0
C-acquisition enzymes	8.5	4.0	0.0	−4.6	**9.1**
N-acquisition enzymes	6.7	3.3	16.7	−17.6	−0.5
Oxidase	−10.7	**−10.9**	42.8	**−47.2**	**−6.8**

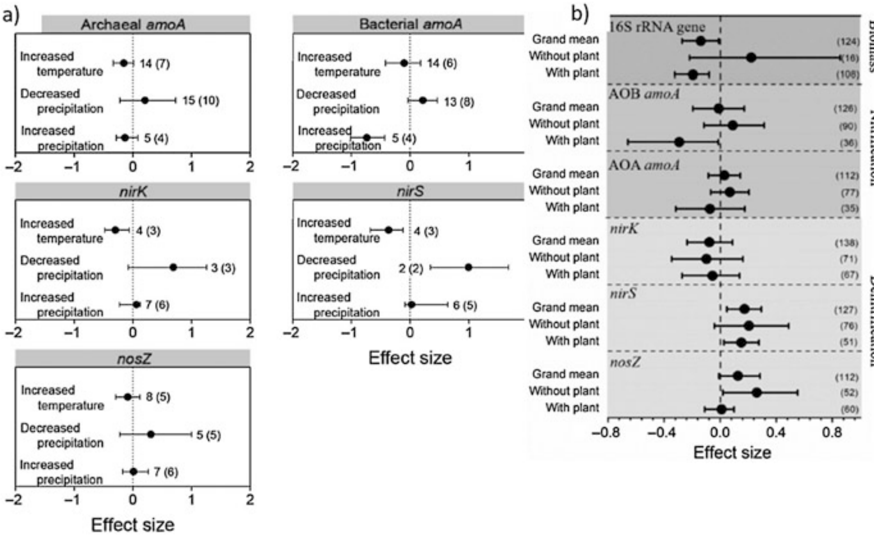

Fig. 14 Effects of global warming and precipitation change on soil functional genes. (**a**) Temperature and precipitation changes. (Adapted from Li et al. 2020). (**b**) Temperature. (Adapted from Dai et al. 2020)

oxidizing bacteria *amoA* and *nirK* genes (Fig. 14b). These findings infer that elevated temperatures have a profound impact on global N cycling processes with implications of a positive feedback to global climate and emphasize the close linkage between soil microbial C and N cycling.

Biogeochemical and Ecosystem Modeling

Process-based modeling is an important tool in estimation and prediction of soil GHG emissions in terrestrial ecosystems in response to global climate changes (Tian et al. 2016). Many ecosystem models have been developed and used to estimate ecosystem C and nutrient cycling over the past two decades. Multiple model intercomparison projects (MIPs) have been established to evaluate model uncertainties in simulating the terrestrial C and nutrient processes and dynamics. For example, the Vegetation-Ecosystem Modeling and Analysis Project (VEMAP) was a pioneer MIP activity providing multi-model ensemble estimates of C fluxes and storage in response to changing climate and atmospheric CO_2 (Melillo et al. 1995; Schimel et al. 2000). Global methane (CH_4) MIPs and synthesis activities such as Global Carbon Project (GCP) global CH_4 budget synthesis (Saunois et al. 2016; Poulter et al. 2017) were implemented a few years ago. Recently, an MIP for the N models to assess the global N_2O budget was established to investigate the uncertainty sources in N_2O estimates and provide multi-model N_2O emissions estimates from natural and agricultural soils (Tian et al. 2015, 2019) (Fig. 15). Ten ecosystem models including CLM-CN, DLEM, LM3V-N, O-CN, LPJ-GUESS, LPX-Bern, ORCHIDEE, ORCHIDEE-CNP, TRIPLEX-GHG, and VISIT are participating in model simulations (Zaehle and Friend 2011; Xu et al. 2012a; Inatomi et al. 2010; Stocker et al. 2013; Saikawa et al. 2014; Zhu et al. 2014; Huang and Gerber 2015; Goll et al. 2017). Most of the models implemented in these MIP include soil CO_2, CH_4, and N_2O emission modules that can be used to estimate and forecast soil CO_2, CH_4, and N_2O emissions at both small and large spatial scales.

Site and Stand Levels

As elevated CO_2 increases plant growth, biomass and litterfall, and enhances C inputs to soil, it is expected that elevated CO_2 will stimulate more soil CO_2 and N_2O and perhaps CH_4 emissions. Warming and precipitation are two important factors regulating plant photosynthesis, respiration and ecosystem C sequestration, and play an important role in soil GHG emissions. Nitrogen availability is often limiting plant growths and microbial activities, and will influence soil GHG emissions, particularly soil N_2O emission. Biogeochemical models have been built to investigate the impacts of global climate change based on field experiments. For example, Luo et al. (2001) developed a partially process-based component model and tested the impacts of elevated CO_2 and other factors in a forest ecosystem. They found that elevated CO_2 increases soil CO_2 emission, mostly by enhanced soil Rh. Hui and Luo (2004) simulated the effects of elevated CO_2 on soil CO_2 emission in Duke Forest using a process-based soil CO_2 efflux model (PATCIS), and found that elevated CO_2 increases annual soil CO_2 efflux by 26% in 1997 and 18% in 1998, mainly due to the enhanced live fine root biomass and litterfall. Rafique et al. (2014) examined the effect of climate change on GHG emissions in no-till croplands using a process-based model, DAYCENT, and found that, in the altered climate scenario (i.e., 2 °C warming and 40% precipitation regime shift from dry season to wet season), total N_2O and CO_2 fluxes are decreased by 9% and 38% respectively, whereas CH_4 fluxes are increased by 10%. They concluded that the main difference in all GHG emissions is observed in summer period due to drought conditions created by reduced precipitation and increased temperatures. Using TECO, Zhou et al. (2008) simulated Rh and found that Rh responds to temperature from −2 °C to +10 °C following a parabolic-curve response. Rh increases with temperature, reaches a peak at 6 °C and declines. He et al. (2018) estimated soil N_2O emissions in Southwestern Ontario, Canada using a regionalized DNDC model, and found that the mean annual N_2O emissions for winter wheat are significantly increased by about 38.1% for conventional tillage and 17.3% for no-tillage. Increased CO_2 has a positive effect on reducing N_2O emissions (only significant for winter wheat under conventional tillage) compared to the baseline CO_2 under future climate change scenarios. High atmospheric CO_2 concentrations can improve water and N use efficiency leaving less excess N available in the soil for N_2O emissions and these effects were captured by the modified DNDC model.

Using the DNDC model, Li et al. (1992) found increasing the annual precipitation slightly decreases N_2O due to a greater proportion of the denitrification reactions continuing to N_2. Saggar et al. (2007) also found a decrease in net N_2O emissions

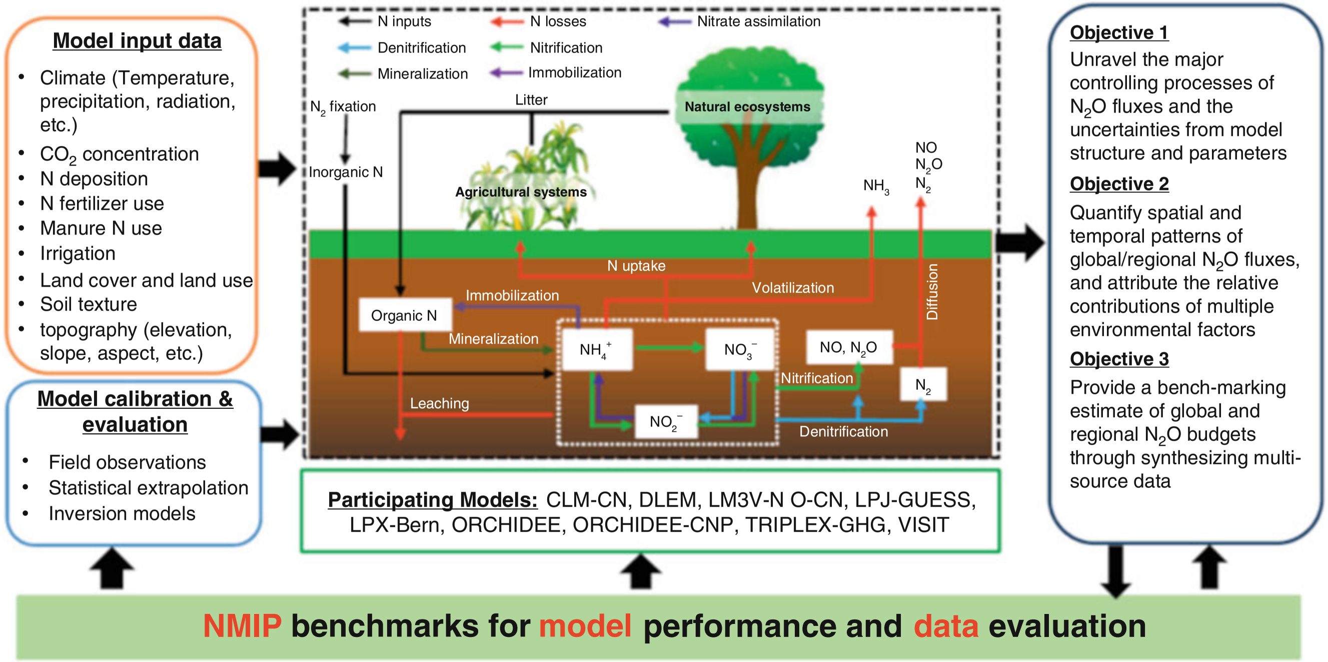

Fig. 15 Framework of Nitrogen Multiple model Intercomparison Projects (NMIP). (Adapted from Tian et al. 2018). Three objectives are achieved by simulated soil N_2O (and other gases) using ten models with common input data

with increasing rainfall due to increased NO_3^- leaching. Deng et al. (2016) also found that soil moisture plays an important role in soil N_2O emission. Lu et al. (2008) used Forest-DNDC to simulate the effects of climate factors, temperature and precipitation changes on GHG emissions in *Abies fabric* forest and found soil CO_2 emissions increases with the increase of temperature, and CO_2 emissions change little with increased baseline precipitation. But total annual soil N_2O emissions increases with increases in precipitation. It seems that precipitation is not a principal factor affecting soil CO_2 emissions, but has strong effects on soil N_2O emissions.

Regional and Global Scales

At regional and global scales, many ecosystem models have been applied to simulate soil and ecosystem GHG emissions. For example, Tian et al. (2015) estimated the combined global warming potential (GWP) of CO_2, CH_4 and N_2O fluxes in North American terrestrial ecosystems using the DLEM, and quantified the relative contributions of global change and other environmental factors to the GWP during 1979–2010. They found that the best estimate of net GWP for CO_2, CH_4 and N_2O fluxes is -0.50 ± 0.27 Pg CO_2 eq year^{-1}. About two thirds of the land CO_2 sink is offset by CH_4 and N_2O emissions from terrestrial ecosystems in the North American continent. Climate change and elevated tropospheric ozone concentration have contributed the most to GWP increase, while elevated atmospheric CO_2 concentration have contributed the most to GWP reduction (Fig. 16). The DLEM model was also used in an integrated analysis with bottom-up (inventory, statistical extrapolation of local flux measurements, and process-based modelling) and top-down (atmospheric inversions) approaches to quantify the global net biogenic GHG balance between 1981 and 2010 (Tian et al. 2016). They found that the cumulative warming capacity of concurrent biogenic CH_4 and N_2O emissions is larger than the cooling effect resulting from the global land CO_2 uptake from 2001 to 2010, resulting in a net positive cumulative impact of the three GHG on the planetary energy budget (Fig. 17). Ito et al. (2018) used a process-based terrestrial ecosystem model (VISIT) and simulated N_2O emission in East Asia from 1901 to 2016. They reported that the mean regional N_2O emission rate in the 2000s is 2.03 Tg N_2O year^{-1}, more than triple the rate in 1901. The increase of N_2O emissions is mainly due to the increase of agricultural inputs from fertilizer, and positively linked to precipitation and slightly by temperature. In a comprehensive synthesis of global N_2O emission with seven process-based terrestrial biosphere models, Tian et al. (2019) assessed the effects of multiple anthropogenic and natural factors on N_2O emissions. They found that global soil N_2O emissions are increased from 6.3 ± 1.1 Tg N_2O-N year^{-1} in the preindustrial period to 10.0 ± 2.0 Tg N_2O-N year^{-1} in the recent decade (2007–2016). The hotspots of N_2O emission and increases occur in the middle and southeastern USA, southeast Asia, and tropical and subtropical regions (Fig. 18). Eighty-two percent of the total increase is from cropland soil emissions (3.3 Tg N_2O-N year^{-1}). N fertilizer application, N deposition, manure N application, and climate change contributes 54%, 26%, 15%, and 24%, respectively, to the total increase. Rising atmospheric CO_2 concentration reduces soil N_2O emissions by 10% through the enhanced plant N uptake.

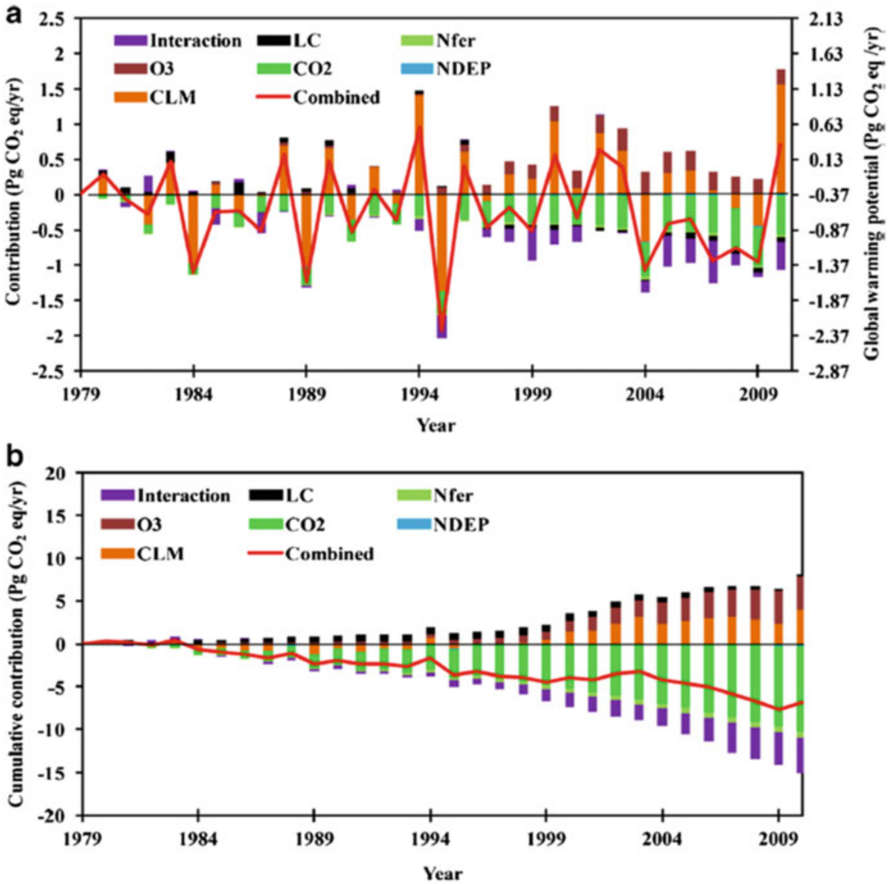

Fig. 16 Annual (**a**) and cumulative (**b**) contributions of different environmental factors to changes in global warming potential (Pg CO_2 eq year^{-1}) in the terrestrial ecosystems of North America. (Adapted from Tian et al. 2015)

Closing Remarks and Future Research

Considerable progresses have been made during the past several decades to better understand the effects of global climate change on soil GHG emission using both laboratory incubation, field experiment, meta-analysis, and ecosystem modeling approaches. Among these approaches, laboratory incubation allows us to test specifically how changes in climatic factor and nutrients could influence soil CO_2, CH_4 and N_2O emission. It is ideal to link these changes to some underlying mechanisms related to soil microbial activities and composition changes. Field experimental study is still a power tool to evaluate the responses of climate change on soil GHG emission in a more realistic way and quantify the amounts of total emissions from soils. It has been extensively used and will continue to be a major tool in global

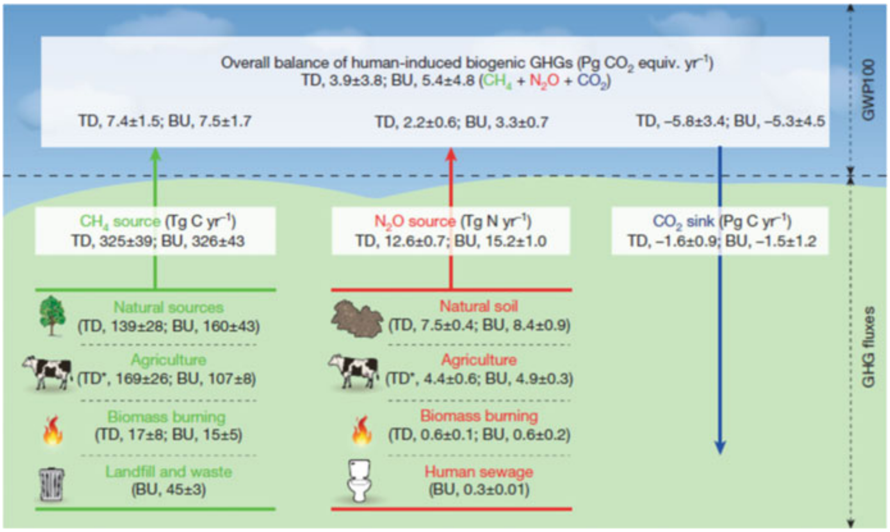

Fig. 17 Global greenhouse gases (GHG, CO_2, CH_4 and N_2O) budgets of the terrestrial biosphere in the 2000s. (Adapted from Tian et al. 2016). TD, top-down approach; BU, bottom-up approach

change studies. As more and more data accumulating, meta-analytic techniques provide another mean to quantitatively integrate the individual studies and generate a grand conclusion on a common topic. With the advancement of sensor technology, automatic recording, and satellite images, more and more observations and measurements such as using eddy covariance technique in terrestrial ecosystems under natural conditions can be made over long terms and at large spatial scales. Biogeochemical and ecosystem modeling becomes more and more important, especially at scaling up from plot level experiments to large spatial scales and forecasting future responses.

While laboratory incubation and manipulative field studies provide useful insight into how climate change influences soil GHG emission, logistical constraints often prevent the examination of some climatic factors, and more importantly, the complex interactions between multiple and changing climatic factors (Norby and Luo 2004; Zhou et al. 2016a). Due to the facility limitation, the early studies on the elevated CO_2 effects mostly used growth chamber or open-top chamber. Only after the FACE facility is constructed, the effect of CO_2 can be truly investigated without the confounding changes in temperature and air relative humidity. Due to the cost of construction and fire concerns, warming studies in forests are often limited to soil warming. The whole ecosystem warming studies have seldom been done (Hanson et al. 2016). Temperature and precipitation forcing are also codependent, requiring a complex systems approach to understand the impacts of global climate change on soil GHG emission. Additionally, experimental studies typically conducted at plot or stand scales and over relatively short time scales leave gaps in our understanding about the long-term effects of climate change at landscape and regional scales. The long-term and multi-factor experiments should be conducted (Norby and Luo 2004).

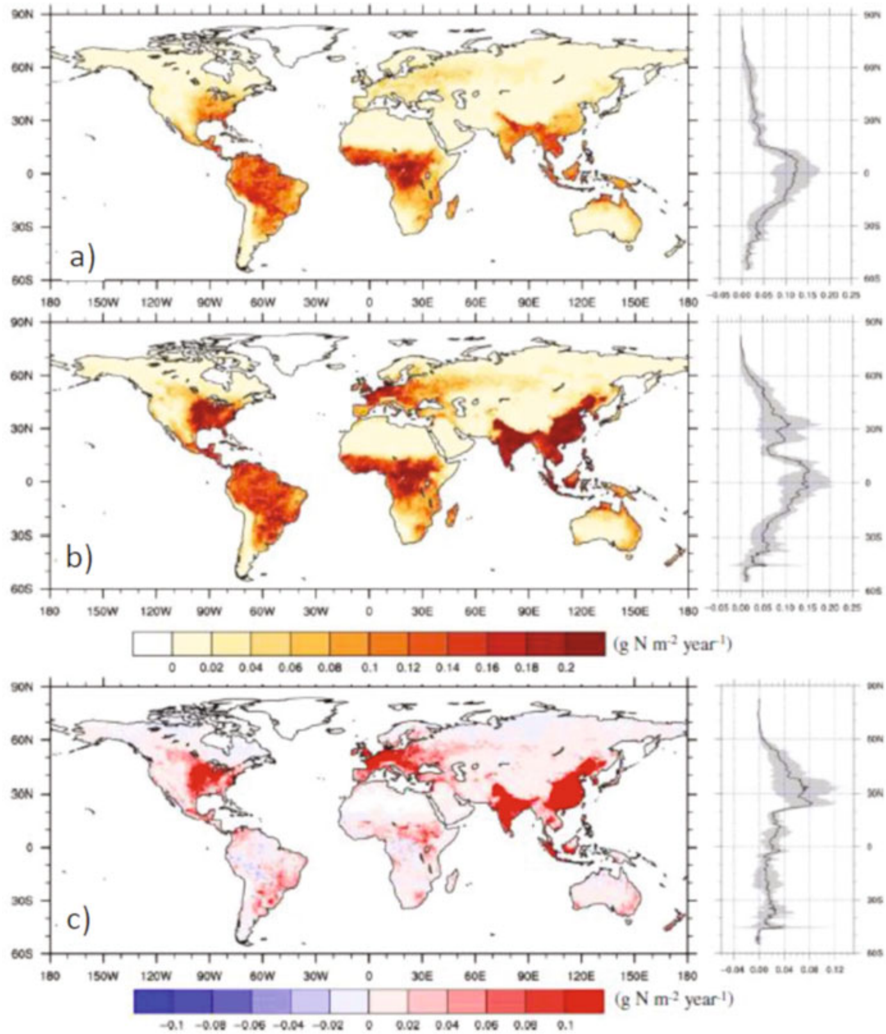

Fig. 18 Model ensemble mean of soil N_2O emission density across global land surface in the preindustrial period (**a**) and the recent decade (**b**, 2007–2016), and the difference between the two periods (**c**). (Adapted from Tian et al. 2018)

Meta-analysis is a power tool to generate an overall picture of climate change impacts on soil CO_2, CH_4 and N_2O emissions. As demonstrated in this chapter, multiple meta-analyses provided different results, similar to field experimental studies. The discrepancy was mostly caused by the data collection process and criteria used for data collection, as different data/studies are included in different meta-analyses, based on the major purposes of the studies. When the sample size is small, the focus of the study should be to increase and invest more on field studies in different ecosystems and environmental conditions. One issue often revealed by the meta-analysis is the imbalance of sample size. More global climate change studies have been conducted in temperate grasslands and forests than in other ecosystems. More field experiments in under-represented ecosystems need to be conducted. But when the sample size is large (e.g., >70 based on an unpublish simulation), adding one or two studies would not really influence the results and conclusion. At this stage, a through and comprehensive meta-analysis would benefit more than adding a single study. Some other issues revealed by meta-analysis, such as the duration of study and magnitude of treatment factor, should be addressed in future field studies.

For ecosystem biogeochemical modeling study, data-model need to be better integrated. In recent decades, inverse modeling and data assimilation techniques have been applied to improve model structure and parameters, reduce uncertainty, and increase the accuracy of model forecasting (Luo et al. 2011; Xu et al. 2006). Further improvement in ecosystem models with soil GHG emissions should focus more, as revealed by Xu et al. (2012b) on (1) the mechanisms underlying soil GHG emissions, with an emphasis on improving and validating individual GHG emission processes over depth and horizontal space, (2) capability of model to simulate soil GHG emissions across highly heterogeneous spatial and temporal scales, particularly hotspots for CH_4 and N_2O, and (3) development model benchmarking frameworks that can easily be used for model improvement, evaluation, and integration with data from molecular to global scales.

As more data are accumulating in many manipulative experiments, inverse modeling and data-assimilation will play a more important role in global change ecology. Besides climate variability, climate disturbances such as drought, cold-spell, and heat-wave need to be built into biogeochemical models. Thus, great effort will be required to integrate previous and new data from experimental results, and process knowledge into ecosystem models. Further development of data-model assimilation tools, analytical methods, and ecosystem models to improve understanding of climate change effects on soil GHG emission will be needed in order to actually forecast the climate-C cycle feedbacks, inform policy makers, and provide guideline for soil conservation and ecosystem management.

Cross-References

- ► Greenhouse Gas Emission Reduction Using Advanced Heat Integration Techniques
- ► How to Think about Climate Change Responses
- ► Introduction to Climate Change
- ► Life Cycle Assessment of Greenhouse Gas Emissions
- ► Natural Resource Management And Sustainable Agriculture
- ► Vulnerability Assessment of the Indian Himalayan Forests in Terms of biomass Production and Carbon Sequestration Potential in Changing Climatic Conditions

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