Sources and Sinks of Major Greenhouse Gases Associated with New York State's Natural and Working Lands: Forests, Farms, and Wetlands

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Final Report

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Abstract

This report presents preliminary estimates of the exchange of primary greenhouse gasses (GHG) between the atmosphere and the landscape surface in New York State. The focus is on "natural and working lands," including forests, agricultural lands, and wetlands. Carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are considered. Results are intended to be used for improving understanding of the extent to which these landcover types may contribute to or sequester atmospheric GHGs on a net basis. Forest ecosystem carbon (C) stocks associated with live aboveground and belowground biomass, dead wood, litter, and soil organic C were determined for individual years between 1990 and 2017 using the U.S. Forest Carbon Accounting Framework, which is fundamentally based on annual estimates of forest C stocks determined from forest biomass data collected by the Forest Inventory and Analysis (FIA) program administered by the U.S. Forest Service. The net CO_2-C flux for a given year was determined based on the C-stock difference between two adjacent years. Estimates of GHG emissions and GHG mitigation potential for the agricultural sector were derived from published sources, supplemented with calculations. Net GHG fluxes associated with palustrine and intertidal wetlands were determined from values reported in the literature from site-level studies located within the vicinity of the State, in conjunction with estimated wetland area as represented in the National Wetland Inventory (NWI). All GHG flux estimates were reported as carbon dioxide equivalents (CO_2e).

The annual estimate of CO_2e flux associated with forest ecosystems in 2017 was -25.5 million metric tons (MMt) CO_2e, indicating a net sink. The strength of the forest CO_2e sink has been steadily decreasing and has been reduced by 2.8 MMt CO_2e since 1990. Linear extrapolation of this historical trend yielded an estimated net CO_2e flux of -22.5 MMt CO_2e in the year 2050. However, future trends in the forest CO_2e sink are expected to be dependent on future conditions of climate, atmospheric deposition, disturbance, and the extent to which forests are managed to maximize CO_2 sequestration. Agriculture was calculated to be an annual source of 8.38 MMt CO_2e under ambient levels of sector activity. The largest agricultural GHG emission rates originated from enteric fermentation, manure management, and soil fertilizer production and use. Implementation of agricultural best management practices aimed at reducing GHG emissions, reductions in food waste, and reforestation of former agricultural land were determined to have the potential to mitigate more than the full extent of the agricultural GHG contribution (9.23 MMt CO_2e per year). The net annual CO_2e flux associated with wetlands in this study was estimated to be 4.79 MMt CO_2e. In general, the various wetland types acted as sinks for CO_2 and sources of CH_4. Although there was a wide range of uncertainty, forested and non-forested palustrine wetlands were judged most likely to be net sources of CO_2e and intertidal wetlands were likely to be approximately

CO_2e neutral. The land cover types included in this study were considered to have an overall net annual CO_2e flux of -12.33 MMt CO_2e. A value of -18.56 MMt CO_2e was estimated for the year 2050. Improved forest management and prevention of forest loss are expected to be important strategies for further increasing the GHG mitigation potential of natural and working lands of New York State.

Keywords

Greenhouse gas, carbon dioxide, nitrous oxide, sequestration, carbon, forest, agriculture, wetland, New York

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Acronyms and Abbreviations

BMP	best management practices
C	carbon
CAFO	Confined Animal Feeding Operations
CH_4	methane
CO_2	carbon dioxide
CO_2-C	carbon dioxide expressed as carbon content
CO_2e	carbon dioxide equivalents
CRM	component ratio method
DBH	diameter at breast height
DEC	Department of Environmental Conservation
DOM	dissolved organic matter
EPA	United States Environmental Protection Agency
FIA	Forest Inventory and Analysis program
FRF	Forest Land Remaining Forest Land
GHG	greenhouse gas
g	gram
GWP	global warming potential
IPCC	Intergovernmental Panel on Climate Change
K	potassium
Kg	kilogram
Kt	kiloton
LCF	Land Converted to Forest Land
m^2	square meter
MMt	million metric tons
MTBS	Monitoring Trends in Burn Severity
N	nitrogen
N_2O	nitrous oxide
NEE	net ecosystem exchange
NFI	National Forest Inventory
NO_3	nitrate
NPP	net primary production
NLCD	National Land Cover Data
NYSERDA	New York State Energy Research and Development Authority
NWI	National Wetlands Inventory

P	phosphorus
SOC	soil organic carbon
SOCCR2	Second State of the Carbon Cycle Report
Tg	teragram
TOC	total organic carbon

Executive Summary

The main greenhouse gases (GHG) emitted into the atmosphere in New York State are carbon dioxide (CO_2; from fossil fuel combustion), methane (CH_4; from domestic livestock enteric fermentation, natural gas, and landfills), and nitrous oxide (N_2O; from agricultural soils, fuel combustion, and manure management; EPA 2018). This report presents preliminary estimates of GHG exchange of these primary gases between the atmosphere and the landscape surface in the State. The focus is on "natural and working lands," namely forests, agricultural lands, and wetlands. Results are intended to be used for improving understanding of the extent to which these landcover types may contribute to or sequester atmospheric GHGs on a net basis.

Forest ecosystem carbon (C) stocks associated with live aboveground and belowground biomass, dead wood, litter, and soil organic C were determined for individual years between 1990 and 2017 using the U.S. Forest Carbon Accounting Framework (Woodall et al. 2015). This framework is fundamentally based on annual estimates of forest C stocks determined from forest biomass data collected by the Forest Inventory and Analysis (FIA) program administered by the U.S. Forest Service. The net CO_2-C flux for a given year was determined based on the C-stock difference between two adjacent years (IPCC 2006). Estimates of GHG emissions and GHG mitigation potential for the agricultural sector in New York State were derived from published sources, supplemented with calculations based on data provided in published sources. The focus was primarily on the largest emission sources and the largest agricultural mitigation opportunities. Estimates of smaller sources and mitigation opportunities are also included where available. Net GHG fluxes associated with palustrine and intertidal wetlands were determined from values reported in the literature from site-level studies located within the vicinity of the State, in conjunction with estimated wetland area as represented in the National Wetland Inventory (NWI) database. All GHG flux estimates were reported as carbon dioxide equivalents (CO_2e).

The annual estimate of CO_2e flux associated with forest ecosystems in 2017 was -25.5 million metric tons (MMt) CO_2e (note that the minus sign indicates a net sink). The strength of the forest CO_2e sink has been steadily decreasing, having been reduced by 2.8 MMt CO_2e since 1990. Linear extrapolation of this historical trend yielded an estimated net CO_2e flux of -22.5 MMt CO_2e in the year 2050. However, future trends in the forest CO_2e sink are expected to be dependent on future conditions related to climate, atmospheric deposition, disturbance (e.g., insect infestation), and the extent to which forests can be managed to maximize CO_2 sequestration.

Agriculture was estimated to be an annual source of 8.38 MMt CO_2e under ambient levels of sector activity. The largest agricultural GHG emission rates originated from enteric fermentation, manure management, and soil fertilizer production/use. Implementation of agricultural best management practices aimed at reducing GHG emissions, reductions in food waste, and reforestation of former agricultural land were determined to have the potential to mitigate more than the full extent of the agricultural GHG contribution (9.23 MMt CO_2e per year).

The net annual CO_2e flux associated with the wetland types considered in this study was estimated to be 4.79 MMt CO_2e. In general, the various wetland types acted as sinks for CO_2 and sources of CH_4. Although there was a wide range of uncertainty, forested and non-forested palustrine wetlands were judged most likely to be sources of CO_2e and intertidal wetlands were likely to be approximately CO_2e neutral.

In aggregate, the land cover types included in this study were considered to have a net annual CO_2e flux of -12.33 MMt CO_2e. Although it is challenging to estimate future conditions, a value of -18.56 MMt CO_2e can be considered as a reasonable estimate of future (approximately year 2050) net GHG flux conditions based on linear extrapolation of the forest flux, full implementation of agricultural GHG mitigation potential, and continued constant GHG fluxes associated with wetlands. Improved forest management and prevention of forest loss are expected to be important strategies for further increasing the GHG mitigation potential of natural and working lands of New York State.

1 Background

1.1 Introduction

The main greenhouse gases (GHGs) emitted into the atmosphere in the United States are carbon dioxide (CO_2; ~81.6% from fossil fuel combustion), methane (CH_4; from domestic livestock enteric fermentation, natural gas, and landfills), and nitrous oxide (N_2O; from agricultural soils, fuel combustion, and manure management; EPA 2018). These emissions are augmented and/or offset, in part, by natural emissions sources and carbon (C) sequestration in terrestrial and aquatic environments. The U.S. Environmental Protection Agency (EPA 2019) estimated that, in total, the combination of forest sequestration, urban trees, agricultural soils, landfilled vegetative residues, and coastal wetlands nationwide offset 11.3% of total GHG emissions in 2017.

Here, a preliminary estimate to quantify GHG exchange between the atmosphere and the landscape surface in New York State is made. The focus is on “natural and working lands,” namely forests, agriculture, and wetlands. This C exchange can potentially both add to and offset a portion of the human-caused GHG emissions in the State. By adopting EPA protocols for identified sources and sinks, this report strives to be consistent with Intergovernmental Panel on Climate Change (IPCC) international conventions for GHG inventories. This analysis will help improve understanding of New York State’s role in exacerbating or moderating climate change (cf., Le Quéré et al. 2014). The report presented here focuses on the three primary GHGs, presented using a common metric, the CO_2 equivalent (CO_2e), such that the 100-year warming potential of each GHG can be presented in equivalent terms and combined in C flux calculations.

The New York State Energy Research and Development Authority’s (NYSERDA) New York State Greenhouse Gas Inventory for the period 1990 to 2015 (NYSERDA 2018) provided an accounting of greenhouse gas emissions associated with different sectors and source types, including energy, industry, agriculture, and waste. The total statewide GHG emissions were estimated to be about 218 million metric tons of CO_2e (MMt CO_2e), of which 84% was attributed to the energy sector and 4% to agriculture. The State’s per capita GHG emissions were about half the U.S. average (NYSERDA 2018). The main contributor to the total GHG emissions was identified as the transportation sector. The total GHG emissions in New York State increased from about 1990 to 2005, followed by a decrease, and emissions levels in 2015 were about 8% lower than 1990 emissions (NYSERDA 2018).

Terrestrial, freshwater, and estuarine ecosystems have some capacity to offset GHG emissions from fossil fuel combustion. The role of land management in C sequestration has become more important as society searches for ways to limit the warming potential of human-caused GHGs (Anderegg et al. 2013a, Anderegg et al. 2013b). Development of policy to partly mitigate climate change should account for spatial and temporal heterogeneity, which requires spatially referenced estimates of C sources and sinks and improved understanding of controls on C fluxes and their interactions (King et al. 2015). The analysis reported here represents a step in that direction for New York State.

Emissions of CO_2, CH_4, and N_2O collectively contribute the majority of global radiative heat forcing attributable to GHGs. Although some of these emissions are natural, they are largely human caused, resulting from fossil fuel combustion, land use (especially agriculture), and land use change. It is critical to improve scientific understanding of GHG flux in order to develop sound policy to mitigate fossil fuel combustion and other sources of GHGs by improving land-based C sequestration (King et al. 2015) and reducing GHG emissions. The cycles of these GHGs and the climate system are closely intertwined (Ciais et al. 2013).

Griscom et al. (2017) identified a list of 20 conservation, restoration, and land management actions that can be used to increase C sequestration and/or reduce GHG emissions from forests, wetlands, grasslands, and agricultural lands. Most of these measures also yielded co-benefits pertaining to water filtration, flood control, biodiversity, and/or improved resilience to climate change (Griscom et al. 2017). Improved land stewardship was identified as particularly helpful in this regard. Griscom et al. (2017) estimated that these conservation actions might provide more than one-third of the identified cost-effective mitigation that would be needed to limit global temperature rise to below 2°C. For the United States, a subsequent analysis using this approach found that emissions could be reduced by 21% (Fargione et al. 2018).

1.2 Greenhouse Gas Sources and Sinks

Sources and sinks of GHGs are assigned for this analysis either positive or negative signs, whereby sources are positive (+) and sinks are negative (-). The term “lateral transfer” is used to represent redistribution of C from one pool to another, such as, for example, from soil to surface water or coastal wetland to estuary. Lateral transfers involve only limited gas exchange with the atmosphere and therefore are not included in GHG flux calculations. Metrics, units, and terms used in this report are highlighted in Table 1.

Table 1. Metrics and Units

Metrics	
flux	mass per year (measurement of movement between atmosphere and landscape)
stock	mass (quantity in a given pool at a given time)
source	more C moves from the reservoir to the atmosphere than vice versa
sink	more C moves from the atmosphere to the reservoir than vice versa
lateral transfer	movement of C between reservoirs without any appreciable atmospheric exchange
Units and Terms	
MMt/yr	million metric tons per year—equals 1 teragram (Tg; see below) per year
CO_2e – CO_2 equivalent	amount of CO_2 that would produce the same effect on radiative balance as another GHG; each kg of CO_2 eq is equivalent to 0.273 kg C; calculated in this study over a period of 100 years
Tg C	mass equal to 10^{12} grams; also equal to 1 MMt Conversion of C to CO_2—multiply the mass of C by 3.67
GHG	greenhouse gas; for the purposes of this report: CO_2, CH_4, N_2O

To improve understanding of the role of C in climate change, researchers measure or estimate C stocks and fluxes as well as other forms of GHG generation and destruction. Stocks represent the amount of C, or CO_2e, stored in a pool such as forest trees or agricultural soils. Flux calculations represent movements between pools that include the atmosphere, such as the flux of CO_2 from the atmosphere into forest trees as driven by the process of photosynthesis. This report attempts to estimate the magnitude of most of these C stocks and GHG fluxes in New York State, based on recent measurements, model simulations, and calculations.

Carbon cycling entails fluxes among the atmosphere, soil, plant litter, and vegetation biomass (aboveground, belowground, and dead and decaying wood). These cycles are influenced by photosynthesis, respiration, disturbance, and land management. Part of the C that is taken up during photosynthesis, and that is not respired, is allocated to plant growth and stored as wood. As vegetation biomass dies and decomposes, C is transferred to litter and soil in various stages of decay. The C in dead plant material can be stored in the soil for decades to centuries and is gradually released back to the atmosphere or to drainage water by mineralization and microbial decomposition. Carbon in the wood that is harvested through forestry is transferred to wood products such as lumber and slash, which may be buried or stored for short to long periods of time. Combustion or decomposition, on site or elsewhere, releases C back to the atmosphere.

King et al. (2015) provided a synthesis of net CO_2 exchange between the atmosphere and the land surface of North America during the period 1990–2009. It was based on multiple methods of estimating fluxes: atmospheric inversion modeling, inventories, and terrestrial biosphere modeling. Land surfaces can represent both C sources and C sinks. All methods suggested that the North American land surface represented an overall net sink for C (King et al. 2015).

Methane is an end product of bacterial decomposition of organic matter in an anaerobic environment. Flooded soils of wetlands are major sources of atmospheric CH_4. Emissions tend to be highest in spring and summer in association with plant growth and decomposition (Wilson et al. 1989). Major sources of CH_4 emissions in the State are landfills, natural gas leakage, enteric and manure management emissions from livestock, and municipal waste water treatment (NYSERDA 2018). Upland forests are not expected to store or release large quantities of CH_4 (Covey and Megonigal 2019). Soils can capture, oxidize, and store atmospheric CH_4, while wetlands (forested or otherwise) release CH_4 to the atmosphere. Living and dead trees can emit some CH_4 that is produced in soil. Microorganisms in trees located in wetland environments can also produce significant amounts of CH_4 which may be emitted to the atmosphere. Scaling up from stem or leaf measurements to forests is limited by variability and uncertainty (Covey and Megonigal 2019). Additional CH_4 is also released through forest fire (EPA 2018).

The C cycle in soils is tightly coupled to the nitrogen (N) cycle. Because plant, animal, and microbial organisms require relatively large amounts of C and N, changes in the availability of one influences the other (Gruber and Galloway 2008). Nitrogen is contributed to the soil in the form of atmospheric deposition of both oxidized (NO_x) and reduced (NH_x) N originating from emissions from such sources as motor vehicles, agriculture, power plants, and industry (Sullivan 2015). On agricultural lands and some forest lands, synthetic and manure fertilizer application is also an important source of N to the soil.

Past atmospheric N emissions and deposition during the industrial era increased forest primary production (Elser et al. 2007). This increase in production facilitated sequestration of C in the forest floor and upper soil horizons, as well as in the forest vegetation. The increase in the soil C stock that has occurred in response to atmospheric N deposition, which has been relatively high in recent decades in New York State[1] may have been appreciable (Janssens et al. 2010).

Soils emit CO_2 and CH_4 into the atmosphere through processes of litter decomposition by microorganisms (microbial respiration). Roots and mycorrhizal fungi also release CO_2 into the atmosphere (Tang et al. 2005). As the climate continues to warm, soil microbial respiration and plant respiration may increase (Hashimoto et al. 2015), but the emissions of CO_2 will also vary with soil moisture availability. Emissions of CO_2 and CH_4 will most likely be highest under moderate levels of moisture saturation (Lajtha et al. 2018).

Methane has a much higher global warming potential (GWP) in the atmosphere than does CO_2 (Saunois et al. 2016). On a 100-year time scale, methane is about 25 times more effective than CO_2 at trapping heat (IPCC 2007). The emissions of CH_4 are both natural and human caused. The human sources are mostly associated with agriculture, landfills, and waste management (EPA 2017, Lajtha et al. 2018). Natural sources of CH_4 in New York State are mainly associated with freshwater wetlands. Part (perhaps about half) of the net CH_4 emissions from wetlands may be balanced by CH_4 uptake by wetland methanotrophic microbes (Tate 2015, Lajtha et al. 2018).

Nitrous oxide is produced by biological processes in soil and water as well as by human activities associated with energy production and use, agriculture, and waste management. Emissions of N_2O are derived from N cycling reactions such as nitrification and denitrification (Galloway et al. 2003). The human-caused components include management of agricultural soils, fuel combustion (stationary and mobile), and manure management (EPA 2018).

For some representations of CH_4 and N_2O presented here, CO_2e units are used. As previously described, one CO_2e is equal to the amount of CO_2 that would have the same effect on the climate as a given amount of CH_4 or N_2O. This amount is time sensitive because different GHGs have different atmospheric residence times. In this report, CO_2e is calculated over a period of 100 years using IPCC, AR4 estimates (where N_2O=298, CH_4=25).

1.3 Forests

Forests contain dense reservoirs of C in both living and dead biomass (Prentice et al. 2001, Batjes 2014). Forest ecosystems constitute by far the largest land-based C sink in New York State, due both to their geographic extent (approximately 64% of New York State land area) and capacity for high-density and relatively stable long-term storage of C in woody biomass and soils. The ability of forests to take up and store C is fundamental to their structural and functional characteristics as ecosystems, shaped by a host of biophysical and anthropogenic factors and their interactions (Birdsey et al. 2006, Zhang et al. 2012, Domke et al. 2018). Forests are highly complex systems, but to understand their role in climate mitigation via C cycling there are two fundamental things to consider: the relative rates of tree growth and mortality, and their geographic extent.

Tree growth and mortality largely determine the source-sink dynamics of a forest. Growth represents a net C flux into the system (a sink), but new growth also increases the amount of energy that plants must allocate to maintenance of living tissue that, via respiration, generates and releases CO_2 back to the air.

Trees are able to achieve net C sequestration as they grow because much of their wood is composed of dead tissue (heartwood) that no longer requires maintenance energy. Mortality represents (or results in) a transient net C flux out of the system (a source), as microbes, fungi, and soil fauna decompose dead plant material and convert it to energy via respiration, which releases CO_2. This is a transient C source because, in nearly all cases, the death of a tree opens growing space for other plants that will grow more rapidly than previously. Also, variable amounts of C from dead biomass are not fully decomposed but become effectively sequestered in the soil for long periods of time. As a result, the system functions as a self-sustaining C sink, demonstrating why simply maintaining forest cover is an effective climate mitigation strategy. Moreover, the science and practice of silviculture offers proven techniques for managing tree populations and their growth and mortality for multiple objectives, including maintaining and improving C sinks in the face of diverse stressors and drivers of change.

The role of forests in global C cycling is second only to that of the oceans. However, acre-for-acre, no other biome on Earth sequesters more C than tropical rainforests. While the rate and magnitude of C cycling varies dramatically among forested biomes, from the boreal taiga to tropical rainforests, the existence of a forest is almost always indicative of a stronger C sink relative to any other land use (or land cover type). Globally, this ecosystem service is impacted most directly by deforestation (conversion of forest to another land use) especially for agriculture and development of human settlements in the tropics. Practices such as afforestation (establishing forests on lands not previously forested) and reforestation (establishing forests on lands previously forested) that increase overall forest cover, as well as regulations, policies, and landowner incentive programs that promote conservation and sustainable use of extant forests, have become a priority.

The total amount of C currently stored in forests in the United States has been estimated at 100,000 teragram (Tg) C (Domke et al. 2018, EPA 2018). Recent increases in stored C over the previous decade are due mostly to increases in aboveground biomass in eastern forests, including those in New York State (Domke et al. 2012).

The storage of C per unit of forested land area (C density) varies across the United States and has been estimated to average about 142.5 megagrams (Mg) C per hectare. The highest storage per hectare in the United States is in the Pacific Northwest, the Northeast (including New York State), Alaska's coastal forests, and the Upper Midwest (EPA 2018).

Overall, the forests in the State provide a net C sink, meaning they remove and store more CO_2 from the atmosphere than they release (King et al. 2015). This conclusion is made on the basis of field inventory data that shows, on average across New York State, there has been net growth of forest biomass over time, after factoring in mortality due to natural causes and harvesting. Harvest and use of long-lived wood products also contribute to net emission reductions from the forest sector. Some of that net C uptake into woody materials is offset by subsequent C emissions caused by wood processing, decay, and combustion. Net C emissions are increased by conversion of land cover from forest to all other land uses (i.e., deforestation), including agriculture and settlements.

Net ecosystem production (NEP) reflects the balance between photosynthetic C uptake (gross primary production or GPP) and CO_2 release via respiration by plants, microbes and animals. From a scientific standpoint, if NEP is positive, the ecosystem is considered a net C source, and if NEP is negative the ecosystem is a net C sink. However, C accounting protocols typically do not include all forest C pools and fluxes nor attempt to estimate net ecosystem exchange (NEE), but rather focus on changes in aboveground biomass of vegetation as a proxy for estimating source/sink status and magnitude. Such approaches are targeted more at estimating what ecologists know as net primary production (NPP), or the amount of C sequestered in plant biomass (i.e., new growth) after plant respiration (releasing CO_2) has occurred. This measure of ecosystem performance is most directly analogous to growth. All other forms of respiration, by microbes and fungi that promote decomposition, as well as all other biota, must be deducted from NPP to calculate NEE. These individual sources of CO_2 flux to the atmosphere are difficult to measure but the overall C flux (NEE) can be estimated by use of expensive instrument arrays such as eddy flux covariance towers.

The NEE of an undisturbed forest is generally positive but highly variable over space and time. Net growth can be decreased by prolonged drought (Domke et al. 2018). Forest harvest and fire remove biomass (living and dead) and facilitate CO_2 loss via microbial respiration as residual wood decays. Fire releases C to the atmosphere, mainly as CO_2, with some additional emissions of CH_4. Carbon is released to the atmosphere post-harvesting by way of processing removed wood or from biofuel energy. Some of this C is retained in lumber and other wood products and can be stored for decades or longer in buildings or landfills (Domke et al. 2018). The movement of C from forest to wood product is not depicted as, and is not intended to represent, a source or a sink, but rather a lateral transfer.

Net ecosystem exchange in forest lands can be estimated using atmospheric models (cf., Peylin et al. 2013), calculations based on tree inventories (cf., Pan et al. 2011), eddy diffusion measurements (cf., Amiro et al. 2010), or ecosystem modeling (Sitch et al. 2015, Domke et al. 2018). Fluxes estimated in these ways can show substantial variability across the landscape and over decadal time scales (Williams et al. 2016).

In most temperate and high-latitude forest biomes, forest soils contain more C than trees and other terrestrial biota. However, the historical C pools in forest soils have been depleted by land use practices, particularly the clearing of forest land for agriculture and grazing, as well as forestry operations prior to (or in the absence of) contemporary best management practices. Soil C pools in forested wetlands are often depleted as a result of wetland draining, which rapidly increases decomposition in soils from which the water table has receded. The organic horizons of the soil are comprised of decomposing plant residues, soil biota, and materials synthesized by soil biota (Lajtha et al. 2018). Soil organic matter is important for storing nutrients in the soil, holding moisture that supports plant growth, and providing soil structure (Oldfield et al. 2015, Lajtha et al. 2018). Because both plants and all other organisms responsible for decomposition of soil organic matter respire (emit) CO_2 under aerobic conditions, substantial fluxes of C from soils to the atmosphere exist as a biogenic source of emissions. There is additional release of CO_2 to the atmosphere (known as "off-gassing") from surface waters that receive C in the form of dissolved organic matter (DOM), a byproduct of decomposition, via lateral transfer from forest and wetland soils (Ciais et al. 2013). The latter movements of DOM from the land to surface waters are commonly measured as part of water quality testing, and those data can be used (with varying degrees of confidence) to estimate or model this lateral transfer at the watershed or landscape level. Off-gassing, in which DOM gives rise to CO_2 gas bubbles that are released from the water surface to the air, is a much more difficult process to quantify, but it is generally believed to be only a minor source of CO_2 to the atmosphere.

Erosion from agricultural lands, and to a lesser extent from managed forest lands, contributes C to drainage water (Berhe et al. 2007). Erosion strips the soil of some of its organic matter and redeposits it where it can more readily be oxidized (Lal 2003). The erosion potential of soils is highly variable and depends on such factors as slope, aspect, soil texture, vegetation coverage, and soil disturbance.

1.4 Agriculture

Unlike some other sectors, N_2O and CH_4 are dominant GHGs in the agriculture sector. The C and N cycles are closely linked, so that management practices that affect one affect the other. All major agricultural management decisions influence the C and N cycles and the associated emissions of CO_2, CH_4, and N_2O to the atmosphere. These include, among others, crop type, crop rotation, tillage, fertilizer application, irrigation, residue treatment, and manure management (Paustian et al. 1997, Smith 2008, Lajtha et al. 2018). Organic C in soil and biomass is lost when forest land is converted to agricultural land and gained when agricultural land reverts to forest, which was an important historical trend during the last century in the northeast United States (Woodbury et al. 2007). This C loss is mainly attributable to a decrease in the steady sequestration of C in perennial biomass on site (i.e., trees) combined with increased microbial decomposition due to soil disturbance during tilling, and increased surface runoff and erosion (Paustian et al. 2016, Lajtha et al. 2018). Historically, tillage has reduced soil C stocks by an estimated 25% (Woodbury et al. 2007), and this soil C can be increased if land reverts to perennial vegetation.

Crop yields have increased over time due to the use of improved varieties and increased management, especially the use of fertilizer. However, the use of N fertilizer has also caused water pollution, air pollution, and emission of N_2O, a potent GHG. Increases in the intensity of livestock production have also posed challenges for managing manure, including increases in emissions of both CH_4 from manure storage and N losses to the environment, including N_2O (Wightman and Woodbury 2016).

Many agricultural practices can significantly reduce GHG emissions (Fargione et al. 2018) and many of these opportunities are low cost (Woodbury 2018). Many of these practices also have other important benefits, including improved air quality, improved water quality, improved soil quality, and increased biodiversity (Fargione et al. 2018). For example, winter cover crops can increase C sequestration, improve soil health, and increase crop yields. The same benefits are obtained with overwintering double crops, with the added benefit of increasing agricultural production. Another example is using precisionagricultural techniques to apply fertilizer only when and where the crop needs it (McLellan et al. 2018, Sela et al. 2018). These improved practices greatly reduce N_2O emissions, with additional benefits for soil, air, and water quality (Fargione et al. 2018, Woodbury 2018). Thus, while agriculture emits substantial amounts of GHGs, there are also many opportunities to reduce emissions with improved practices. There is also the potential to reforest substantial amounts of former agricultural land, sequestering C in trees and soil.

1.5 Wetlands

For the purposes of this report, nontidal and freshwater wetlands are referred to as terrestrial wetlands, in keeping with the terminology of the wetland chapter of the Second State of the Carbon Cycle Report (SOCCR2; Kolka et al. 2018). These wetlands are characterized by high water table and surface soils that are periodically saturated by ground water or surface water, supporting vegetation adapted to periodically or entirely saturated conditions (EPA 2015). Terrestrial wetlands can be subdivided into peatland systems and mineral soil systems and/or as forested versus non-forested wetlands (Kolka et al. 2018).

Fresh surface waters play important roles in GHG emissions and sinks. Carbon moves out of inland waters to the atmosphere, mainly as CH_4 and CO_2 gases. In addition, C moves by way of lateral transfer through river systems to estuaries and coastal environments (Butman et al. 2018). The burial of C in lake and reservoir sediments is a relatively small sink. Butman et al. (2018) estimated that, on average, 20.6 grams (g) C per year (yr) is emitted per square meter (m^2) of continental land area in the United States via flux of C from inland waters to the atmosphere.

Estuaries process total organic carbon (TOC) inputs that enter via river flow from the estuary watershed and from tidal wetlands (Raymond and Bauer 2001, Bauer et al. 2013). In the act of microbial processing of TOC, CO_2 is emitted to the atmosphere. Estuaries also store C in their sediments for short or long periods of time (Mcleod et al. 2011, Hopkinson et al. 2012). Windham-Myers et al. (2018) estimated C density in tidal wetland sediment in the Atlantic coast region of the United States of 2 kilograms (kg) C per m^2 (in the top 1 m of sediment or soil), based on a literature review. The C loss rate for the continental United States (CONUS) was estimated by Couvillion et al. (2017) to be 1.8 Tg C per year for tidal wetlands.

Coastal wetlands often exhibit lower CH_4 emissions than do freshwater wetlands (Chmura et al. 2003). This is, in part, because methanogenesis is suppressed in saline wetlands due to high sulfate concentrations in salt water (Bartlett et al. 1987).

Terrestrial wetlands can act as both sources and sinks of CH_4 (Harriss et al. 1982). Under flooded conditions, the wetland can be a net CH_4 source to the atmosphere because soil microbial populations are typically net producers of CH_4 due to the relative lack of oxygen under water saturation. During dry periods, wetland soil can become a net CH_4 sink due to CH_4 oxidation.

Bonneville et al. (2008) demonstrated that the NEE of CO_2 in a temperate cattail marsh in Ontario was variable intra-annually. The marsh functioned as a net CO_2 sink from June to September, and a net source for the remaining months of the year.

Temperate tidal salt marshes can function as C sinks with long-term C storage potential (Artigas et al. 2015). Carbon burial in tidal wetland sediment is expressed as the amount of C accumulated in sediment per unit time (for example, in $g/m^2/yr$). It reflects all forms of C, including old and new C, derived onsite or transported down-river to the wetland (Windham-Myers et al. 2018). The C accumulation rate (flux) in tidal wetlands was estimated by Windham-Myers et al. (2018) for the Atlantic coast region including New York State as 126 $g/C/m^2$ per year.

Preliminary estimates of C fluxes from the Laurentian Great Lakes suggest that Lakes Superior, Huron, and Michigan may be net C sources, whereas Lakes Erie and Ontario may be net sinks (Butman et al. 2018). Robust flux estimates for the Great Lakes are not yet available. Severe human-caused disturbances produced by nutrient inputs and invasive species complicate budget estimates for these ecosystems.

2 Stocks and Fluxes

2.1 Forests

Terrestrial plants and soils absorb about one-fifth of human-caused GHG emissions. However, this important C sink is partially offset or augmented by forest growth, forestry practices, agriculture, and land-use changes. Extensive logging and regrowth are important drivers of C fluxes, although tree harvesting at moderate intensity probably does not have a large impact on the amount of C stored in forest soils (Johnson and Curtis 2001). Carbon stocks and fluxes in forest systems include five storage pools (IPCC 2006), identified as the following:

- aboveground biomass
- belowground biomass
- dead wood
- litter
- soil organic C

Amiro et al. (2010) evaluated 180 site years of eddy covariance measurements of CO_2 flux at forested sites throughout North America. This study revealed that when a forest experiences a stand-replacing disturbance, such as a fire, harvest, or windthrow event, there is typically a brief but variable period that the stand becomes a net C source to the atmosphere, followed by transition back to a C sink as vegetation regenerates. The magnitude and timing of these transitory fluxes vary widely based on site factors, including the type and severity of the disturbance itself.

The Earth's forests have retained and sequestered human-caused CO_2 emissions (Pan et al. 2011). This global forest C sink has partly been driven by the joint fertilizing effect of increased atmospheric CO_2 concentration and atmospheric N deposition (Bellassen and Luyssaert 2014). Increases in temperature, changes in precipitation, and changes in forest management may have been less important globally thus far.

Changes in forest C sinks are largely due to tree growth and regrowth, harvest, fertilization from increased atmospheric CO_2, and atmospheric deposition of N (Ciais et al. 2008, Bellassen et al. 2011, Schultz 2011, Williams et al. 2012, Ciais et al. 2013). Decomposition of harvest residue (slash) and roots contribute to increases in short-term emissions of CO_2 via microbial respiration. However, it can take many years, or longer, for wood use (which stores C) to compensate for these emissions. Therefore, delaying or avoiding harvest might increase C sequestration. This is a topic of active debate in the scientific community. Actions such as replacing low-productivity forest stands, planting resilient tree species, and introduction or addition of N-fixing plants might further mitigate human-caused GHG emissions (Bellassen and Luyssaert 2014).

The largest terrestrial C store is soil organic C (SOC). Mitigation of atmospheric GHGs must consider the dynamics of this C pool (Jobbágy and Jackson 2000, Lal 2003, Tian et al. 2016). Fluxes of CO_2e respond to changes in land use, management practices, and changing climate (Guo and Gifford 2002, Davidson and Janssens 2006, Heimann and Reichstein 2008, Nave et al. 2013, Domke et al. 2017). Forest SOC in the United States has been inventoried since 2001 by the Forest Inventory and Analysis (FIA) program of the USDA Forest Service.

2.2 Agriculture

The agriculture sector removes CO_2 from the atmosphere through the process of photosynthesis during the growth of crops. In intensively managed cropping systems, such removal can be at a much higher rate than in natural ecosystems due to selection of fast-growing species and cultivars and intensive management practices, including the use of fertilizers to enhance growth rates. However, most of the fixed C is harvested and does not remain in vegetation or soils and therefore emissions of GHGs from the sector exceed removal from the atmosphere such that agriculture represents 8.4% of total net anthropogenic GHG emissions (EPA 2019). The agriculture sector emits GHGs through many different processes related to C and N cycling for both crops and livestock in agroecosystems. For national and New York State GHG inventories, the major reporting categories are (1) enteric fermentation in livestock, (2) manure management, and (3) soil management (EPA 2019). Methane, N_2O, and CO_2 are the major greenhouse gases emitted from agriculture. In the United States, CH_4 emissions from enteric fermentation and manure management represent 36% of total anthropogenic CH_4 emissions (EPA 2019). In the United States, N_2O emissions from agricultural soil management practices represent 74% of total anthropogenic N_2O emissions (EPA 2019). Related GHG emissions occur both upstream of the sector, for example in manufacturing of fertilizer, lime, and farm equipment, and also downstream of the sector, for example, through emission of N_2O from nitrate (NO_3) leached from farms into surface waters that is later volatilized to N_2O and from food waste that is disposed of in landfills. The agriculture sector differs from the forest and wetland sectors because of the comparatively high level of management intensity of agricultural systems for both crop and livestock production.

2.3 Wetlands

Relatively undisturbed tidal wetlands store large quantities of C that has been fixed both on-site and off-site (Canuel et al. 2012). The latter includes C delivered to the wetland via stream or river transport. However, human development in coastal areas contributes to removal of some of that stored C in association with loss of tidal wetlands to human development and sea level rise. The main C flux from terrestrial wetlands is CO_2 exchange with the atmosphere in both forested and non-forested wetlands (Kolka et al. 2018). In general, forested wetlands sequester more C than non-forested wetlands per unit area, and peatlands sequester more than mineral soil wetlands. Peatlands are comprised of fens and bogs. Fens derive most of their water from ground water, while bogs derive most of their water from precipitation and tend to have low pH and sphagnum moss vegetation.

Windham-Myers et al. (2018) estimated that the export of total C from tidal wetlands in the United States is about 421 g C/m^2 per year. This estimate was based on the summary of Hermann et al. (2015; 185 g C/m^2 per year, average TOC exchange at 12 sites in the eastern United States) and estimates of dissolved inorganic C (DIC) exchange at four eastern United States sites (average of 236 g C/m^2 per year), based on Najjar et al. (2018).

It is known that estuaries are important sources of GHG, both CO_2 and CH_4, from the water surface to the atmosphere. In contrast, tidal wetlands constitute one of the largest C sinks per unit area (Windham-Myers et al. 2018). This is largely because TOC tends to accumulate in tidal wetland sediments, and this can be influenced by sea level rise (Chmura et al. 2003). The land-sea interface is dynamic. It is influenced by erosion, accretion, and both short (e.g., storm surge) and long-term (global warming) changes in sea level. Other important factors include the sediment deposition rate, the balance between photosynthesis and respiration in these highly productive environments, and shoreline land slope (Cahoon 2006, Windham-Myers et al. 2018).

Wetlands in the United States are stressed by human activities related to plant removal, soil compaction, and drainage (EPA 2016b). There was a reduction in the wetland area in the United States of more than 50% between 1870 and 1980 (Dahl 1990). Other existing wetlands were degraded. More recently, the rate of wetland loss nationwide has been a small fraction of the historical loss (USFWS 2011).

Observed and modeled GHG fluxes from a watershed vary substantially with hydrology, especially during severe or long-lasting high-water events or droughts. As the wetland water table comes closer to the surface, the CO_2 flux usually decreases and the CH_4 flux increases (Olson et al. 2013). Such changes in hydrology have effects on the prevalence of aerobic versus anaerobic conditions, with substantial influences on the wetland CH_4 budget, thereby altering the rate of organic matter decomposition (Drexler et al. 2009). Drainage promotes decreased CH_4 flux, increased CO_2 flux, and less C storage (Bridgham et al. 2006).

Anaerobic conditions in wetlands decrease the rate of organic matter decomposition and alter GHG fluxes. The CH_4 flux is controlled largely by water table position, oxygen supply, soil temperature, and vegetation type (Bansal et al. 2016, Hanson et al. 2016).

Peatland C accumulation and C cycling vary with precipitation patterns and groundwater dynamics. Peat C accumulation in North America commonly ranges between about 20 and 30 g C/m^2 per year (Manies et al. 2016). When the water table lowers, CO_2 production and decomposition increases and CH_4 production decreases (Waddington et al. 2015). Changes in hydrology also affect fire periodicity and intensity, further influencing the C cycle. Increasing temperature caused by climate change might change the GHG source strength of terrestrial wetlands in the future (Kolka et al. 2018).

3 Methods

Preliminary estimates of emissions and mitigation potential are presented in this report in units of million metric tons (MMt; equivalent to Tg) of CO_2e. In general, the following 100-year GWP values were used: $CH_4 = 25$ and $N_2O = 298$. However, some values taken from the literature and reported here may be based on other 100-year values. In other words, in some cases the best available estimates from the published literature were used, and these published sources may have used other GWP values. However, any such differences should not change the overall conclusions as differences among different versions of 100-year GWP values are not large. Also, it should be noted that for short-lived gases such as CH_4, more recently derived GWP values are considerably higher than previously estimated. As such, categories that include CH_4 would be relatively larger if these more recent higher GWP values were used (e.g., enteric fermentation and manure management via storage).

In this report, positive values for a given sector or sector component indicate a net addition of GHG emissions to the atmosphere and negative values indicate net reduction of GHGs from the atmosphere. Estimates of emissions and emission reductions presented are considered preliminary and are subject to revision in future analyses. A more comprehensive analysis is highly recommended.

3.1 Forests

3.1.1 CO_2 Flux

Forest area estimates were compiled following Ogle et al. (in preparation) and Bechtold and Patterson (2005). Carbon densities in aboveground and belowground biomass were estimated following Woodall et al. (2011) and Birdsey (1996). The dead wood pool includes standing and downed dead wood and litter. Carbon density in standing dead wood was estimated following Woodall et al. (2011) and Domke et al. (2011). Carbon densities in downed dead wood were estimated following Domke et al. (2013), and C density in litter was estimated following Domke et al. (2016). Soil organic C density was estimated following Domke et al. (2017).

Estimates of net CO_2 flux (reported as CO_2e) associated with forest land in New York State were compiled using the same methods as the forest land for the United States in the latest National Greenhouse Gas Inventory (EPA 2019). Carbon stocks associated with forest ecosystems were generated based on data from the network of annual national forest inventory (NFI) plots established and measured by the FIA program of the USDA Forest Service (Frayer and Furnival 1999, USDA Forest Service 2018a, c, b).

Forest dynamics associated with forest growth/aging and disturbances (e.g., harvesting, fire, wind, and insect infestation) and land-use dynamics associated with afforestation and deforestation are all included in the compilation of the estimates. All FIA plots in New York State have been remeasured during recent years, which allows for direct incorporation of disturbance and land-use change impacts on the net CO_2-C flux associated with New York State forests. The forest C stock estimates were obtained from a design-based probability sample and are assumed to be representative of all forest land in New York State. Forest land was comprised of two common reporting format (CRF; IPCC 2006) categories: Forest Land Remaining Forest Land (FRF; CRF category 4A1; forested for at least 20 years) and Land Converted to Forest Land (LCF; CRF category 4A2; land converted to forest within the previous 20 years). Forest land was defined as the following:

> Land at least 120 feet (37 meters) wide and at least 1 acre (0.4 hectare) in size with at least 10 percent cover (or equivalent stocking) by live trees including land that formerly had such tree cover and that will be naturally or artificially regenerated. Trees are woody plants having a more or less erect perennial stem(s) capable of achieving at least 3 inches (7.6 cm) in diameter at breast height, or 5 inches (12.7 cm) diameter at root collar, and a height of 16.4 feet (5 meters) at maturity in situ. The definition here includes all areas recently having such conditions and currently regenerating or capable of attaining such condition in the near future. Forest land also includes transition zones, such as areas between forest and non-forest lands that have at least 10 percent cover (or equivalent stocking) with live trees and forest areas adjacent to urban and built-up lands. Unimproved roads and trails, streams, and clearings in forest areas are classified as forest if they are less than 120 feet (36.6 meters) wide or an acre (0.4 hectare) in size. Forest land does not include land that is predominantly under agricultural or urban land use. (Oswalt et al. 2014)

The NFI consists of permanent base-intensity ground plots which are measured every five to seven years and are distributed approximately every 6000 ac. Site- and tree-specific (e.g., diameter at breast height [DBH], tree height) variables are measured on all base-intensity plots with at least one forest land condition (i.e., domains mapped on each plot using land use, forest type, stand size, among other site variables; there may be multiple conditions on a single inventory plot). Intensive plots constitute a subset of the base-intensity plots (every 16th base-intensity plot is an intensive plot) and are sampled for a wider variety of forest health metrics (e.g., crown condition, soil chemistry, down woody material) in addition to the tree size measurements made on base-intensity plots.

Expansion factors, which expand plot-level estimates to population-level estimates were used to estimate both area and C stocks and stock changes in the NFI (Bechtold and Patterson 2005). Specifically, the area of an FIA plot is known, as is the area of the conditions on each plot. Since each plot represents approximately 6000 acres (ac), the proportion of each plot that is forested is expanded over the 6000 ac to obtain an estimate of forest land area. In a similar way, each tree on a plot has an expansion factor based on the size of the plot relative to the size of the area that plot represents across the landscape. Plots are grouped into similar strata (e.g., percent canopy cover) within the population as a mechanism to reduce the variance of population estimates. Net CO_2-C fluxes associated with forest ecosystems were developed from C stock estimates for each year during the period 1990–2017. Age transition matrices were used to define the proportion of forest area among different age classes that changes to a different age class during a given time step. They were used to incorporate the combined effects of forest growth and disturbance (e.g., tree harvesting, fire, windthrow, and insect infestations) on forest C stocks through time (EPA 2018).

Forest ecosystem C stocks associated with live aboveground and belowground biomass (live trees and understory vegetation), dead wood (standing dead trees and downed dead wood), litter (duff, humus, and fine woody debris), and soil organic C were determined for individual years between 1990 and 2017. The net CO_2-C flux for a given year was determined based on the C stock-difference between two adjacent years (IPCC 2006). Methods used to derive C stocks associated with the various pools of forest ecosystem C are described in the sections below.

3.1.2 Live Aboveground and Belowground Biomass

3.1.2.1 Trees

Living trees with DBH ≥ 2.54 cm are included in the C pool. The component ratio method (CRM; Woodall et al. 2011) was used based on measurements from FIA plots (Phase 2) used for estimating aboveground biomass. The CRM method uses regional volume models and information on specific gravity to estimate individual tree sound volume (i.e., stem volume after deductions for rotten/missing portions of bark and bole). The bole sound volume was then used to estimate biomass of tree tops, branches, and coarse roots using adjusted component proportions based on Jenkins et al. (2003). Stump biomass was estimated from equations described by Raile (1982). Estimates of foliage biomass were included based on an analogous approach to the CRM method (EPA 2019). Tree sapling (DBH ≥ 2.54 cm and < 12.7 cm) biomass was determined from observed DBH and an adjustment factor as describe by Woodall et al. (2011). Estimated oven-dry biomass of all components was multiplied by

0.5 for conversion to C, based on the assumption that 50% of the dry biomass weight is comprised of C (IPCC 2006).

3.1.2.2 Understory Vegetation

Understory vegetation was defined as all biomass of undergrowth plants, which includes herbaceous plants, woody shrubs, and trees < 2.54 cm DBH (Woodall et al. 2015). The ratio of understory C density to live tree (aboveground and belowground) density was determined according to Jenkins et al. (2003), as given in Equation 1.

Equation 1 $\quad$ **Ratio = e(A − B × ln(live tree C density))**

where "live tree C density" is expressed in t C/ha derived from information in Birdsey (1996). These ratios were then multiplied by tree C density on each plot to generate an estimate of understory vegetation biomass. The coefficients "A" and "B" are specific to a given forest type (Table 2). Forest types for the Northeast region were defined according to Smith et al. (2003). If the model estimated ratio was greater than the maximum ratio (Table 2), the ratio was set to the value for the maximum ratio.

Table 2. Coefficients for Estimating the Ratio of C Density of Understory Vegetation (Aboveground and Belowground, t C/ha)[a] in the Northeast Region

Forest Type[b]	A	B	Maximum Ratio[c]
Aspen-Birch	0.855	1.032	2.023
MBB/Other Hardwood	0.892	1.079	2.076
Oak-Hickory	0.842	1.053	2.057
Oak-Pine	1.96	1.235	4.203
Other Pine	2.149	1.268	4.191
Spruce-Fir	0.825	1.121	2.14
White-Red-Jack Pine	1.000	1.116	2.098
Non-stocked	2.020	2.020	2.060

[a] See equation 1

[b] Regions and types as defined in Smith et al. (2003).

[c] Any estimate predicted by the model to be greater than the maximum ratio is set equal to the maximum ratio.

3.1.2.3 Dead Wood

Carbon stock estimates for standing dead wood were primarily based on NFI plot-level measurements and the CRM method used for live trees (Domke et al. 2011, Woodall et al. 2011), including additional methods to account for decay and other structural losses, which can significantly affect C mass of standing dead wood (Domke et al. 2011, Harmon et al. 2011). This C pool includes aboveground and belowground (coarse root) mass and includes trees of at least 12.7 cm DBH.

Downed dead wood is defined as individual pieces of dead wood > 7.5 cm in diameter at point of observation along the NFI plot transect (unattached to standing dead trees); stumps and roots of harvested trees are also included (Woodall et al. 2015). The ratios applied for estimating downed dead wood by forest type for the northeast region are provided in Table 3. This ratio, for a given forest type, is multiplied by the estimated live tree density to estimate the downed dead wood C density in units of C/ha. Coefficients for estimating the logging residue component of downed dead wood are given in Table 4. The downed dead wood estimates reported here also incorporate measurements of downed dead wood sampled on a subset of NFI plots (Domke et al. 2013, EPA 2019).

Table 3. Ratio for Estimating Downed Dead Wood by Forest Type in the Northeast Region as Defined by Smith et al. (2003)

Forest Type	Ratio[a]
Aspen-Birch	0.078
MBB/Other Hardwood	0.071
Oak-Hickory	0.068
Oak-Pine	0.061
Other Pine	0.065
Spruce-Fir	0.092
White-Red-Jack Pine	0.055
Non-stocked	0.019

[a] The ratio is multiplied by the live tree C density on a plot to produce downed dead wood C density (t C/ha).

Table 4. Coefficients for Estimating Logging Residue Component of Downed Dead Wood in the Northeast Region as Defined by Smith et al. (2003)

Forest Type Group[a]	Initial C Density (t/ha)	Decay Coefficient
Hardwood	13.9	12.1
Softwood	12.1	17.9

[a] Forest types are according to majority hardwood or softwood species.

3.1.2.4 Litter Carbon

Carbon stock estimates for litter (duff, humus, and fine woody debris < 7.5 cm) of forest ecosystems were based on relations between measurements on a subset of NFI plots and plot attributes including geographic position, elevation, forest type group, live aboveground biomass, precipitation, temperature, and potential evapotranspiration (Domke et al. 2016). These estimates were updated for each year during the period 1990–2017 according to changes in aboveground live tree C.

3.1.2.5 Soil Organic Carbon

Carbon stock estimates associated with SOC in forest ecosystems represent the upper 1 m of soil (Domke et al. 2017). Measurements of SOC available from a subset of NFI plots (0–20.32 cm) and the International Soil Carbon Network (0–100 cm)[2] were used to develop estimates of SOC (0–100 cm) for this subset of NFI plots (Domke et al. 2017). Relations between these SOC estimates and NFI plot attributes—including geographic position (latitude, longitude, elevation); forest type group; aspects of mean annual air temperature, precipitation, and evapotranspiration; soil order; and surficial geologic type—were used to develop a statistical model to predict SOC at all NFI plots (Domke et al. 2017).

3.1.3 Non-CO_2 Emissions

Emissions of CO_2 associated with fire are addressed in CO_2-C flux estimates for individual pools of forest ecosystem C generated according to the above described methods. Emissions of GHGs other than CO_2 (e.g., CH4 and N2O) associated with fire were determined from Equation 2 (IPCC 2006), which used estimates for the extent of burned area, available fuel, a combustion factor, and the gas-specific emissions factors listed in Table 5 and followed the generalized equation:

Equation 2 **Emissions =Area Burned x Fuel Available x Combustion Factor x Emission Factor x 10^{-3}**

Area burned was derived from Monitoring Trends in Burn Severity (MTBS; Eidenshink et al. 2007, MTBS Data Summaries 2018) and National Land Cover Data (NLCD; Homer et al. 2015).

Table 5. Emission Factors for CH_4, N_2O, and CO_2 Forest Burning

Emission Factor (g per kg dry matter burned)[a]	
CH4	4.7
N2O	0.26
CO2	1,569

[a] IPCC (2006)

Fuel availability (i.e., live aboveground biomass, dead wood, and litter) was estimated from data collected on NFI plots. Specification of combustion factors and other details associated with development of non-CO_2 emissions from fires are described in EPA (2019).

3.2 Agriculture

Preliminary estimates of GHG emissions and GHG mitigation potential for the agricultural sector in New York State were derived from published sources whenever feasible, supplemented with calculations based on data provided in published sources. The focus was primarily on the largest emissions sources and the largest mitigation opportunities. Estimates of smaller sources and mitigation opportunities are also included when available.

Estimates are divided into agricultural categories and sub-categories to facilitate use and to help avoid double-counting. All categories and sub-categories are closely related to agriculture. However, some sub-categories were identified as "possibly outside the sector" if they were (1) potentially occurring outside of New York State (e.g., production of N fertilizer in another state) or (2) potentially already included in an inventory of another sector in the State such as fossil fuel combustion emissions (e.g., field tractor emissions).

Brief summaries of key aspects of the methods are provided in the sections below, including the category definition, important methods assumptions, and key data or literature sources. Methods for specific categories and sub-categories of emissions are presented.

3.2.1 Category: Enteric Fermentation

Methane emissions from livestock operations occur as part of the normal digestive process in ruminant livestock. Methane is produced by bacteria that break down carbohydrates in the rumen by enteric fermentation. For estimating recent GHG emissions, the EPA National GHG Inventory (EPA 2019), based on data from 2017, was used. For estimating mitigation potential, results of a modeling study from the literature conducted for representative 1,500-cow and 150-cow farms in New York State were used (Veltman et al. 2018). From this study, a combined "feed mitigation" strategy that included four dietary practices was selected. These practices were (1) 50% forage, (2) neutral detergent fiber digestibility, (3) high feed efficiency, and (4) reduced protein. The combined "feed mitigation" strategy increased milk production (~11%) and farm profitability (~37%) and reduced the C footprint (~22%). The C footprint reduction for a 150-cow farm was about 25%. Half of that benefit was from replacing forage with grain. This mitigation pathway assumes that increased efficiency of milk production will reduce total emissions in New York State by allowing fewer dairy cows to be more productive.

3.2.2 Category: Manure Management (Storage)

This category represents manure that is stored and treated on livestock operations. See also Manure Management (field) below. Emissions from different types of manure storage units were estimated using methods developed primarily by the EPA (2019) and modified based on data and methods specific for New York State (Wightman and Woodbury 2016). Estimates of the amount of manure stored in different types of storage units were taken from published data and Confined Animal Feeding Operations (CAFOs) in the State (Doug Ashland, personal communication, New York State Department of Environmental Conservation, 2019).

Mitigation potential was estimated following previously published methods (Wightman and Woodbury 2016), updated with more recent data on livestock numbers (USDA-NASS 2019). The mitigation potential is based on a scenario of covering and flaring only manure stored in liquid manure storage units, thus capturing the CH_4 and combusting it to CO_2, greatly reducing the GWP (for details, see Wightman and Woodbury 2016).

3.2.3 Category: Agricultural Soil Management

This category includes effects of crop production practices such as tillage, fertilization with inorganic fertilizers, application of organic materials such as manure, and crop harvest. This is a large category of emissions, evidenced by the long list of sub-categories listed in Table 6 and described briefly below. Some of these sub-categories may be included in other sectors, such as emissions from fossil fuel combustion by tractors and other farm equipment. These categories are identified in Table 6 as "possibly outside the sector."

For many of the sub-categories, GHG emissions for individual crops were estimated based on published literature specific to New York State (Wightman et al. 2015a). Previously published life cycle analyses of GHG emissions for maize and soybeans were used (Wightman et al. 2015a), as well as bioenergy feedstock grasses such as switchgrass (Woodbury et al. 2010). Some emissions may be considered outside the agricultural sector; these are indicated as such in Table 6. For other New York State crops, the published emissions estimates for maize and soybeans were modified based on similarities and differences in management between crops. For example, an estimate of GHG emissions from wheat was developed based on that for maize by assuming that most sub-category emissions were identical (such as tillage and harvest), but that N-related emissions were lower based on the ratio of average wheat yield to maize yield. For other categories, the most suitable published estimates were used. Cost estimates for some mitigation sub-categories were provided, when available, under the "Mitigation Cost" field of Table 6. Further brief definition and explanation is provided below for each sub-category.

3.2.3.1 Sub-Category: Manure Management (Field)

This sub-category includes manure that is applied to fields year-round and assesses direct and indirect N_2O emissions from volatilization, deposition, and leaching. See also Category Manure Management (storage) above.

3.2.3.2 Sub-Category: Field Emissions from Liming

Portions of New York State crop fields are too acidic for optimal crop production and require regular lime application for cultivation of most crops. This category reflects direct emissions of CO_2 from crop fields due to liming. The emissions estimate is based on published and unpublished analyses conducted for the State, as described above. A mitigation estimate was not provided because acidic soils require lime in order to produce most crops.

Table 6. Preliminary Estimates of Greenhouse Gas Emissions from New York State Agriculture

Units are million metric tons of carbon dioxide equivalents per year. Positive values are emissions, negative values are mitigation or sequestration.

Category & Sub-Categories	Recent Year Emission		Mitigation Potential		Possibly Outside Sector?	Comments
	Category	Sub-Category	Category	Sub-Category		
	------------------ MMt CO_2e/yr ------------------					
Enteric Fermentation	3.71		-0.70			Mitigation is from improved diet management.
Manure Management (storage)	2.17		-1.29			2030 emissions & mitigation potential will increase.
Agriculture Soil Management	4.08		-1.68			
Manure management (field)		0.01		0.00		Partially counted under crop N_2O.
Field emissions from liming		0.28		0.00		Liming is required for soils that are too acidic.
Crop N_2O emissions (direct & indirect)		2.20		-0.20		Mitigation includes a small amount upstream.
Cover crops		?		-0.85		Could include double crops.
Reduced tillage		0.00		0.00		Some technical potential, but not permanent.
Drained wetlands		0.07		?		N_2O emissions only, should add CO_2 & CH_4.
Replace annual with perennial crops		n/a		-0.62		Technical potential but may not be feasible.
Equipment (fuel)		0.26		0.00	Yes	
Equipment (embodied)		0.09		0.00	Yes	
Production of herbicide, P, K, seed		0.18		0.00	Yes	
Production of lime		0.79		0.00	Yes	
Production of synthetic N		0.19		?	Yes	Technical potential, but outside NY.
Reduce Food Waste	n/a		-1.19		Yes	Only counts NY agricultural production mitigation.
Farm Energy Conservation	?		?		Yes	Farm energy use emissions not included herein.
Wind & Solar Energy on Agricultural Land	?		?		Yes	Also involves some land use change.
Avoided Grassland Conversion	n/a		0.00			Recent data show no net grassland conversion.
Forested Riparian Buffer	-0.06		?			Mitigation feasible if riparian buffer area increased.
Alley Cropping	0.00		-0.67			
Bioenergy	?		?		Yes	Mutually exclusive with reforestation (same land)
Reforestation of Former Agricultural Land	0.00		-4.90			Technical potential but needs further research.
TOTAL	**9.90**		**-10.43**			
TOTAL, IN SECTOR ONLY	**8.38**		**-9.23**			

3.2.3.3 Sub-Category: Crop N_2O Emissions (Direct and Indirect)

Nitrous oxide emissions occur directly from crop fields. Additionally, some N is lost from fields in other chemical forms via volatilization, leaching through the soil profile, and runoff. A fraction of this lost N is subsequently denitrified and emitted to the atmosphere as N_2O "downstream" of the crop production as "indirect emissions." The N_2O emissions estimate is based on New York State data and accounts for direct and indirect emission of field N_2O. Estimates of emissions, mitigation potential, and mitigation costs were derived from a national estimate (Fargione et al. 2018) that was developed and downscaled to the state level in the study reported here. Mitigation potential based on future improved management of N fertilizer was estimated. A portion of the estimated mitigation is due to reductions in upstream GHG emissions from manufacturing activities, which is outside the sector and the State.

3.2.3.4 Sub-Category: Cover Crops

A cover crop is often planted during fallow periods between main crops, which is usually over winter in New York State. Typically, a cover crop is not harvested for productive use. If harvested for use, it is usually called a "double crop." However, many of the benefits provided by cover crops also occur for double crops, with the added benefit of a double crop providing an additional product.

An estimate of current GHG emissions is not provided for this sub-category because reliable recent data have not been obtained on the current area on which cover crops have been planted in the State. For reporting GHG mitigation potential associated with cover crops, the estimate from Fargione et al. (2018) was used. They estimated the flux magnitude for this category at the national scale and subsequently dis-aggregated this national estimate to the state level.[3]

3.2.3.5 Sub-Category: Reduced Tillage

Reduced tillage, including no-till practices and many types of conservation tillage, has many benefits and is attractive to New York State farmers who are increasingly adopting the method. There is the potential to increase soil C stocks with reduced tillage, and this can reduce GHG emissions. However, there can also be increased emissions of N_2O which might offset such reductions for many years. Furthermore, the degree to which total soil C stocks are increased with this practice is unclear because most measurements have been made only in the surface soil horizon. Lastly, any gains in soil organic C with reduced tillage can be lost quickly if tillage is increased subsequently. Therefore, mitigation potential for this practice was not estimated.

3.2.3.6 Sub-Category: Drained Wetlands

In New York State, there are about 12,000 ha of drained peatlands, also called mucklands, which most likely belong to the Histosol soil order (NYCRS 2008). They are highly productive, with very high amounts of soil organic matter. However, because the wetlands have been drained from their naturally saturated state, they can have high emissions of CO_2 due to decomposition of organic matter. The estimate of GHG emissions is for CO_2 only and is derived using methods from IPCC (2006) as implemented by the EPA (2016a). A potential future improvement would be to add estimates for CH_4 and N_2O.

It would be possible to mitigate these drained wetland emissions by allowing them to revert to their natural undrained condition, which would reduce the area available for crop production. For that reason, an estimate for mitigation potential was not included.

3.2.3.7 Sub-Category: Replace Annual with Perennial Crops

Replacing annual crops with perennial crops has many potential benefits for soil health and can increase C storage in agricultural soils. The land area projected to become available due to ongoing increases in the yields of major crops and in increased dairy production efficiency was previously estimated by Wightman et al. (2015b). These lands could be used for another purpose while maintaining total agricultural production. The annual increase in soil C storage that would occur if these lands were converted to perennial crops was also estimated. Note that the Natural Climate Solutions project (Fargione et al. 2018) estimates a pathway of Grassland Restoration which overlaps with this annual-to-perennial conversion pathway.

3.2.3.8 Sub-Category: Equipment (Fuel)

This sub-category includes fossil fuel combustion GHG emissions from farm equipment (Wightman et al. 2015a). This emission sub-category is “possibly out of sector.”

3.2.3.9 Sub-Category: Equipment (Embodied)

These are GHG emissions from “upstream” processes for creating farm equipment, such as steel manufacturing and equipment manufacturing. (Wightman et al. 2015a). This emission sub-category is “possibly out of sector.”

3.2.3.10 Sub-Category: Production of Herbicide, Phosphorus, Potassium, Seed

These are GHG emissions associated with the production of agricultural chemicals and phosphorus (P), and potassium (K) fertilizers, as well as production of seed (Wightman et al. 2015a). This emission sub-category is “possibly out of sector.”

3.2.3.11 Sub-Category: Production of Lime

These are GHG emissions associated with the production of lime products for agricultural use (Wightman et al. 2015a). This emission sub-category is “possibly out of sector.”

3.2.3.12 Sub-Category: Production of Synthetic N

These are GHG emissions associated with the production of N fertilizer for agricultural use (Wightman et al. 2015a). This emission sub-category is “possibly out of sector.”

3.2.4 Category: Reduce Food Waste

In the United States, 31% of food is estimated to be lost at the retail or consumer level based on analysis of 2010 data (Buzby et al. 2014). Thus, reducing food loss and waste could greatly reduce GHG emissions from the agricultural sector per unit product used. The United States Department of Agriculture (USDA) and EPA set a goal to reduce food loss and waste by half by the year 2030.[4] The New York State Department of Environmental Conservation (DEC) also promotes reduction in food waste (DEC 2010).

For GHG mitigation via reducing food waste, the reduction in emissions from the agricultural sector only (not landfill emissions, etc.) was calculated here. This was done by multiplying the reduction goal (50%), the estimate of food waste (31%), and the total estimated agricultural GHG emissions. The calculation was intended only as a preliminary or placeholder estimate in order to bring attention to the importance of the topic for GHG mitigation in the State. Reduced food waste is “possibly out of sector” but has a direct impact on the amount of agricultural land required and associated emissions that result from producing food.

3.2.5 Category: Farm Energy Conservation

The category includes all types of energy conservation measures undertaken on farms and is likely already accounted for outside the agricultural sector. As a result, an estimate is not provided. However, the category has been listed in this report because it could be an important GHG mitigation category associated with agriculture, with many additional benefits, such as reduced air pollution and reduced farm overhead.

3.2.6 Category: Agricultural Land Wind and Solar Energy

The category includes wind and solar energy produced on farms, which can displace fossil fuel GHG emissions. Estimates are not provided for this category because they are likely already accounted for outside the agricultural sector. Wind and solar energy produced on farms are included in this report because they could be an important GHG mitigation category associated with agriculture and could have additional co-benefits such as reduced air pollution and possible reduced energy costs and/or income generation.

3.2.7 Category: Avoided Grassland Conversion

The category reflects an avoidance of the conversion of grassland and shrubland to tilled cropland. Such conversion greatly reduces soil C stocks and causes substantial emission of CO_2 to the atmosphere over the first few years of tillage (Woodbury et al. 2007).

Fargione et al. (2018) estimated the category at the national scale, and subsequently dis-aggregated the national estimate to the state level. Estimates of future emissions were made by analyzing recent historical rates of conversion and projecting that these rates would continue to the year 2030 (Lark et al. 2015). However, through the analysis of more recent data on rates of conversion in New York State, it was found that the tilled agricultural area decreased by 775 ha^{-1} yr^{-1} from 2012 to 2017 (USDA-NASS 2019). Therefore, this category is not a likely source of emissions in New York State currently or in the near future.

3.2.8 Category: Forested Riparian Buffer

Riparian buffers are strips of trees and other vegetation, often planted in agricultural land adjacent to streams, rivers, and estuaries. The main motivation is to improve water quality by reducing movement of N, P, fecal bacteria, and sediment from tilled fields into surface waters and by shading the stream. Multiple programs in the State are promoting and implementing riparian buffers (Conley et al. 2018).

Data on the area of State forested riparian buffers were used, along with estimates of C sequestration by growing vegetation in the buffer strips, to estimate the GHG sink benefits of riparian buffers. The agricultural area in New York State that was enrolled in federal buffer installation programs in 2010 was 5,460 ha (Pape et al. 2016). The figure was multiplied by the average C sequestration rate of maple-beech-birch forest stands in the Northeast from 0 to 50 years old (Smith et al. 2006). Maple-beech-birch is a common forest type near agricultural lands in this region.

Additional mitigation values for the category were not estimated because of the high-financial cost of installing such buffers and the removal of land from agricultural production. It is possible that the area of these buffers could increase in the future to help achieve multiple environmental protection and enhancement goals.

3.2.9 Category: Alley Cropping

Alley cropping entails planting rows of trees at wide spacing with a companion crop grown in the alleyways between the rows. Fargione et al. (2018) estimated the C balance for this category at the national scale, and subsequently dis-aggregated the national estimate to the state level.[5] The published estimate for New York State was used.

3.2.10 Category: Bioenergy

Bioenergy is a form of renewable energy that is derived from recently living organic materials known as biomass. Bioenergy can be used to produce transportation fuels, heat, electricity, and wood or residue products. The potential to produce bioenergy feedstocks and bioenergy throughout New York State has been described previously (NYSERDA 2010, Woodbury et al. 2010, Wightman et al. 2015b, Wightman et al. 2015a). Specifically, yield estimates of purpose-grown bioenergy feedstocks in dry tons, along with potential area for producing them, are presented in Tables 2 and 3 of Wightman et al. (2015b). Estimates of GHG emissions for production of these feedstocks is presented in Wightman et al. (2015a). The category is included here because it can offer a large GHG mitigation opportunity. However, a quantitative estimate is not provided because it may be considered outside the sector, and because it competes for land with other categories, such as Reforestation described below.

3.2.11 Category: Reforestation of Former Agricultural Land

The category represents human-induced conversion of non-forest land to forest through planting, seeding, or human promotion of natural seed sources. Approximately 707,863 ha of former agricultural land in New York State currently exists as herbaceous or shrub-scrub cover that is not in agricultural use (Wightman et al. 2015b). These lands could be reforested, which would increase C sequestration and provide other benefits. This potential area of conversion was multiplied by the average C sequestration rate of maple-beech-birch forest stands in the Northeast that range from 1 to 50 years of age (Smith et al. 2006).

For estimating mitigation cost, results of Fargione et al. (2018) for the State were used here, but were adjusted proportionally to reflect the reduced estimate of total mitigation potential described above. Thus, the estimated total mitigation potential reported here is lower than the result from the national analysis of Fargione et al. (2018) downscaled to New York State.

3.3 Wetlands

Data on GHG fluxes from wetlands were taken and down-scaled mainly from Appendix 13B of the SOCCR2 report (Trettin et al. 2018). The data estimated atmospheric CO_2 -uptake by wetlands in the United States as well as the net fluxes of CH_4 from wetlands to the atmosphere, which are opposing processes in terms of source/sink dynamics. The main source documents used by Trettin et al. (2018) included SOCCR1 CCSP (2007) and IPCC (2013). More recent measurements were also considered (see references in Trettin et al. [2018]), especially for CH_4. There were fewer new studies of CO_2 flux. Area C flux density factors (g C/m^2 per year) were used to estimate ecosystem exchanges and fluxes for freshwater wetlands.

3.3.1 Terrestrial Wetlands

Carbon fluxes for terrestrial wetlands in New York State were based on spatial data developed by the NWI (National Wetland Inventory; U.S. Fish and Wildlife Service 2019) representing palustrine wetlands. Total flux estimates were based on net C flux densities for CO_2 and CH_4 associated with these wetland types (Trettin et al. 2018). Palustrine wetlands (NWI Code = P) are defined as follows:

> …all nontidal wetlands dominated by trees, shrubs, persistent emergents, emergent mosses or lichens, and all such wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 ‰. It also includes wetlands lacking such vegetation, but with all of the following four characteristics: (1) area less than 8 ha (20 acres); (2) active wave-formed or bedrock shoreline features lacking; (3) water depth in the deepest part of basin less than 2 m at low water; and (4) salinity due to ocean-derived salts less than 0.5 ‰. (U.S. Fish and Wildlife Service 2019)

The vast majority (93%) of palustrine wetlands in the State are dominated by trees, shrubs, persistent emergents, or emergent mosses/lichens. Palustrine wetlands dominated by trees were considered forested wetlands and all other palustrine wetland types were considered non-forested wetlands. Net C flux densities for CO_2 and CH_4 were assigned separately for forested and non-forested wetlands based on literature values (Harriss et al. 1982, Wilson et al. 1989, Yavitt et al. 1993, Yavitt et al. 1997, Miller et al. 1999, Werner et al. 2003, Bonneville et al. 2008, Chu et al. 2015, Strachan et al. 2015). It was assumed that these literature values from studies located in and around New York State were generally representative of the palustrine wetlands found in the State. The average and standard error among literature values were used in conjunction with the total forested/non-forested palustrine wetland area to generate average estimates and associated quantitative uncertainty of the net CO_2e flux (MMt CO_2e per year) from all forested and non-forested palustrine wetlands. Literature values for wetland CO_2 and CH_4 fluxes used in this analysis are provided in Table 7.

The NWI codes used to define wetland classes are presented in Table 8. General depictions of forested and non-forested palustrine wetlands located in New York State are shown in Figures 1 and 2.

Table 7. Literature Values for CO_2 and CH_4 Fluxes to and from Wetlands and Associated Standard Error

Wetland Type	Flux (CO_2-C per m^2)	Location	Citation
CO_2			
Terrestrial–Forested Palustrine	-67	Various	Trettin et al. (2018)
Terrestrial–Non-forested Palustrine	-264	Ottawa, Ontario, Canada	Bonneville et al. (2008)
	65.4	Ohio	Chu et al. (2015)
	-223.8	Ottawa, Ontario, Canada	Strachan et al. (2015)
Estuarine–Intertidal	-255.6	Massachusetts	Forbrich and Giblin (2015)
	-336	Massachusetts	Forbrich and Giblin (2015)
	-279.6	Massachusetts	Forbrich and Giblin (2015)
	-160	Massachusetts	Moseman-Valtierra et al. (2016)
	984	New Jersey	Schäfer et al. (2014)
	-64.8	New Jersey	Schäfer et al. (2014)
	-309.6	New Jersey	Schäfer et al. (2014)
	-213.6	New Jersey	Artigas et al. (2015)
	-256.8	New Jersey	Weston et al. (2014)
	61.2	New Jersey	Weston et al. (2014)
	93.6	New Jersey	Weston et al. (2014)
	-45.6	New Jersey	Weston et al. (2014)
	-115.2	New Jersey	Weston et al. (2014)
	-171.6	New Jersey	Weston et al. (2014)

Table 7 continued

Wetland Type	Flux (CO_2-C per m^2)	Location	Citation
CH_4			
Terrestrial–Forested Palustrine	0.375	Virginia	Harriss et al. (1982)
	51.75	New York	Miller et al. (1999)
	9.3	Wisconsin	Werner et al. (2003)
	31.95	Virginia	Wilson et al. (1989)
	41.475	Virginia	Wilson et al. (1989)
Terrestrial–Non-forested Palustrine	93.975	New York	Yavitt et al. (1997)
	13.331	New York	Yavitt et al. (1997)
	41.906	New York	Yavitt et al. (1997)
	10.688	New York	Yavitt et al. (1993)
	8.438	New York	Yavitt et al. (1993)
	0.9	New York	Yavitt et al. (1993)
Estuarine–Intertidal	4.3	New Jersey	Reid et al. (2013)
	3.8	New Jersey	Reid et al. (2013)

Table 8. NWI Codes used to define the various wetland types

Wetland Type	NWI Codes
Terrestrial–Forested	PFO
Terrestrial–Non-forested	P minus PFO (i.e. all Palustrine wetlands except for forested)
Estuarine–Intertidal	E2

Figure 1. General Depiction of Forested Wetlands in New York State

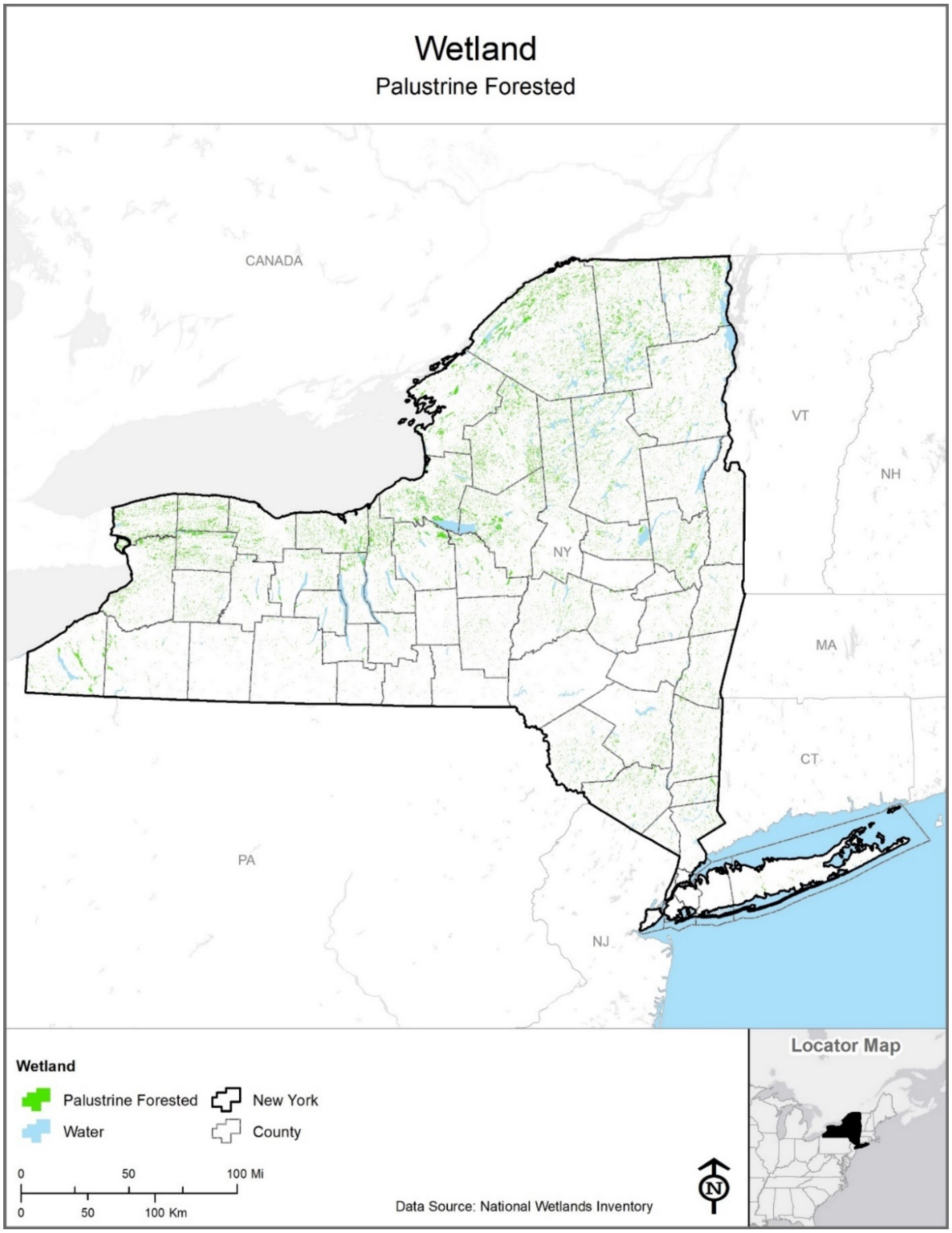
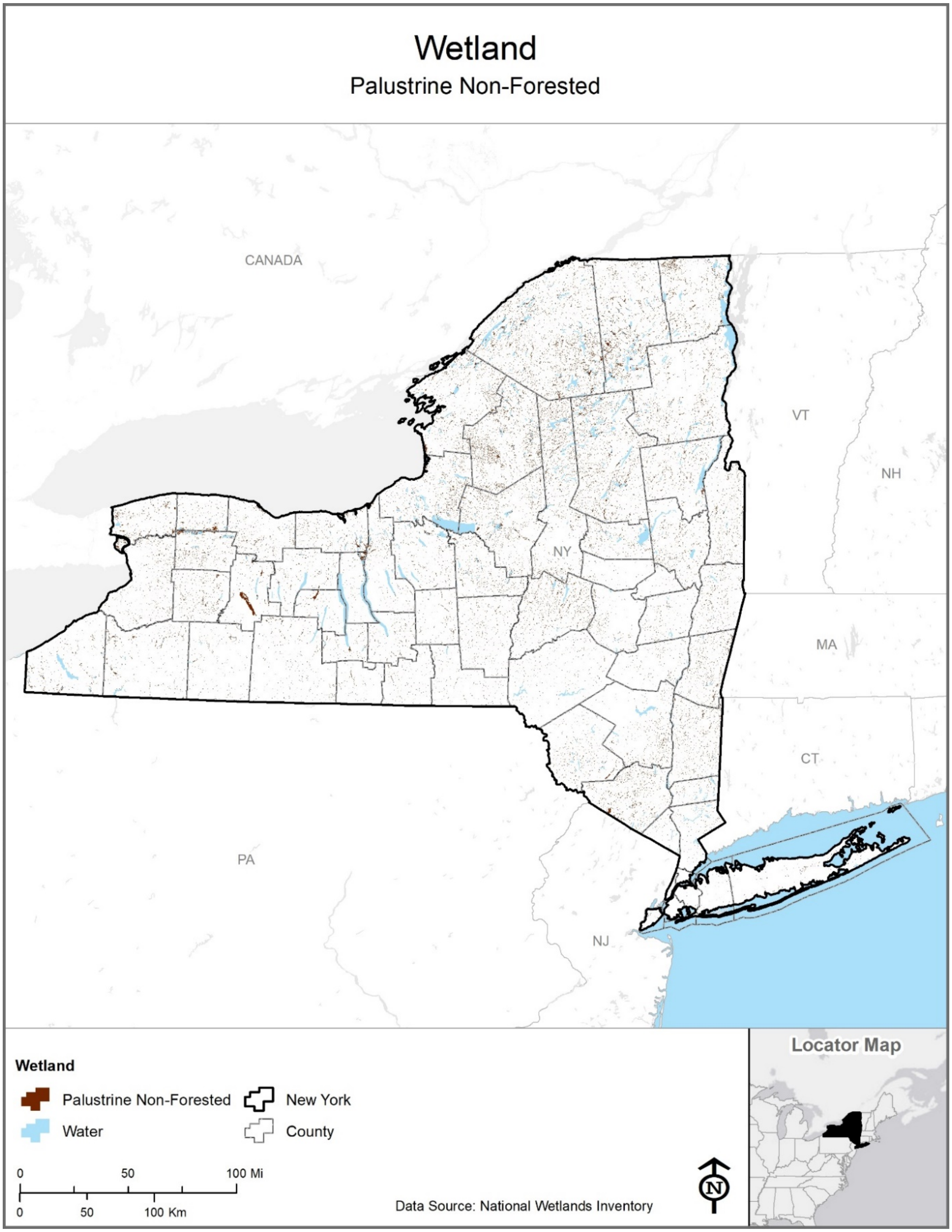


Figure 2. General Depiction of Non-forested Wetlands in New York State



3.3.2 Estuarine Wetlands

Carbon fluxes for estuarine wetlands of New York State were based on spatial data developed by the National Wetlands Inventory (NWI; U.S. Fish and Wildlife Service 2019) representing intertidal wetlands. Total flux estimates were based on net C flux densities for CO_2 and CH_4 associated with these wetland types (Windham-Myers et al. 2018). Estuarine wetlands (NWI Code = E) are defined as follows:

> …deepwater tidal habitats and adjacent tidal wetlands that are usually semi-enclosed by land but have open, partly obstructed, or sporadic access to the open ocean, and in which ocean water is at least occasionally diluted by freshwater runoff from the land. The salinity may be periodically increased above that of the open ocean by evaporation. Along some low-energy coastlines there is appreciable dilution of sea water. Offshore areas with typical estuarine plants and animals, such as red mangroves (*Rhizophora mangle*) and eastern oysters (*Crassostrea virginica*), are also included in the Estuarine System. (U.S. Fish and Wildlife Service 2019)

Intertidal wetlands (NWI Code = E2) in the NWI comprise a subset of estuarine wetlands. The substrate in these habitats is flooded and exposed by tides and the delineated area includes the associated splash zone (U.S. Fish and Wildlife Service 2019). Intertidal wetlands in New York State are often termed salt marshes."

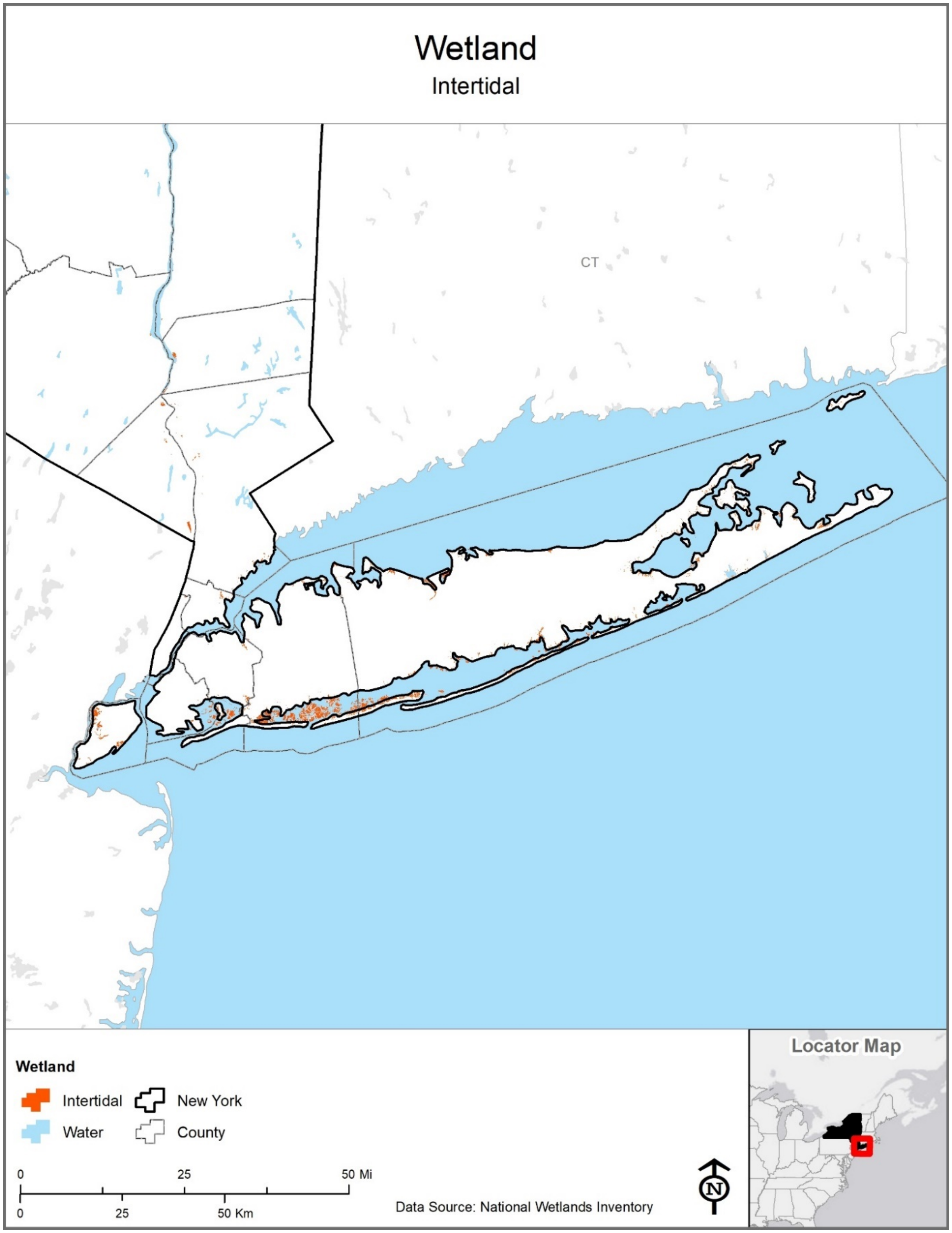
Net C flux densities for CO_2 and CH_4 were assigned to intertidal wetlands based on literature values (Reid et al. 2013, Schäfer et al. 2014, Weston et al. 2014, Artigas et al. 2015, Forbrich and Giblin 2015, Moseman-Valtierra et al. 2016). It was assumed that these literature values from studies located in and around New York State were generally representative of the intertidal wetlands found in the State. The average and standard error among literature values were used in conjunction with the total intertidal wetland area to generate average estimates and associated uncertainty of the net CO2e flux (MMt CO2e per year) from all intertidal wetlands (Table 9). A general depiction of the locations of intertidal wetlands in New York State is shown in Figure 3.

Table 9. Net annual CO_2 Equivalent Flux (MMt CO_2e yr^{-1}) for Wetland Types Found in New York State

Values for the sum of CO_2 and CH_4 are included along with CO_2 and CH_4 shown separately.

		Greenhouse Gas Flux (MMt CO_2e yr^{-1})		
Gas Type	**Wetland Type**	**Average**	**Average Plus Standard Error**	**Average Minus Standard Error**
CO_2 + CH_4	Terrestrial–Forested	3.26	5.61	0.90
	Terrestrial–Non-forested	1.55	4.69	-1.59
	Intertidal	-0.02	0.03	-0.07
CO_2	Terrestrial–Forested	-1.22	-0.47	-1.98
	Terrestrial–Non-forested	-1.89	-0.50	-3.28
	Intertidal	-0.04	0.01	-0.08
CH_4	Terrestrial–Forested	4.48	6.08	2.87
	Terrestrial–Non-forested	3.43	5.18	1.69
	Intertidal	0.02	0.02	0.02

Figure 3. General Depiction of Intertidal Wetlands in New York State



3.4 Leakage

One important issue for GHG accounting and analysis of mitigation opportunities is "leakage," which refers to an emissions reduction strategy implemented in one location that creates an increase or decrease of emissions in another location. If the other location is outside the boundary of the region being analyzed, such as New York State, it can greatly affect the interpretation of the efficacy of a GHG mitigation practice. For example, a lumber company in the State may place 1,000 acres of forest under a permanent conservation easement preventing any harvest in order to sequester C. However, to meet the market demand for lumber, another company may deforest 1,000 acres outside New York State. In this example, leakage occurs due to market forces, but it can occur by many mechanisms, including management or mitigation policies. The effect is that GHG emissions from a local farm, sector, or State are in effect transferred to another location. Analyzing across all locations, leakage can cause either increased total GHG gas emissions (negative leakage) or decreased total GHG emissions (positive leakage). Negative leakage can occur across sectors, for example, if reforesting New York State agricultural lands to sequester C results in an increase of imported food grown on land in Pennsylvania, causing an out-of-State increase of agricultural GHG emissions. An example of positive leakage would be implementation of policies that cause New York State farms to use less synthetic N fertilizer (manufactured in Ohio), thus reducing the fossil fuels and associated GHG emissions used in Ohio to make that synthetic N fertilizer. For this reason, the focus is primarily on mitigation categories that reduce the potential for negative leakage and to point out when such leakage might occur.

4 Results for New York State

4.1 Forests

Results are presented here for forested systems by category. Estimates of the area of Forest Land Remaining Forest Land in New York State and the associated C pools in that forest land are given in Table 10. There was an estimated 7.4 million ha of this forest land type, with a small decrease over the past several decades. The forest C pool has gradually increased over time from about 1,802 MMt C in 1990 to 1,976 MMt C in 2018. Most (~58%) of the C in this pool was estimated to be in the soil.

Table 10. Forest Area and C Stocks in the Forest Land Remaining Forest Land (MMt C) Category

		Individual Forest Ecosystem Carbon Pool					
Year	Forest Ecosystem C Pool	Above ground Biomass	Below ground Biomass	Dead Wood	Litter	Soil	Forest Area (1000 ha)
1990	1802	428	85	48	99	1142	7527
1991	1808	433	86	48	99	1142	7525
1992	1815	438	87	49	100	1142	7522
1993	1821	442	88	49	100	1143	7520
1994	1828	447	88	49	100	1143	7516
1995	1834	452	89	50	100	1143	7513
1996	1841	456	90	50	101	1144	7510
1997	1847	461	91	50	101	1144	7507
1998	1854	465	92	51	101	1144	7504
1999	1860	470	93	51	101	1145	7500
2000	1866	474	94	51	102	1145	7497
2001	1872	479	95	52	102	1145	7494
2002	1879	483	96	52	102	1145	7490
2003	1885	488	97	52	102	1146	7487
2004	1891	492	98	53	102	1146	7483
2005	1897	497	98	53	103	1146	7480
2006	1903	501	99	53	103	1147	7476
2007	1910	506	100	54	103	1147	7473
2008	1916	510	101	54	103	1147	7469
2009	1922	515	102	55	103	1147	7466
2010	1928	519	103	55	104	1148	7462
2011	1934	523	104	55	104	1148	7459
2012	1940	528	105	56	104	1148	7455
2013	1946	532	105	56	104	1148	7451
2014	1952	537	106	56	104	1148	7448
2015	1958	541	107	57	104	1149	7445
2016	1964	545	108	57	105	1149	7442
2017	1970	550	109	58	105	1149	7439
2018	1976	554	110	58	105	1149	7436

The CO_2e flux from forest pools associated with Forest Land Remaining Forest Land and Land Converted to Forest Land for years 1990–2017 is shown in Table 11. These net fluxes are plotted in Figure 4 and linearly extrapolated to years 2030 and 2050 to provide estimates of the possible future net CO_2e flux associated with forest ecosystems of New York State. Net CO_2e flux estimates for years 1990 to 2017 are presented separately for Forest Land Remaining Forest Land and Land Converted to Forest Land in Tables 12 and 13. Most (~70%) of the estimated C sequestration in Forest Land Remaining Forest Land and Land Converted to Forest Land was allocated to aboveground biomass, followed by underground biomass (~ 14%) and dead wood (~7%). Table 14 provides the net C flux estimates, which were used to derive the net CO_2e flux estimates. Tables 15 and 16 include information related to non-CO_2 emissions associated with forest ecosystems. Estimates of GHG (and related) emissions, other than CO_2, from forest fires in Forest Land Remaining Forest Land and Land Converted to Forest Land are given in Tables 15 and 16 in MMt CO2e and kiloton (kt) respectively. A general depiction of forested areas of New York State is shown in Figure 5.

Table 11. Net CO_2 Flux from Forest Pools in Forest Land Remaining Forest Land and Land Converted to Forest Land (MMt CO_2e) Categories for Years 1990–2017

Minus signs signify that the forest lands are acting as a net C sink.

Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux: Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil	Forest Area (1000 ha)
1990	-28.3	-19.6	-3.9	-1.6	-1.8	-1.3	7660
1991	-28.2	-19.6	-3.9	-1.6	-1.8	-1.3	7658
1992	-28.1	-19.6	-3.9	-1.6	-1.7	-1.3	7655
1993	-28.0	-19.5	-3.9	-1.6	-1.7	-1.3	7653
1994	-27.9	-19.4	-3.9	-1.7	-1.7	-1.2	7649
1995	-27.7	-19.3	-3.8	-1.7	-1.7	-1.2	7646
1996	-27.6	-19.2	-3.8	-1.7	-1.7	-1.2	7643
1997	-27.5	-19.2	-3.8	-1.7	-1.7	-1.1	7640
1998	-27.4	-19.1	-3.8	-1.7	-1.7	-1.1	7636
1999	-27.2	-19.0	-3.8	-1.7	-1.7	-1.1	7633
2000	-27.1	-18.9	-3.8	-1.7	-1.6	-1.1	7630
2001	-27.0	-18.8	-3.7	-1.7	-1.6	-1.1	7626
2002	-26.9	-18.8	-3.7	-1.7	-1.6	-1.0	7623
2003	-26.8	-18.7	-3.7	-1.7	-1.6	-1.0	7619
2004	-26.8	-18.7	-3.7	-1.7	-1.6	-1.0	7616
2005	-26.7	-18.7	-3.7	-1.8	-1.6	-1.0	7612
2006	-26.7	-18.7	-3.7	-1.8	-1.6	-1.0	7609
2007	-26.6	-18.6	-3.7	-1.8	-1.6	-0.9	7605
2008	-26.6	-18.6	-3.7	-1.8	-1.6	-0.9	7602
2009	-26.5	-18.6	-3.7	-1.8	-1.6	-0.9	7598
2010	-26.4	-18.5	-3.7	-1.8	-1.5	-0.8	7594
2011	-26.3	-18.5	-3.7	-1.9	-1.5	-0.8	7591
2012	-26.3	-18.5	-3.7	-1.9	-1.5	-0.7	7587
2013	-26.1	-18.4	-3.7	-1.9	-1.5	-0.7	7583
2014	-26.0	-18.3	-3.6	-1.8	-1.5	-0.7	7580
2015	-25.8	-18.2	-3.6	-1.8	-1.5	-0.7	7577
2016	-25.7	-18.1	-3.6	-1.8	-1.5	-0.7	7574
2017	-25.5	-18.0	-3.6	-1.8	-1.5	-0.7	7571

Figure 4. Forest Land Remaining Forest Land (FRF) and Land Converted to Forest Land (LCF) Net Carbon Stock Change (i.e., CO_2e Flux)

For years 1990–2017 (open circles) and extrapolations to years 2030 and 2050 (solid triangles) based on linear historical trend (line). Total flux (from Table 12) is shown in panel A and the flux per unit area (from Table 17) is shown in panel B.

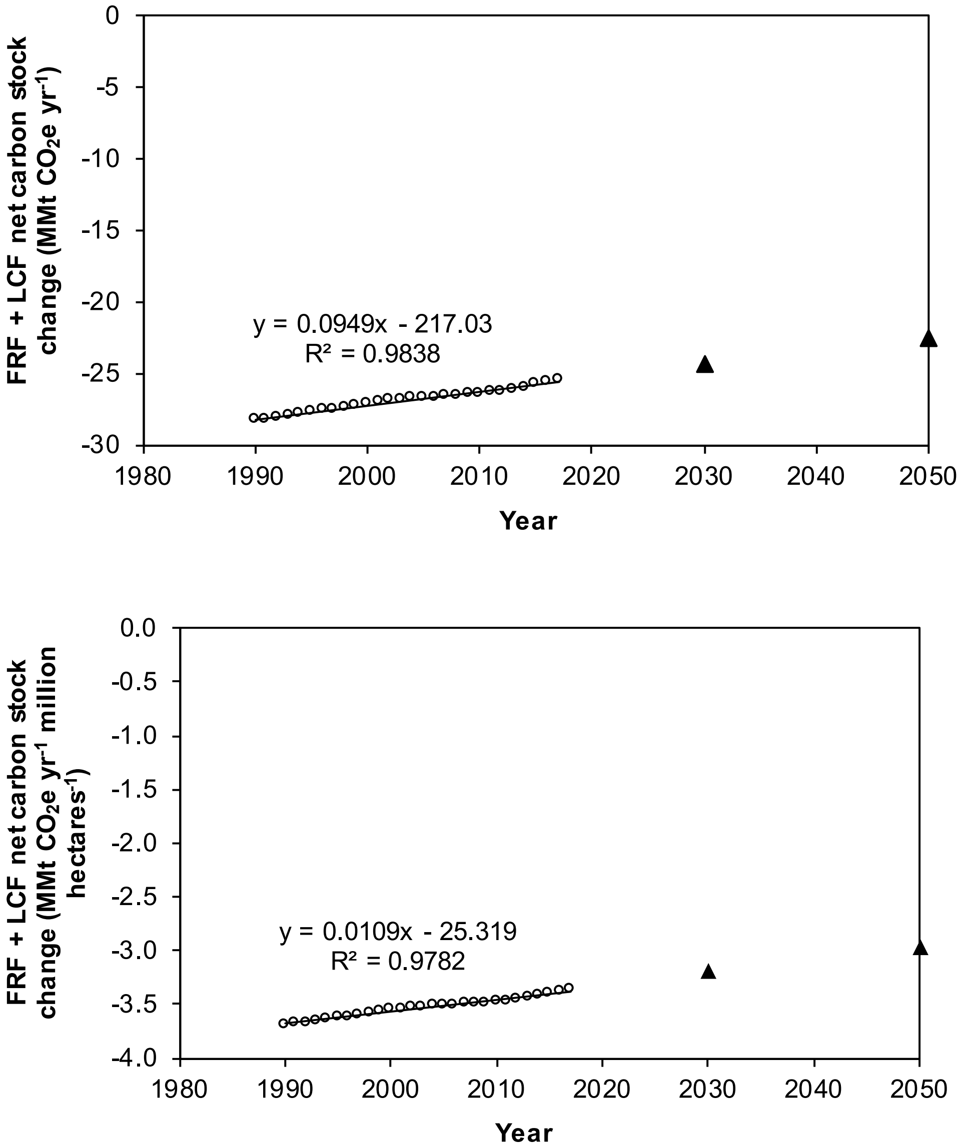


Table 12. Net CO_2 Flux from Forest Pools in the Forest Land Remaining Forest Land (MMt CO_2e) Category for Years 1990–2017

Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux: Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil
1990	-24.1	-17.3	-3.4	-1.1	-0.9	-1.3
1991	-24.0	-17.2	-3.4	-1.2	-0.9	-1.3
1992	-24.0	-17.2	-3.4	-1.2	-0.9	-1.3
1993	-23.8	-17.1	-3.4	-1.2	-0.9	-1.3
1994	-23.7	-17.0	-3.4	-1.2	-0.9	-1.2
1995	-23.6	-17.0	-3.4	-1.2	-0.9	-1.2
1996	-23.5	-16.9	-3.4	-1.2	-0.8	-1.2
1997	-23.3	-16.8	-3.3	-1.2	-0.8	-1.1
1998	-23.2	-16.7	-3.3	-1.2	-0.8	-1.1
1999	-23.1	-16.6	-3.3	-1.2	-0.8	-1.1
2000	-23.0	-16.6	-3.3	-1.3	-0.8	-1.1
2001	-22.9	-16.5	-3.3	-1.3	-0.8	-1.1
2002	-22.7	-16.4	-3.3	-1.3	-0.8	-1.0
2003	-22.7	-16.4	-3.3	-1.3	-0.7	-1.0
2004	-22.6	-16.4	-3.3	-1.3	-0.7	-1.0
2005	-22.6	-16.3	-3.3	-1.3	-0.7	-1.0
2006	-22.6	-16.3	-3.2	-1.3	-0.7	-1.0
2007	-22.5	-16.3	-3.2	-1.3	-0.7	-0.9
2008	-22.4	-16.2	-3.2	-1.4	-0.7	-0.9
2009	-22.4	-16.2	-3.2	-1.4	-0.7	-0.9
2010	-22.3	-16.2	-3.2	-1.4	-0.7	-0.8
2011	-22.2	-16.1	-3.2	-1.4	-0.7	-0.8
2012	-22.2	-16.1	-3.2	-1.4	-0.7	-0.7
2013	-22.0	-16.0	-3.2	-1.4	-0.7	-0.7
2014	-21.8	-15.9	-3.2	-1.4	-0.7	-0.7
2015	-21.7	-15.8	-3.2	-1.4	-0.7	-0.7
2016	-21.5	-15.7	-3.1	-1.4	-0.6	-0.7
2017	-21.4	-15.6	-3.1	-1.3	-0.6	-0.7

Table 13. Net CO_2 Flux from Forest Pools (MMt CO_2e) in Land Converted to Forest Land by Land Use Change Category for Years 1990–2017

Land Use Conversion	Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux: Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil[a]
Cropland Converted to Forest Land	1990	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1991	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1992	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1993	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1994	-1.1	-0.6	-0.1	-0.1	-0.3	ND

Table 13. Continued

Land Use Conversion	Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux: Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil[a]
	1995	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1996	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1997	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1998	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	1999	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	2000	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	2001	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	2002	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	2003	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	2004	-1.1	-0.6	-0.1	-0.1	-0.3	ND
	2005	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2006	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2007	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2008	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2009	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2010	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2011	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2012	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2013	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2014	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2015	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2016	-1.1	-0.5	-0.1	-0.1	-0.3	ND
	2017	-1.1	-0.5	-0.1	-0.1	-0.3	ND
Other Land Converted to Forest Land	1990	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1991	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1992	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1993	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1994	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1995	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1996	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1997	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1998	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	1999	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2000	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2001	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2002	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2003	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2004	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2005	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2006	-1.9	-1.1	-0.2	-0.2	-0.3	ND

Table 13. Continued

Land Use Conversion	Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux: Above ground Biomass	Belowground Biomass	Dead Wood	Litter	Soil[a]
	2007	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2008	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2009	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2010	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2011	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2012	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2013	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2014	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2015	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2016	-1.9	-1.1	-0.2	-0.2	-0.3	ND
	2017	-1.9	-1.1	-0.2	-0.2	-0.3	ND
Settlements Converted to Forest Land	1990	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1991	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1992	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1993	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1994	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1995	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1996	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1997	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1998	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	1999	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2000	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2001	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2002	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2003	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2004	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2005	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2006	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2007	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2008	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2009	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2010	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2011	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2012	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2013	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2014	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2015	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2016	-1.2	-0.7	-0.1	-0.1	-0.2	ND
	2017	-1.2	-0.7	-0.1	-0.1	-0.2	ND

[a] ND = No data

Table 14. Breakdown of Net C Flux from Stock Changes (MMt C) in Land Converted to Forest Land by Land Use Change Category for Years 1990–2017

Land Use Conversion	Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux				
			Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil[a]
Cropland Converted to Forest Land	1990	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1991	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1992	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1993	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1994	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1995	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1996	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1997	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1998	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	1999	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2000	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2001	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2002	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2003	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2004	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2005	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2006	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2007	**-0.29**	-0.15	-0.03	-0.04	-0.08	ND
	2008	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2009	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2010	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2011	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2012	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2013	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2014	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2015	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2016	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
	2017	**-0.29**	-0.15	-0.03	-0.04	-0.07	ND
Other Land Converted to Forest Land	1990	**-0.51**	-0.31	-0.06	-0.05	-0.09	ND
	1991	**-0.51**	-0.31	-0.06	-0.05	-0.09	ND
	1992	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1993	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1994	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1995	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1996	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1997	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1998	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	1999	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2000	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND

Table 14. Continued

Land Use Conversion	Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux: Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil[a]
	2001	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2002	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2003	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2004	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2005	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2006	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2007	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2008	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2009	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2010	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2011	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2012	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2013	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2014	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2015	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2016	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
	2017	**-0.51**	-0.30	-0.06	-0.05	-0.09	ND
Settlements Converted to Forest Land	1990	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1991	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1992	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1993	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1994	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1995	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1996	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1997	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1998	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	1999	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	2000	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	2001	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	2002	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	2003	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	2004	**-0.33**	-0.19	-0.04	-0.03	-0.07	ND
	2005	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2006	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2007	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2008	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2009	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2010	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2011	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND

Table 14. Continued

Land Use Conversion	Year	Total Forest Ecosystem Flux	Individual Forest Ecosystem Flux				
			Aboveground Biomass	Belowground Biomass	Dead Wood	Litter	Soil[a]
	2012	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2013	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2014	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2015	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2016	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND
	2017	**-0.33**	-0.19	-0.04	-0.03	-0.06	ND

[a] ND = No data

Table 15. Estimated Non-CO_2 Emissions from Forest Fires (MMt CO_2e)[a]

Gas	2006	2008	2012	2015	2016	2017[b]
CH_4	0.001	0.005	0.003	0.007	0.003	0.003
N_2O	0.001	0.004	0.002	0.005	0.002	0.002
Total						

[a] These estimates include Non-CO_2 Emissions from Forest Fires on Forest Land Remaining Forest Land and Land Converted to Forest Land.

[b] The fire data for 2017 were unavailable when these estimates were developed, therefore 2016, the most recent available estimate, is applied to 2017.

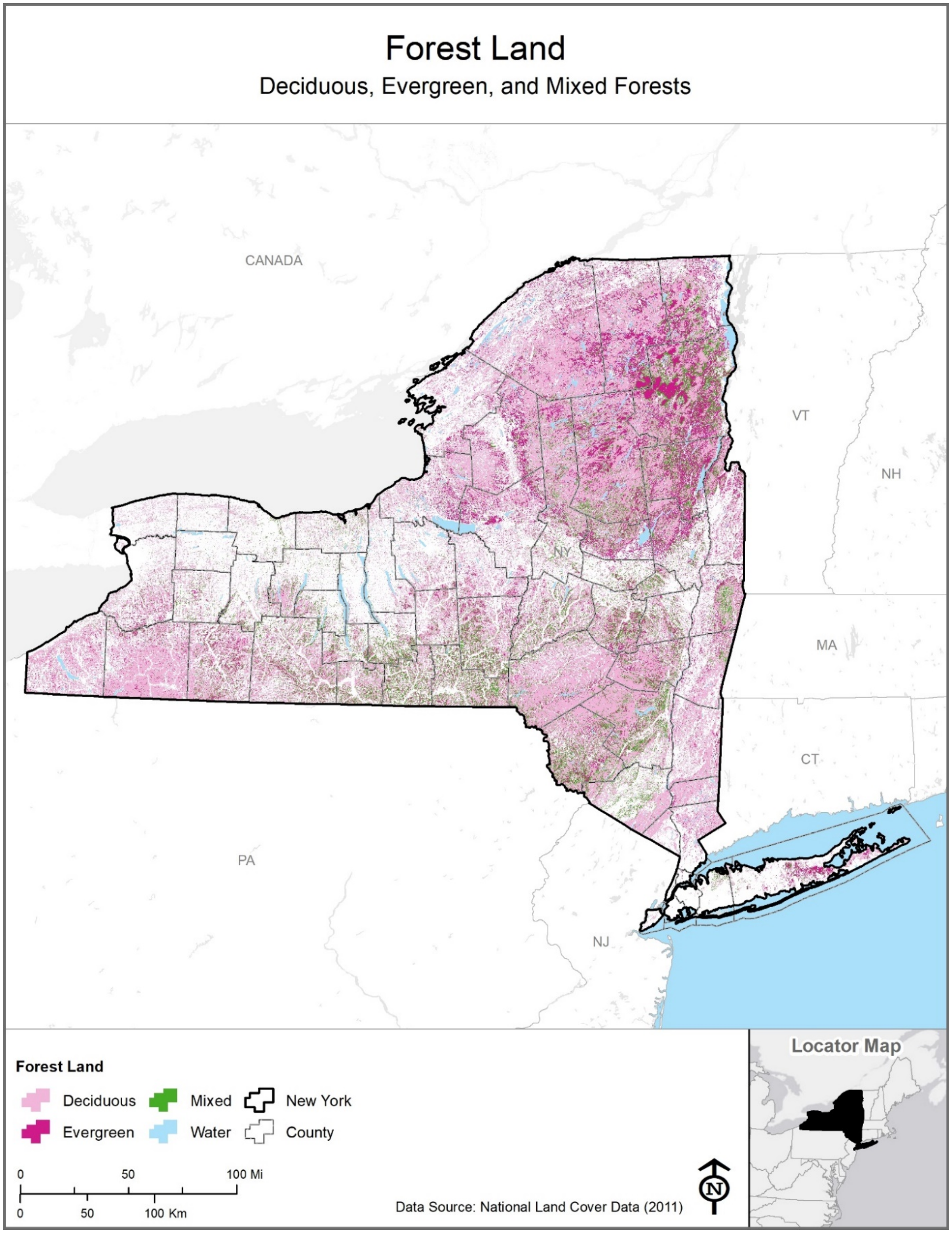
Table 16. Estimated Non-CO_2 Emissions from Forest Fires (kt)[a]

Gas	2006	2008	2012	2015	2016	2017[b]
CH_4	0.03	0.22	0.13	0.29	0.11	0.11
N_2O	0.00	0.01	0.01	0.02	0.01	0.01
CO	0.73	4.93	2.99	6.57	2.52	2.52
NO_x	0.02	0.14	0.08	0.18	0.07	0.07

[a] These estimates include Non-CO_2 Emissions from Forest Fires on Forest Land Remaining Forest Land and Land Converted to Forest Land.

[b] The fire data for 2017 were unavailable when these estimates were developed. Therefore, 2016, the most recent available estimate, is applied to 2017.

Figure 5. General Depiction of Forest Land in New York State



4.2 Agriculture

Net greenhouse gas emissions and mitigation potential for agricultural land are summarized in Table 6. Note that a few categories and sub-categories have a question mark in the table in place of a numerical value. This was done to allow the category to be included even though a numerical estimate could not be provided at this time or to indicate the category is outside the sector. Brief comments are also provided to highlight the most important caveat or provide interpretation of the value provided.

Below, the three agricultural categories with the highest GHG mitigation opportunity identified in Table 6 are discussed, followed by a brief discussion of most other categories and sub-categories following the order shown in Table 6. As discussed in Methods (section 3) and indicated in Table 6, some categories or sub-categories are identified as being "possibly outside the agriculture sector" (see section 3.2). A general depiction of agricultural areas of New York State is shown in Figures 6 and 7.

4.2.1 Category: Reforestation of Former Agricultural Land

The largest mitigation potential could be achieved by growing forest on former agricultural land that is not currently in commercial agricultural production (Table 6). This potential is large (4.9 MMt CO_2e/yr) because of (1) the large amount of land identified as being potentially available for reforestation and (2) the large potential for C-sequestration that could be stored in rapidly growing forests. However, there is substantial uncertainty about the amount of land that could realistically be reforested. The estimate herein is focused on lands that are not currently forested and that are not in active commercial agriculture, based on published geospatial analysis (Wightman et al. 2015b). Specifically, an estimated 707,863 ha of former agricultural land is currently in herbaceous (352,480 ha) or shrub-scrub cover (355,383 ha). The estimate in Table 6 makes a critically important assumption that all these lands are available for reforestation for a total estimated GHG mitigation of 4.9 Tg CO_2e/yr1. This estimate is provided as a likely maximum potential of mitigation of this land area if managed as forest (as compared to these same lands being used to grow bioenergy crops; see below). An alternative lower reforestation mitigation value of 3.81 Tg CO_2e/yr based on an estimate that only 55% of lands in herbaceous cover might be available due to landowner preferences is presented (Wightman et al. 2015b). This estimate of landowner preferences is based on expert forester prediction of landowner attitudes toward forest management, as well as road density in different regions of the State. Thus, it is only an approximation of the fraction of landowners who might be interested in reforestation. More information on landowner attitudes toward, and interest in, reforestation of former agricultural lands would be valuable. There is also uncertainty about the cost and feasibility of reforesting large amounts of land to mixed species native forests. Costs were estimated based on published average values for the United States (Fargione et al. 2018) and may not fully capture the costs for former agricultural lands in New York State. The feasibility of reforestation depends on the ability to establish native species, including obtaining plants, planting, and managing weeds, diseases, pests and herbivores, particularly deer. A statewide study based on FIA data found that forest regeneration was inadequate in 32% of forest plots for all canopy species and 57% of plots for timber species (Shirer and Zimmerman 2010). In another study at 12 New York State forested sites, high deer herbivory (70-90%) was found for unprotected oak seedlings versus none in fenced areas (Blossey et al. 2017). Thus, any reforestation effort will require considerable investment including adequate fencing to exclude deer. The estimate of forest C sequestration used in the results reported here was based on average growth rates of maple-beech-birch stands in the northeastern United States (6.92 Mg CO_2e/ha/yr', derived from Smith et al. [2006 #248]). If growth rates on former agricultural lands in New York State are limited by establishment costs, herbivory, competition from other vegetation including invasive species, the need to compensate landowners, or other factors, it may be costly to achieve the mitigation potential estimated herein. Additionally, there is the potential to plant faster growing species of trees, but the results reported here focused on the example of maple-beech-birch as a common forest type in New York State.

Figure 6. General Depiction of Agricultural Land in Cultivated Crops in New York State

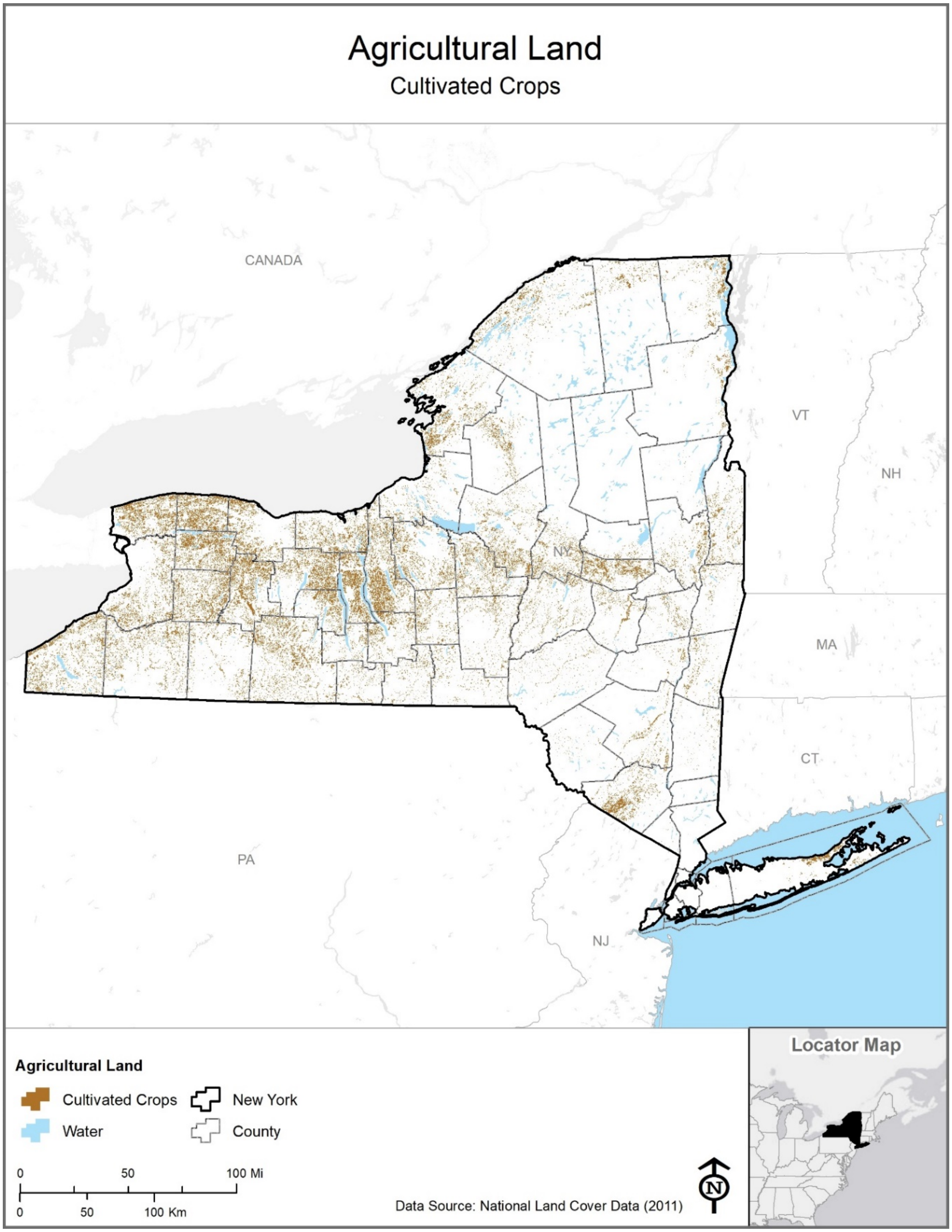
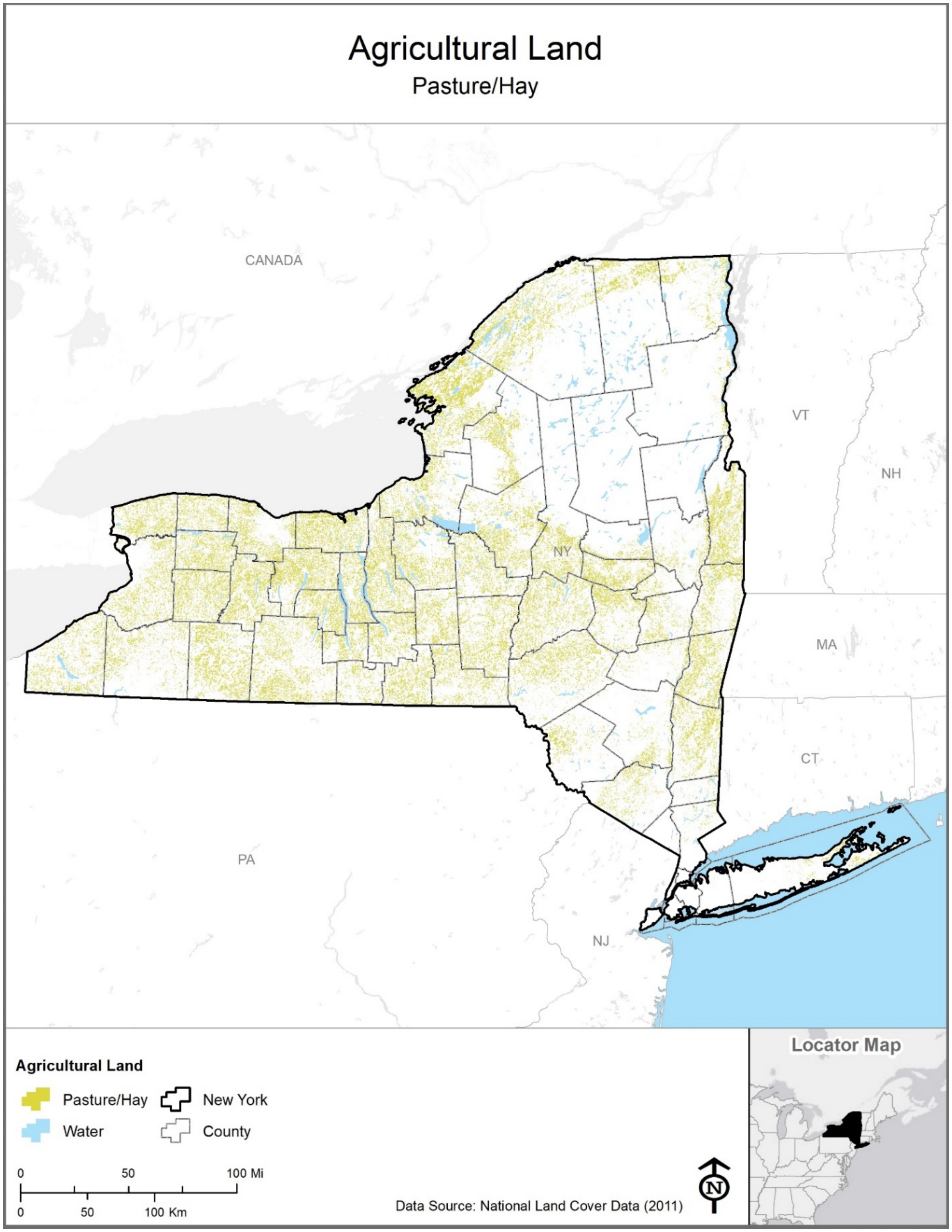


Figure 7. General Depiction of Agricultural Land in Pasture and Hay in New York State



When considering GHG mitigation costs, it is important to note that there are other benefits to reforestation besides climate mitigation, such as biodiversity, improved water quality, flood prevention, soil and nutrient regulation, and many cultural and human health benefits. The following amounts of mitigation are available at the following costs based on U.S. average values from Fargione et al. (2018): $10 per Mg CO_2e for 3% of total mitigation potential, $50 per Mg CO_2e for 82% of mitigation potential, with the remaining potential at greater than $100 per Mg CO_2e. However, additional information on these costs specific to New York State, and to different regions and site conditions of the State, would be valuable.

In addition to reforestation, there are other uses of former agricultural lands that could contribute to GHG mitigation. The potential for bioenergy feedstock production throughout all lands in New York State, including growing grasses such as the perennial switchgrass or short-rotation willow on former agricultural lands has been previously analyzed (see section 3.2.10) and published (Wightman et al. 2015b, 2015a) but is not listed in Table 6 for two reasons. First, analysis of bioenergy GHG benefit is dependent on the conversion processes used. Second, growing bioenergy competes for the same land used in the reforestation value presented, making them mutually exclusive. This last point is significant; the reforestation potential discussed above is presented as the upper limit of mitigation potential for former agricultural lands. However, the proportion of these lands that is actually reforested, used for energy generation (solar, biomass), turned into new housing stock, or left for aesthetics will be a function of markets, policies, and landowner preferences.

4.2.2 Category: Agricultural Soil Management

The second largest category of agricultural mitigation potential is Agricultural Soil Management. This category is defined here using the existing New York State GHG Inventory (NYSERDA 2018) and covers many sub-categories, including fossil fuel use for farm equipment, soil C sequestration, and field N_2O emissions. A total mitigation potential of 1.68 MMt CO_2e/yr was estimated. It includes planting cover crops, replacing some annual crops with perennial crops, and reducing N_2O emissions with improved fertilizer management.

4.2.2.1 Sub-Category: Cover Crops

For cover crops, it is important to note that this term is often used for a crop that is not harvested. However, growing a “double crop” that is harvested can provide the same or greater GHG benefits than a cover crop that is not harvested, as long as it is managed to avoid additional GHG emissions.

This is because the double crop can help increase total production on the same area of land; if it requires few inputs, it can increase the annual production and potentially reduce feed or food imports. It was estimated that cover cropping could provide mitigation of 0.85 MMt CO_2e/yr. This estimate of mitigation potential is taken from a national study (Fargione et al. 2018), which was based on a meta-analysis of data from the literature. There is substantial uncertainty in this potential mitigation for three reasons: (1) potential for increased N_2O emissions, especially with leguminous cover crops; (2) potential for a cover crop to decrease yield of the subsequent crop if not managed correctly; and (3) the uncertainty and impermanence of increasing soil C by means of cover cropping. Additionally, it is possible that some of this potential has already been achieved in New York State and should therefore be counted as a reduction of current emissions rather than a new mitigation potential. When considering GHG mitigation costs for cover crops, it is important to note that there can be other benefits besides climate mitigation, such as improved air and water quality. The following amounts of mitigation are available at the following costs based on U.S. average values from Fargione et al. (2018): \$10 per Mg CO_2e for 0.83 MMt CO_2e per year, \$50 per Mg CO_2e for 0.83 MMt CO_2e per year, and \$100 per Mg CO_2e for 0.85 MMt CO_2e per year. In sum, 96% of this mitigation potential could be achieved for < \$10 per Mg CO_2e.

4.2.2.2 Sub-Category: Replacing Annual with Perennial Crops

Replacing annual crops with perennial crops could mitigate 0.62 MMt CO_2e/yr. This practice will increase the amount of C stored in the soil. This practice is technically feasible, but it may be difficult to find suitable perennial crops to replace annual crops. For example, substantial area in New York State is devoted to production of corn silage as dairy feed, an annual crop. Theoretically, some of this area could be replaced with perennial forage crops. However, corn silage has very high yield and high feed value, and farmers have a great deal of experience in managing it as feed. Perennial replacements may have lower yields or may not provide the same value as livestock feed, limiting the feasibility of this mitigation practice. If yields are lower, it could induce negative leakage by requiring purchase of additional livestock feed from off the farm and outside the State. For example, a portion of the Enteric Fermentation category is from replacing forage with annual crops to improve dietary efficiency for dairy animals, which was found to decrease total farm and life cycle GHG emissions (see Enteric Fermentation sections). This practice could also reduce N_2O emissions because perennial crops are more efficient at taking up N than annual crops, reducing losses of reactive N. This potential benefit was not included in this estimate.

4.2.2.3 Sub-Category: Crop N_2O Emissions (Direct and Indirect)

The third sub-category of mitigation in Agricultural Soil Management is reducing N_2O emission with improved fertilizer management, which could mitigate 0.2 MMt CO_2e/yr. This practice can improve profitability due to reducing fertilizer costs while maintaining yields. As noted in Table 6, a small portion of the estimated mitigation is due to reductions in upstream (synthetic-N manufacturing) GHG emissions. This practice will also avoid loss of reactive N in the form of ammonia which can cause air pollution or leaching of nitrate which can cause water pollution. Thus, while the GHG mitigation total potential is modest, there are many co-benefits and thus implementation of this practice might be a high priority. The following amounts of mitigation are available at the following costs based on U.S. average values from Fargione et al. (2018): $10 per Mg CO_2e for 0.15 MMt CO_2e per year, $50 per Mg CO_2e for 0.17 MMt CO_2e per year, and $100 per Mg CO_2e for 0.20 MMt CO2e per year. In sum, 77% of this mitigation potential could be achieved at <$10 per Mg CO_2e. The following four sub-categories do not contribute to estimated mitigation potential and may be considered "outside the sector" or outside the State: (1) Equipment (embodied), (2) Production of Herbicide, P, K, Seed, (3) Production of Lime, and (4) Production of Synthetic N (included in crop N_2O emission and mitigation potential estimates above).

4.2.3 Category: Manure Management (Storage)

The third largest category of agricultural mitigation opportunity is manure management. This potential is based on covering and flaring methane emissions from manure storage units and is thus permanent. This estimate has low uncertainty for an individual farm compared to many other practices as it is based on well-established methods applied using emissions data specific to New York State dairy farms (Wightman and Woodbury 2016). The number of farms with appropriate storage units for the practice is based on estimates from CAFO permits submitted by large farms. There is uncertainty about the number of smaller farms with manure storage units. As mentioned in Table 6, by year 2030 both emissions and mitigation potential are expected to increase because there will be more farms with liquid manure storage units than at present. However, it is difficult to predict the rate of increase in such units because it is affected by farm size, changes in water quality rules, income and milk prices as well as other farm management challenges. Mitigating GHG emissions from liquid manure storage units is a cost-effective practice estimated at $13 per Mg CO_2e for application in New York State (Wightman and Woodbury 2016) based on a 100-year time scale for the GWP of methane. It also provides other benefits such as reduced risk of runoff under heavy precipitation, reduced odor, and reduced loss of ammonia to the atmosphere. Thus, it can reduce both air pollution and water pollution while improving community relations regarding odor.

4.2.4 Category: Enteric Fermentation

The estimate of emissions from enteric fermentation is based on a modeling study conducted for small and medium sized representative farms in New York State. In addition to reducing total GHG emissions (using a life-cycle approach) the combined "feed mitigation" strategy increased milk production (~11%) and farm profitability (~37%). The increased profitability suggests that this practice could be practical to implement. The increased milk production represents a potential for positive leakage, as milk efficiency could be increased without increasing inputs, provided appropriate markets are available. In sum, this mitigation is a low- to no-cost practice and may even make farms more financially viable.

4.2.5 Category: Reduce Food Waste

As indicated in Table 6, the reduced food waste estimate includes only reductions due to reduced emissions in the agricultural sector. These emissions reductions could occur by reducing production-related emissions in the State as a result of reduced food system losses. Alternatively, these emissions reductions could occur by maintaining production, but reducing food and feed imports to New York State (thus reducing GHG emissions outside the State). In addition to the estimate presented in Table 6, there would be additional GHG reductions outside of the agricultural sector, for example from landfills or from displaced fossil fuel emissions due to energy requirements for production and distribution of food that is subsequently wasted. This is intended only as a preliminary placeholder estimate to bring attention to the importance of the topic for GHG mitigation, and to the interaction between food waste reduction and potential changes in land use in the agricultural sector in New York State and elsewhere.

4.2.6 Category: Alley Cropping

Alley cropping is not common in New York State, and little information on this practice is available. However, it has been judged to be feasible for many regions of the United States as a GHG mitigation practice, and more feasible than other agroforestry practices, and is therefore included herein (cf., Fargione et al. 2018). Even so, implementation would require substantial research, demonstration, and outreach in the State.

When considering GHG mitigation costs, it is important to note that there are other benefits from alley cropping in addition to GHG mitigation, such as improved water quality and reduced erosion. The following amounts of mitigation are available at the following costs based on U.S. average values from

Fargione et al. (2018): \$10 per Mg CO_2e for 0.04 MMt CO_2e per year, \$50 per Mg CO_2e for 0.54 MMt CO_2e per year, and \$100 per Mg CO_2e for 0.67 MMt CO_2e per year. In sum, 80% of this mitigation potential could be achieved at <\$50 per $MgCO_2e$.

4.3 Wetlands

Terrestrial wetlands in the United States generally serve as C sinks for CO_2 but as sources of CH_4 (Kolka et al. 2018). The net CO_2e fluxes estimated for wetland types in New York State are presented in Table 9. Terrestrial wetlands, especially forested palustrine wetlands, were estimated to represent net CO_2e sources. Intertidal wetlands were estimated to represent net CO_2 sinks. All three general wetland types were expected to act as sources of CH_4.

It is not clear whether C fluxes from inland waters to the coast are increasing or decreasing in New York State. This represents a lateral transfer. The largest component of this uncertainty stems from the fact that C export is closely coupled to discharge (Mulholland and Kuenzler 1979). If discharge increases in the State, C export in rivers will probably also increase. Extreme hydrological events will likely contribute disproportionately. Prolonged drought will decrease this C flux.

Enhanced ecosystem respiration during warm periods was a strong determinant of NEE of CO_2 in a temperate freshwater marsh in Ontario, suggesting that climate warming may influence future wetland CO_2 sequestration (Strachan et al. 2015). A tidal marsh in the Hudson-Raritan estuary in New Jersey was found to be a net CO_2 source during winter and a net CO_2 sink during summer (Schäfer et al. 2014). Methane emissions from wetland soils are spatially and temporally variable. Miller et al. (1999) found that temperature and C availability controlled the fluxes of CO_2 and CH_4 more than other environmental variables at a red maple and hemlock swamp in central New York State. Moseman-Valtierra et al. (2016) suggested that more focus on coastal marsh zonation and additional flux measurements across time will be needed to improve the accuracy of C accounting in coastal marshes.

The CH_4 budgets of wetlands are very uncertain (Bridgham et al. 2006, Reid et al. 2013). High productivity and C storage under anoxic conditions cause wetland soils to act as substantial CO_2 sinks, but CH_4 production and release can offset much, or all, of the C sequestration (Reid et al. 2013). The fluxes of CO_2 and CH_4 per unit area of water surface might be relatively high for small streams and ponds (Holgerson and Raymond 2016), which are common in the mountains of New York State such as the Adirondacks and Catskills. Data on CH_4, in particular, are lacking, although some new research has recently been conducted (cf., Holgerson and Raymond 2016, Wik et al. 2016).

4.4 Trends and Future Conditions

4.4.1 Forests

Estimates of annual net C flux from the Forest Land Remaining Forest Land category over the period 1990 to 2017 are provided as CO_{2e} flux in Table 12. Forest Land Remaining Forest Land accounted for most (~85%) of the forest C sequestration (Tables 11 and 12). Net annual CO_2 and C fluxes from other land use categories converted to forest land are given in Tables 13 and 14, respectively. These land conversions have only small effects on the overall amount of forest C sequestration. The CO_2 fluxes per million hectares from Forest Land Remaining Forest Land and Land Converted to Forest Land are given in Table 17 and graphed in Figure 4B. The strength of the C sink, or the rate of net CO_2 removal from the atmosphere by growing vegetation, has decreased slowly but steadily at the statewide level since 1990. Total land areas in the categories Forest Land Remaining Forest Land and Land Converted to Forest Land decreased by 1.2% (89,510 ha) between 1990 and 2017. However, sequestration of CO_2-C per unit area has decreased by 8.7% during this timeframe. There are several potential contributors to this trend, including decreased N fertilization via atmospheric N deposition, climate change, insect infestation, and forest maturation. Anthropogenic N addition, such as atmospheric N deposition, has been shown to increase the CO_2 sink associated with forest growth (Thomas et al. 2010).

Table 17. Net CO_2 Flux per Million Hectares from Forest Pools in Categories Forest Land Remaining Forest Land (FRF) and Land Converted to Forest Land (LCF)

For Years 1990–2017 (MMt CO2e per 1,000,000 Hectares).

	Forest Land Area (1,000,000 ha)	Net CO_2 Flux (CO_2e/yr)	Flux per Area (CO_{2e} yr^{-1} per million ha)
1990	7.7	-28.3	-3.7
1991	7.7	-28.2	-3.7
1992	7.7	-28.1	-3.7
1993	7.7	-28.0	-3.7
1994	7.6	-27.9	-3.6
1995	7.6	-27.7	-3.6
1996	7.6	-27.6	-3.6
1997	7.6	-27.5	-3.6
1998	7.6	-27.4	-3.6
1999	7.6	-27.2	-3.6
2000	7.6	-27.1	-3.6
2001	7.6	-27.0	-3.5
2002	7.6	-26.9	-3.5
2003	7.6	-26.8	-3.5
2004	7.6	-26.8	-3.5
2005	7.6	-26.7	-3.5
2006	7.6	-26.7	-3.5
2007	7.6	-26.6	-3.5
2008	7.6	-26.6	-3.5
2009	7.6	-26.5	-3.5
2010	7.6	-26.4	-3.5
2011	7.6	-26.3	-3.5
2012	7.6	-26.3	-3.5
2013	7.6	-26.1	-3.4
2014	7.6	-26.0	-3.4
2015	7.6	-25.8	-3.4
2016	7.6	-25.7	-3.4
2017	7.6	-25.5	-3.4

Increased N input can also stimulate soil microbial production of N_2O and CH_4, contributing to an offset of the increased CO_2 sink associated with N-induced increases in forest growth (Liu and Greaver 2009). Although N cycling associated with forests and other land cover types is complex and broad-scale patterns are poorly understood, if the recently observed decreased N deposition (Sullivan 2015) is the, or one of the, dominant driver(s) of the trend in the magnitude of the forest CO_2 sink and ambient N deposition has approximately reached a steady-state, it might be expected that the magnitude of the forest CO_2 sink will remain similar to the current magnitude for at least the next several decades.

The forests in the northeastern United States were largely cleared following European settlement; the rate of forest clearing generally slowed by the end of the 19th century or early 20th century. During the latter part of the 20th century, areas of former forest land were reforested or allowed to revert to forest. These land-use changes affected, and continue to affect, C fluxes in the forest sector. The amount of forest land remaining forest land will probably increase slowly in the coming years (EPA 2018). The amount of C stored in lumber and other harvested wood products in the United States will likely increase by a modest amount, perhaps 7 to 8 Tg per year, over the next two and a half decades (U.S. Department of State 2016).

Changes in climate, including temperature and patterns in precipitation, changes in N supply via atmospheric N deposition, and changes in CO_2 supply and ozone exposure all influence the growth (and distribution and survival) of trees. Furthermore, windthrow, insect infestation, fire, disease, and introduction of exotic plants and animals all can influence tree growth and survival at a particular location. Variation in year-to-year C flux in temperate forests, such as are found throughout much of New York State, is controlled largely by precipitation (Piao et al. 2009, Jung et al. 2011). Forest management of these stressors, and decisions made regarding timber harvesting and forest planting further modify forest C sequestration responses.

Conversion of agricultural and other non-forest land to forest land has resulted in substantial uptake of atmospheric CO_2 over the last several decades in the United States (Woodall et al. 2015). However, the rate of C uptake attributable to these lands is expected to decline in the future as these secondary forests age (Nabuurs et al. 2013, Coulston et al. 2015, Domke et al. 2018). The magnitude of the net C sink of New York State forested lands per unit area has decreased steadily between 1990 and 2017 (Figure 4B), with an overall decrease of about 10% over that time period (Figure 4A). The observed trend over that period was extrapolated out to the years 2030 and 2050, yielding estimates of annual forest C sequestration of -24.4 and -22.5 MMt CO_2e, respectively (Figure 4A) in those future years. Note that the minus sign indicates a net sink.

Wear and Coulston (2015) judged that the forest C sink in the eastern temperate area of North America, including New York State, will likely be relatively stable in the coming years. Domke et al. (2018) concluded that forests throughout the United States will likely continue to take up and store C, but in the absence of changes in forest management, the rate of uptake will likely decrease in the coming decades as the forests age and overall growth rates decline. Old-growth forests can continue to function as net C sinks as they age beyond 300 years (Keeton et al. 2011), but changing disturbance regimes may interfere with this capability. Management activities on working forest lands have the potential to shape these C sequestration and storage trends in both positive and negative directions. In working forests, future changes in C sequestration rates will depend to some degree on management actions such as harvest intensity, rotation length, and maintenance of growing stock, including advance regeneration (young trees that will establish the next cohort of canopy dominant trees). For a working forest to rapidly transition from a C source post-harvest to a C sink, effective regeneration is essential. A lack of sufficient regeneration, especially for commercial timber species, is a well-documented issue across forests of the State, resulting in part from historical management practices such as diameter-limit harvesting or "high-grading," as well as the impacts of disease, deer browsing, and introduced pests. Significant restoration efforts, which may involve some harvesting, will be needed to improve regeneration capacity so that replacement of existing canopy trees, whether they are lost due to natural or anthropogenic causes, can occur. Without adequate regeneration of commercial timber species, the likelihood of sustaining ownership of working forests could decrease, in turn increasing the potential for deforestation (land conversion to non-forest land uses) and the degradation or loss of the forest C sink. Moreover, the benefits of such restoration efforts extend well beyond climate mitigation and include biodiversity, the regulation of air, water, and soil quality, as well as provision of material goods (wood products, fuel) and cultural, recreational, and spiritual values. Adaptive management that accounts for uncertainty and change will be needed to deliver such benefits in a rapidly changing landscape. Overall, this situation highlights the need for improved forest stewardship, incorporating a range of practices to achieve landowner objectives and broader policy goals so that forest lands function as resilient C sinks.

Major influences on net C flux from forests in recent years include management actions, disturbance, and the effects of previous land conversions (especially agriculture to forest). These influences alter the amount of forested land and the storage and flux of C. Total annual C sequestration in the land use, land use-change, and forestry sectors in the United States decreased about 11.5% from 1990 to 2017, due mainly to a decrease in the rate of C sequestration in forests and croplands (EPA 2018).

If atmospheric inputs of N to the landscape continue to decrease, the resulting low availability of soil N in the northeastern United States in the future may limit the fertilizing effects of atmospheric CO_2 on forest C sequestration (Ciais et al. 2013). Such effects are not easily quantified. Further study will be needed to better understand the extent to which changes in forest C sequestration are associated with changes in atmospheric N deposition.

Forest growth varies with stand age and is expected to eventually decrease in New York State as forests continue to mature. Since around the turn of the 20th century, or earlier, forest lands in the State began to recover passively through natural regeneration and successional processes on lands previously used for agriculture. This land use transition caused a large increase in the magnitude of the forest C sink (Coulston et al. 2015). The future C sink provided by extant forests, as well as potential forest lands, is unclear because of the uncertainty in future changes in tree growth rates, land use, invasive species, management, major disturbances, and effects of wildlife population dynamics and climate change (Coulston et al. 2015, Domke et al. 2018). Insect pests, deer browsing, and invasive plants likely constitute the most proximate, near-term threats to the forest C sink in New York State.

Four strategies intended to optimize future CO_{2e} sequestration by the forest sector are highlighted below. These strategies are not mutually exclusive, but can be highly complementary in an integrative, overarching plan for forest ecosystem management.

4.4.1.1 Ecosystem Stewardship

Ecosystem stewardship expands on traditional notions of forestry to integrate knowledge and practice from other disciplines, such as conservation biology, restoration science, life-cycle analytics, and systems modeling to achieve a broader range of objectives and outcomes at multiple scales (from stand to landscape). Stewardship does not prescribe active intervention (management) as the default approach but recognizes that a forest landscape that encompasses multiple strategies and a range of management intensities (including none) is more likely to be resilient to multiple drivers of change. Across New York State, the existing mosaic of public and private forest lands, ranging from strict "forever wild" protections to intensive management, provides an ideal landscape for evaluating and adapting different stewardship approaches.

To facilitate this overarching approach of stewardship and its component strategies outlined below, the following initial steps are being taken by DEC and its partners:

- Create and implement a statewide forest C assessment and monitoring protocol.
- Conduct landscape modeling of forest change and ecosystem service outcomes (including C sequestration) resulting from biophysical, economic, and policy scenarios.
- Establish a network of adaptive management research sites across the State.

4.4.1.2 Improved Forest Management

A multifaceted strategy is needed to sustain and enhance the capacity of New York State's existing forests to function as resilient C sinks, while providing many other ecosystem services (or "co-benefits"), under the greater uncertainty and complexity posed by rapid environmental change. It is important to highlight the divergent goals and outcomes of forest management (i.e., harvesting and regenerating trees to maintain forest cover) versus deforestation (i.e., conversion of forest to another land use or cover type). In the absence of legal or regulatory protections, a forest that is managed is more likely to remain forested, relative to unmanaged land (Ruddell et al. 2007, Food and Agriculture Organization 2016). Moreover, management does not represent unchecked exploitation of the forest resource. Silviculture can capably address multiple landowner objectives either in lieu of, or in conjunction with, yields of forest products. For working forests, at the stand or parcel level, this strategy focuses on promoting "climate-smart" best management practices (BMPs), also known as "carbon forestry," that enhance the overall size and resilience of the forest C sink, in accordance with existing forestry BMP and landowner objectives (Malmsheimer et al. 2008, Ducey et al. 2013, Chen et al. 2015). Although there remains ongoing debate over relative benefits and tradeoffs of different silvicultural techniques and approaches (McKinley et al. 2011), examples of carbon forestry practices relevant to forests across New York State may include:

- prescriptions that enhance structural complexity, restore advance regeneration, and promote maintenance of uneven-aged and late-successional characteristics (Gunn et al. 2014, Ford and Keeton 2017).
- maintaining sufficient growing stock, especially advance regeneration, to hasten the stand re-initiation phase (Harmon et al. 1990, Nunery and Keeton 2010).
- experimental techniques to hasten forest recovery from intensified and/or novel disturbance regimes (Seymour et al. 2002).
- regenerating and/or establishing tree species that are expected to be resilient or well-adapted to forecasted changes in climate and disturbance regimes.

A related facet involves management practices that promote yields of forest products that have climate benefits (Birdsey et al. 2006), while minimizing net GHG emissions from an operational standpoint. While considerable debate exists over the net C balance of forest-based biofuels, there is broad consensus that high-quality, long-lived wood products provide substantial climate benefits, both in terms of actual C storage and for their substitution value as structural materials. Engineered wood products can be used in place of fossil fuel intensive products like steel, concrete, and plastic. The substitution benefits (i.e., forgone GHG emissions of the steel and concrete not produced) of wood products can be greater than the actual C stored in the wood products used instead (Malmsheimer et al. 2008).

With rapid advent of mass timber construction, which can incorporate lower-grade wood into engineered products, the potential seems promising for working forests—including those in need of restorative silviculture—to provide climate benefits while generating revenue. As with any management intervention, these potential benefits must be balanced with any undesirable or unintended impacts on the forest ecosystem.

4.4.1.3 Conversion of Non-forest Lands to Forests

This strategy seeks to achieve net gains in the geographical coverage of forests across New York State. Whether by active or passive means, the reforestation of non-forest land creates new C sinks that substantially increase the climate mitigation benefits associated with forests. Reforestation of lands actively being used for other purposes may not be a practical approach; however, efforts to increase tree cover in urban (e.g., tree planting programs) and agricultural (e.g., riparian buffers, silvopasture) landscapes have been proven to be both feasible and beneficial. Although significant reforestation potential exists on agricultural lands no longer in use (see section 4.2.1; Table 6), a common obstacle on fallow lands across New York State has been the establishment of exotic invasive shrubs such as buckthorn (*Rhamnus* spp.) and honeysuckle (*Lonicera* spp.), as well as native goldenrods (*Solidago* spp.), which inhibit native tree establishment and growth. Due to elevated deer populations that prefer to browse native vegetation, native tree seedlings and saplings that do establish beneath shrub cover are at a further disadvantage.

As a result, passive reforestation via "old field" successional processes cannot be relied on as a stand-alone strategy. Instead, vegetation and wildlife management to mitigate these invasions, reduce browsing pressure, and promote tree establishment are likely needed. Although operationally feasible, they will involve significant material and labor costs, and could involve controversial practices such as herbicide use and herd culling. Initial costs as well as stakeholder concerns may outweigh the near-term benefits of such actions, increasing the need for estimates of the long-term benefits associated with establishing healthy, productive forests in terms of GHG mitigation and other co-benefits. Other emerging or experimental practices may also be considered; for example, replacing invasive shrubs with woody biomass crops, such as shrub willow, could enhance the near-term benefits of land management and facilitate the transition to native forest cover.

4.4.1.4 Prevention of Forest Loss

This strategy seeks to maintain the C sink associated with existing forest land by preventing deforestation. In New York State, there are 3.8 million acres of publicly-owned forest preserve and state forest lands and 0.8 million acres of private "working forest" land under State-owned conservation easements; this equates to roughly one in four forested acres in the State being protected in perpetuity from deforestation. While this proportion is substantial, the distribution of these protected lands is almost entirely located in the Adirondack and Catskill regions. The future is less certain for the remaining private forest parcels, which are held by over 675,000 different landowners. Looking forward, programs that incentivize and/or subsidize forest landowners for the provision of ecosystem services that are non-rival public goods, such as climate mitigation and water purification, will likely be needed in areas where strong economic incentives exist (or will emerge) to convert forest land to other uses. Another model involves the use of conservation easements on working forest lands that prohibit deforestation (by current and all subsequent owners) and require a management plan developed by a professional forester that is compliant with "green certification" standards (e.g., SFI or FSC). Because easements are typically confidential agreements, their cost-benefit as a climate mitigation tool (i.e., per ton CO_2e) is unknown, but likely falls between subsidies/incentive payments and fee purchase of the land itself. Because actively managed forest lands are typically less likely to experience deforestation, the objective of preventing forest loss may be closely coupled with opportunities for improved forest management, as outlined above.

4.4.2 Agriculture

The estimates of agricultural mitigation potential presented herein are based on current emissions and one estimation of a technical mitigation potential with current technologies. Due to time limitations, future projections of either agricultural emissions or mitigation potential were not developed. Herein, factors that could affect the estimates of mitigation potential during coming decades are discussed. In New York State, there are ongoing robust trends of increasing efficiency in both crop and livestock production, which require fewer inputs (energy, feed, nutrients, etc.) to produce a unit of harvested crop or livestock product (Wightman et al. 2015b, Wightman and Woodbury 2016). These trends occur for crops such as maize and livestock, such as dairy cows that are managed intensively. Crops such as grass hay that are not generally managed intensively do not show such trends. These trends are likely to continue for some time into the future due to technological improvements and market forces that make intensification both feasible and profitable for farmers. There is also the possibility to manage some crops and lands more intensively to increase total production of crops and livestock if there are markets for additional production. However, the total production of crops and livestock in the State will continue to be driven by market forces, which are regional to global in scale for different agricultural products.

For dairy cows, a dominant agricultural system in the State, the trend during recent decades is for increases in average farm size, and this trend is expected to continue during coming decades (Wightman and Woodbury 2016). Along with increased farm size, there has been a trend towards increased storage of manure as a liquid in manure storage units rather than solid piles or daily spreading. This trend is due to logistics as well as to efforts by the State to reduce the potential for manure to adversely affect water quality due to movement of nutrients and fecal bacteria offsite driven by precipitation, especially on frozen ground. This trend towards increased liquid storage has increased GHG emissions and it is expected to continue during coming decades (Wightman and Woodbury 2016).

During this century, increasing climate change is likely to affect New York State agriculture in many ways, both directly and indirectly. Direct effects are projected to include both increased intensity of precipitation and warmer temperatures, especially during winter (Rosenzweig et al. 2011). There will be increasing opportunities for double-cropping and new crops due to warmer temperatures and longer frost-free periods (Wolfe et al. 2018). However, these same climatic factors will cause increased challenges with weeds and pests (Wolfe et al. 2018). There are also many opportunities to change management practices to adapt to climate change (Rosenzweig et al. 2011). Although it is expected that there will be more of both heavy precipitation events and summer drought, the State is expected to continue to have adequate water resources under climate change, although adaptation may include increased use of irrigation (Wolfe et al. 2018).

Indirect effects of climate change include the likelihood that other agricultural regions in the United States and the world will experience more damaging effects of climate change, which would increase the demand for agricultural products from New York State. It is difficult to project the overall effects of all these expected changes in both human systems (markets) and agricultural systems in coming decades. For this reason, for the preliminary estimates presented herein, future projections of changes to the agriculture sector of the State are not included, with the exception of the estimate for reforesting former agricultural lands, thus removing them from future agricultural production.

4.4.3 Wetlands

There is some potential for GHG mitigation if former wetlands that are currently used for agriculture are converted back to wetland (Richardson et al. 2014, Wang et al. 2016). Wetland restoration can increase

C storage (Lucchese et al. 2010, Kolka et al. 2018). However, the total area of estuaries relative to the area of tidal wetlands is expected to increase with sea level rise in the future (Fagherazzi et al. 2013). The primary production of estuaries may shift to favor phytoplankton as opposed to benthic algae and plants (Hopkinson et al. 2012). In response to such changes, the net effect of total wetlands and estuaries on C fluxes is uncertain, but will probably be reduced, mainly because of expected further loss of tidal wetlands (Windham-Myers et al. 2018).

4.5 Uncertainty

4.5.1 Forests

The uncertainty associated with estimates of forest land carbon stock changes were compiled using error propagation (IPCC Approach I) and included, where possible, information on sampling, measurement, and model uncertainty. Sampling uncertainty is based on the size of the sample and typically decreases with increasing sample size. Measurement uncertainty is based on quality control/quality assurance data from the FIA program where a subset of plots (ca. 9% of measured plots) are remeasured by a field supervisor to ensure that field crews are following protocols and maintaining field data collection standards. The measurement uncertainty of forest variables (e.g., tree diameter, tree height) is incorporated, when available, in the models used to estimate carbon stocks in each of the forest ecosystem carbon pools. Finally, the model uncertainty (residual standard error) is incorporated, when available, and collectively all sources of uncertainty are propagated in the population (State-level) estimates.

Uncertainties in the estimates of GHG fluxes from and to the forest sector presented in this report are summarized in Tables 18, 19, and 20. These uncertainty analyses for total net fluxes of forest C are consistent with the IPCC-recommended Tier 1 methodology (IPCC 2006). They are considered approach 1 (Propagation of error [section 3.2.3.1]; IPCC 2006). The upper and lower bounds are based on +/- one standard deviation. To better understand the effects of covariance, the contributions of sampling error and modeling error were parsed out. The upper and lower bounds in the flux estimates for Forest Land Remaining Forest Land ranged from -26.7 to -16.0 MMt CO_{2e}, reflecting an uncertainty of ±25%. Flux estimates for Land Converted to Forest Land (±9%; Table 20) and forest fires (±15% for CH_4; ±20% for N_2O; Table 18) were estimated to have smaller overall uncertainty because the estimated fluxes were much smaller than those for Forest Land Remaining Forest Land.

Table 18. Quantitative Uncertainty Estimates of Non-CO_2 Emissions from Forest Fires in the Category Forest Land Remaining Forest Land (MMt CO_2e and Percent)[a]

Source	Gas	2017 Emission Estimate	Uncertainty Range Relative to Emission Estimate[b]			
			Lower Bound	Upper Bound	Lower Bound (%)	Upper Bound (%)
Non-CO_2 Emissions from Forest Fires	CH_4	0.0027	0.0031	0.0024	-12%	15%
Non-CO_2 Emissions from Forest Fires	N_2O	0.0018	0.0021	0.0015	-15%	21%

[a] These estimates include Non-CO_2 Emissions from Forest Fires on Forest Land Remaining Forest Land and Land Converted to Forest Land.

[b] Range of flux estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

Table 19. Quantitative Uncertainty Estimates for Net CO_2 Flux from Category Forest Land Remaining Forest Land: Changes in Forest C Stocks (MMt CO_2e and Percent)

Source	2017 Flux Estimate (MMt CO_2e)	Lower Bound	Upper Bound	Lower Bound (%)	Upper Bound (%)
Forest C Pools[a]	-21.4	-26.7	-16.0	-25.0%	25.0%

[a] Range of flux estimates predicted through a combination of sample based and model based uncertainty, Approach 1 (propagation of error; IPCC [2006]).

Table 20. Quantitative Uncertainty Estimates for Net CO_2 Flux from Category Land Converted to Forest Land: Changes in Forest C Stocks (MMt CO_2e and Percent).

Uncertainty in mineral soil stock change for *Land Converted to Forest Land* was not available.

Land Use Category	Stock Change	Lower Bound	Lower Bound (%)	Upper Bound	Upper Bound (%)
Cropland Converted to Forest Land	**-1.1**	**-1.2**	**-17%**	**-0.9**	**17%**
Aboveground Biomass	-0.5	-0.7	-32%	-0.4	32%
Belowground Biomass	-0.1	-0.1	-20%	-0.1	20%
Dead Wood	-0.1	-0.2	-19%	-0.1	19%
Litter	-0.3	-0.3	-12%	-0.2	12%
Mineral Soil	ND	ND	ND	ND	ND
Other Lands Converted to Forest Land	**-1.9**	**-2.1**	**-14%**	**-1.6**	**14%**
Aboveground Biomass	-1.1	-1.4	-23%	-0.9	23%
Belowground Biomass	-0.2	-0.3	-25%	-0.2	25%
Dead Wood	-0.2	-0.2	-24%	-0.1	24%
Litter	-0.3	-0.4	-13%	-0.3	13%
Mineral Soil					
Settlements Converted to Forest Land	**-1.2**	**-1.4**	**-15%**	**-1.0**	**15%**
Aboveground Biomass	-0.7	-0.9	-25%	-0.5	25%
Belowground Biomass	-0.1	-0.2	-27%	-0.1	27%
Dead Wood	-0.1	-0.2	-24%	-0.1	24%
Litter	-0.2	-0.3	-14%	-0.2	14%
Mineral Soil					
Total: Aboveground Biomass	**-2.4**	**-2.7**	**-15%**	**-2.0**	**15%**
Total: Belowground Biomass	**-0.5**	**-0.5**	**-15%**	**-0.4**	**15%**
Total: Dead Wood	**-0.4**	**-0.5**	**-13%**	**-0.4**	**13%**
Total: Litter	**-0.9**	**-0.9**	**-7%**	**-0.8**	**7%**
Total: Mineral Soils					
Total: Land Converted to Forest Land	**-4.1**	**-4.5**	**-9%**	**-3.8**	**9%**

The uncertainty analyses for total net flux of forest C are consistent with the IPCC-recommended Tier 1 methodology (IPCC 2006). Specifically, they are considered approach 1 (propagation of error [section 3.2.3.1]; IPCC 2006). The upper and lower bounds are based on +/- one standard deviation. To better understand the effects of covariance, the contributions of sampling error and modeling error were parsed out.

Lajtha et al. (2018) discussed sources of uncertainty associated with estimation of the amount of C redistribution that takes place in forests as a result of soil erosion. These include various soil and landscape properties, land-use history, and changes in climate, including extreme climatic events associated with droughts, floods, and storms. Two key caveats are important for interpretation of the FIA-based analysis presented here. The first relates to propagation of error due to aggregation of fine-scale plot-level data to the much broader scale of political units, which is a common issue (as noted above). The caveat is not the empirical error itself, but the inability to identify different sources of uncertainty, to parse them out from causal factors, and to assess their relative importance in explaining the numerical results. In a highly variable landscape like New York State, this makes it very difficult to explain the trends being described at a highly aggregated level. For example, it can be reasonably speculated that gradual declining trends in the annual net C sink for the State may be attributed to aging forests, insect mortality, poor regeneration, deer browsing, soil degradation, past land use, or these along with many other factors in various combinations. A more detailed and spatially explicit assessment of the forest landscape is needed to provide a basis for interpreting these trends and their causal factors in ways that promote better understanding and decision-making.

The second caveat follows a similar logic but is more specifically related to the use of FIA plot-level data to estimate land-cover changes over time based on plot resampling. Because FIA plot locations are distributed on a semi-uniform grid for statistical purposes, the plots can end up in non-forest land cover types. In such cases, nominal data are recorded including the current land-cover/land-use type, which allows FIA's system to observe plot-level changes to and from forest cover. On this basis, FIA can offer some highly aggregated (or synoptic) insights on land-use/land-cover change. However, given that this program was not designed to measure/model land-use change, there are limits to the interpretation. Other data products based on remote sensing, such as the National Land Cover Dataset (NLCD) derived from Landsat imagery, provide a more authoritative and spatially explicit estimate of changes in forest cover over time. When estimates of the forest C sink are adjusted or extrapolated using land cover estimates from FIA, this introduces additional uncertainty that can limit interpretation.

Forest C stocks in New York State, especially in the Adirondack Mountains, have generally been recovering from past logging activities since around the early 20th century. In response to this forest recovery, C uptake into forest vegetation has occurred in association with afforestation and tree growth. However, the net effect of forest management on C stocks and uptake fluxes remains uncertain. Forest thinning likely enhances C storage in the long term (Loudermilk et al. 2017) but this is difficult to quantify.

It is clear that increased atmospheric CO_2 concentrations and increased temperature both can stimulate tree growth (Norby et al. 2005, Melillo et al. 2011). In addition, N supply from the atmosphere can influence soil fertility, tree growth, and mortality (Thomas et al. 2010). Drought constitutes an additional important forest stressor, decreasing growth and increasing mortality. The magnitude of such effects in the future remains very unclear, however, due in part to spatial differences in soil fertility, including the availability of N, P, and calcium (Ca), and the dynamics of soil microbes (Finzi and Schlesinger 2002, Terrer et al. 2016). For example, the regeneration and health of sugar maple is limited by Ca availability across the Adirondack region, with a gradient of increasing soil exchangeable Ca from the southwest towards the northeast (Sullivan et al. 2013).

The soil stores much of the C found in forest ecosystems, and soil C turns over slowly (van Groenigen et al. 2014). Soil warming increases the rate of plant material degradation and contributes to more pronounced release of CO_2 from soil into the atmosphere (Melillo et al. 2017). Changes in climatic and atmospheric conditions affect the C cycle in ways that are not well understood. This makes it difficult to predict the effects of such changes on future rates of C sequestration in forests of the State. Forest insects and diseases, such as for example beech bark disease (Clark et al. 2010, Lawrence et al. 2018), further alter C cycling.

Prediction of future changes in C fluxes and stocks will remain uncertain in the absence of more knowledge of expected changes in forest structure and species composition caused by changes in climate, atmospheric chemistry, disturbance, and management (Domke et al. 2018). Climate change may alter the cycling of C in the forests of New York State in ways that are difficult to quantify. With adequate soil moisture supply, higher temperatures can lead to a longer growing season, which would lead to increased C uptake into vegetation. Higher atmospheric CO_2 concentration leads to enhanced photosynthesis. Other factors can have opposite effects. Prolonged drought increases plant stress, reducing growth and increasing mortality. The net effect of these factors in the future is difficult to predict with much certainty. It is clear, however, that modifying tree removal rates and changing forestry practices could have substantial impacts on future C sequestration and forest C stocks (Erb et al. 2013). Management decisions aimed at increasing forest C sequestration will need to be evaluated in conjunction with other objectives regarding habitat for threatened and endangered species, provision of ecosystem services, and other considerations (Ray et al. 2009, Domke et al. 2018).

Lands that were converted from forest to agriculture in the past lost a substantial component of their pre-1800 topsoil C to erosion and CO_2 emissions. Land management in more recent years resulted in relatively smaller amounts of soil C loss (Lajtha et al. 2018). The transfer of C across the land and into surface waters is important, but poorly understood. To further complicate the picture, increased temperature with climate change can accelerate soil C loss. Model projections of the direction and magnitude of change in the fluxes and stocks of C in the soil between now and the turn of the next century are highly variable (Lajtha et al. 2018). Changes in these projected fluxes and stocks vary with management of agricultural, forest, and wetland systems. Most of the organic C in forest soil is stored in the top ~1 m of soil (Liu et al. 2013). This is also the portion of the soil profile that is most strongly influenced by land-use change, extreme events and other disturbances, management actions, and climate change (Lajtha et al. 2018).

Water availability, as reflected in precipitation amounts and patterns, and soil moisture influence soil C cycling. At locations and times where and when moisture is limiting, an increase in water supply can increase soil microbial activity. This causes an increase in soil respiration and an increased outflux of CO_2. Nevertheless, moisture availability also influences vegetative type, plant species distribution, and productivity, potentially increasing or decreasing C storage (Jobbágy and Jackson 2000).

Microorganisms that live in the soil include fungi, bacteria, archaea, and others. They process and break down soil organic matter. In doing so, they release CO_2 and CH_4 through microbial respiration and reduction (Bernal et al. 2016). They play major roles in regulating temperature and moisture availability to plant roots (Yan et al. 2016, Lajtha et al. 2018). Larger soil fauna, including worms, insects, and others, are also active in breaking down soil organic matter (Orgiazzi et al. 2015). They help to promote organic matter decomposition and leaching losses of DOM (de Vries et al. 2013). Within the soil rooting zone (rhizophere), microbial activity is high, and plant roots release exudates that are made up of various organic acids, amino acids, simple sugars, and other carbohydrates (Hirsch et al. 2013).

4.5.2 Agriculture

A quantitative uncertainty analysis was not performed due to data availability and the limited time available. Comments on the uncertainty of some agricultural categories and sub-categories are included in the results section. Additionally, there is uncertainty about all of the estimates provided herein due to limitations in the underlying data used, the need to make assumptions about key factors that are not measured, and the difficulty in estimating the extent to which a technical mitigation potential might be achieved on working farms. There is also great variation in the biophysical characteristics of different farms and fields as well as in the details of management practices that affect GHG emissions. Furthermore, there are challenges with assessing the potential for leakage as discussed previously, which can greatly affect estimates of mitigation potential. For example, the benefits of reducing food waste, as has been estimated herein, depend on an assumption that improved production efficiency reduces GHG emissions, which is only likely to be true if reduced importation of food and/or feed into the State is counted in the net GHG benefit (i.e., positive leakage). For some practices, there are also challenges with the permanence of the mitigation benefit. Permanence is an issue with any practice that depends on long-term C-sequestration in soil and roots or in aboveground vegetation, especially trees. Such practices that are included in Table 6 include cover crops, replacing annual crops with perennial crops, alley cropping, and reforestation. All these practices depend on a change in practice (or a change in land cover) maintained in perpetuity; if not, the GHG mitigation benefit might be lost due to emission of the stored C back to the atmosphere.

4.5.3 Wetlands

Carbon is exported from inland waters in the form of dissolved organic C (DOC), dissolved inorganic C (DIC), particulate organic C (POC), and particulate inorganic C (PIC). These forms of C originate from wetlands, riparian zones, and upper soil horizons throughout the landscape. Dissolved CO_2 is part of the DIC component of water transport. Both natural and human-caused C sources to inland waters can be important, but it is difficult to allocate C sources in this fashion. Rivers deliver C from inland areas to estuaries and coastal areas. As water flows down rivers, CO_2 and CH_4 move to or from the water and vegetation surfaces (Tranvik et al. 2009, Stanley et al. 2016, Butman et al. 2018). The sediments of lakes, rivers, and reservoirs act as C sinks, but can also remobilize organic C to gaseous C which can be emitted back to the atmosphere (Clow et al. 2015). The balance among these transfers is spatially and temporally variable and poorly known (Arntzen et al. 2013).

The total inland water C flux is represented as the sum of the lateral transport of DIC and total organic C (TOC) from surface water to the coast, CO_2 emissions from surface waters, and the amount of C buried in sediments (Butman et al. 2016, Butman et al. 2018). The net flux of C from inland water is highly uncertain, in particular because of variability associated with extreme hydrological events. Also, the levels of CH_4 emissions from inland waters to the atmosphere are largely unknown (Butman et al. 2018). Tidal wetlands in North America act as sinks for atmospheric C. In contrast, estuaries constitute a net C source to the atmosphere. Effects of human-caused natural disturbance on wetland GHG fluxes are uncertain. The CO_2 and CH_4 fluxes from and to wetlands are highly variable. Across North America, the overall effect is that the flux across the combined coastal wetlands-estuary system appears to be negative (representing a net sink; Windham-Myers et al. 2018).

Although estimates of the standard error around the average literature-based palustrine wetland GHG flux values were included in the analysis presented here, a high degree of uncertainty remains in the extent to which these few literature values of GHG fluxes (many of which are from studies located outside of the State) are representative of the population of palustrine wetlands in New York State.

Additional uncertainty exists in the extent to which the CO_2 flux estimates from forested palustrine wetlands are also considered within the CO_2 flux estimates for forest ecosystems described in the Forest Sector section, section 4.1) of this report. As such, the CO_2 fluxes reported for forest ecosystems and forested palustrine wetlands should not be considered mutually exclusive.

A wetland can alternate between acting as a C sink during wet periods and a C source during dry periods (Kolka et al. 2018). Wetlands provide a substantial source of CH_4 to the atmosphere (Kirschke et al. 2013). There is great uncertainty about how wetland C fluxes will respond to future changes in temperature and precipitation, due in large part to uncertain effects on productivity and decomposition. Such conditions will vary with wetland type and especially wetland hydrology (Olefeldt et al. 2013).

Warmer and dryer conditions might promote and intensify wildfire in peatlands, which will contribute to increased peatland C fluxes (Turetsky et al. 2010). Climate change would also be expected to speed up organic matter decomposition and export, mainly for peatland wetland types (Kolka et al. 2018). The overall response of wetlands to ongoing climate change is uncertain. Emissions of CH_4 from wetlands will probably increase in response to higher atmospheric CO_2 levels and warmer climate (Ciais et al. 2013).

One of the most important uncertainties in quantifying the role of tidal wetlands and estuaries in C cycling in New York State pertains to the effects of sea level rise on the C cycle. Other important limitations include the insufficiency of maps of wetland and estuary extent and the amount of gas exchange (CO_2, N_2O, and CH_4) across water surfaces (Windham-Myers et al. 2018). The overall role of coastal areas in the C cycle and balance is not well understood (Borges et al. 2005). Stocks in, and fluxes from, tidal wetlands are highly uncertain (perhaps by a factor of 2) because of extreme climatic events, wetland mapping uncertainty, and unknown future wetland disturbance regimes (Lane et al. 2016, Couvillion et al. 2017, Ward et al. 2017, Windham-Myers et al. 2018).

Methane emissions from estuaries in the United States are not very well known (Borges and Abril 2011). Flux estimates ranged for the studies reviewed by Windham-Myers et al. (2018), from 0.04 to 8 g C per m^2 per year at Atlantic coast sites. Variation in CH_4 flux from wetlands can be pronounced and has been found to be associated with vegetation type, temperature, soil moisture, and water table height (Friborg et al. 2000, Werner et al. 2003). In addition, the close coupling between CH_4 emissions and salinity introduces great uncertainty into CH_4 flux estimates (Windham-Myers et al. 2018). Variations in C export from estuaries is largely due to flooding and other extreme hydrological events (Ren et al. 2015, Tian et al. 2016). Human modifications of estuaries, including such actions as ditching and nutrient addition, influence C fluxes (Kirwan and Megonigal 2013, Windham-Myers et al. 2018). Future sea level rise is expected to have substantial impacts on these C fluxes, especially due to shoreline erosion (Morris et al. 2016).

Ocean warming will alter water flows into and out of estuaries, thereby impacting the C cycle. Such changes to estuary hydrology might change tidal wetlands from sinks to sources, largely due to storms and other disturbance regimes (Pendleton et al. 2012). Shifts from saline to freshwater dominance can affect C cycling, especially as CH_4, emissions (Kroeger et al. 2017). Atmospheric deposition and riverine transport of N to estuaries will affect emissions of N_2O (Moseman-Valtierra et al. 2011).

5 Conclusions

In aggregate, the land cover types included in this study were considered to have a net annual flux of -12.33 MMt CO_2e. Although it is challenging to estimate future conditions, a value of -18.56 MMt CO_2e can be considered as a reasonable estimate of future (approximately year 2050) net GHG flux conditions based on linear extrapolation of the forest flux, full implementation of agricultural GHG mitigation potential, and continued constant GHG fluxes associated with wetlands. Improved forest management and prevention of forest loss might be important strategies for further increasing the GHG mitigation potential of natural and working lands of New York State.

6 References

Amiro, B.D., A.G. Barr, J.G. Barr, T.A. Black, R. Bracho, M. Brown, J. Chen, K.L. Clark, K.J. Davis, A.R. Desai, S. Dore, V. Engel, J.D. Fuentes, A.H. Goldstein, M.L. Goulden, T.E. Kolb, M.B. Lavigne, B.E. Law, H.A. Margolis, T. Martin, J.H. McCaughey, L. Misson, M. Montes-Helu, A. Noormets, J.T. Randerson, G. Starr, and J. Xiao. 2010. Ecosystem carbon dioxide fluxes after disturbance in forests of North America. Journal of Geophysical Research: Biogeosciences 115(G4). 10.1029/2010jg001390.

Anderegg, W.R.L., J.M. Kane, and L.D.L. Anderegg. 2013a. Consequences of widespread tree mortality triggered by drought and temperature stress. Nature Climate Change 3:30. 10.1038/nclimate1635.

Anderegg, W.R.L., L. Plavcová, L.D.L. Anderegg, U.G. Hacke, J.A. Berry, and C.B. Field. 2013b. Drought's legacy: multiyear hydraulic deterioration underlies widespread aspen forest die-off and portends increased future risk. Global Change Biology 19(4):1188-1196. 10.1111/gcb.12100.

Arntzen, E.V., B.L. Miller, A.C. O'Toole, S.E. Niehus, and M.C. Richmond. 2013. Evaluating Greenhouse Gas Emissions from Hydropower Complexes on Large Rivers in Eastern Washington. PNNL-22297. U.S. Department of Energy, Pacific Northwest National Laboratory, Richland, WA. http://www.pnl.gov/main/publications/external/technical_reports/PNNL-22297.pdf].

Artigas, F., J.Y. Shin, C. Hobble, A. Marti-Donati, K.V.R. Schäfer, and I. Pechmann. 2015. Long term carbon storage potential and CO2 sink strength of a restored salt marsh in New Jersey. Agricultural and Forest Meteorology 200:313-321. https://doi.org/10.1016/j.agrformet.2014.09.012.

Bansal, S., B.A. Tangen, and R. Finocchiaro. 2016. Temperature and hydrology affect methane emissions from Prairie Pothole Wetlands. Wetlands 36(s2):371-381. 10.1007/s13157-016-0826-8.

Bartlett, K.B., D.S. Bartlett, R.C. Harriss, and D.I. Sebacher. 1987. Methane emissions along a salt marsh salinity gradient. Biogeochemistry 4(3):183-202. 10.1007/bf02187365.

Batjes, N.H. 2014. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science 65(1):10-21. 10.1111/ejss.12114_2.

Bauer, J.E., W.-J. Cai, P.A. Raymond, T.S. Bianchi, C.S. Hopkinson, and P.A.G. Regnier. 2013. The changing carbon cycle of the coastal ocean. Nature 504:61.

Bechtold, W.A. and P.L. Patterson. 2005. The Enhanced Forest Inventory and Analysis Program--National Sampling Design and Estimation Procedures. Gen. Tech. Rep. SRS-80. U.S. Dept. of Agriculture, Forest Service, Southern Research Station, Asheville, NC. https://doi.org/10.2737/SRS-GTR-80.

Bellassen, V. and S. Luyssaert. 2014. Carbon sequestration: Managing forests in uncertain times. Nature 506(7487):153. 10.1038/506153a.

Bellassen, V., N. Viovy, S. Luyssaert, G. Le Maire, M.-J. Schelhaas, and P. Ciais. 2011. Reconstruction and attribution of the carbon sink of European forests between 1950 and 2000. Global Change Biology 17(11):3274-3292. 10.1111/j.1365-2486.2011.02476.x.

Berhe, A.A., J. Harte, J.W. Harden, and M.S. Torn. 2007. The significance of the erosion-induced terrestrial carbon sink. BioScience 57(4):337-346. 10.1641/b570408.

Bernal, B., D.C. McKinley, B.A. Hungate, P.M. White, T.J. Mozdzer, and J.P. Megonigal. 2016. Limits to soil carbon stability; Deep, ancient soil carbon decomposition stimulated by new labile organic inputs. Soil Biology and Biochemistry 98:85-94. https://doi.org/10.1016/j.soilbio.2016.04.007.

Birdsey, R. 1996. Carbon storage for major forest types and regions in the conterminous United States. *In*: Sampson, R.N. and D. Hair (Eds.). Forest and Global Change, Volume 2: Forest Management Opportunities for Mitigating Carbon Emissions. American Forests, Washington, D.C.

Birdsey, R., K. Pregitzer, and A. Lucier. 2006. Forest carbon management in the United States: 1600-2100. J Environ Qual 35(4):1461-1469. 10.2134/jeq2005.0162.

Blossey, B., A. Dávalos, and V. Nuzzo. 2017. An indicator approach to capture impacts of white-tailed deer and other ungulates in the presence of multiple associated stressors. AoB PLANTS 9(5). 10.1093/aobpla/plx034.

Bonneville, M.-C., I.B. Strachan, E.R. Humphreys, and N.T. Roulet. 2008. Net ecosystem CO_2 exchange in a temperate cattail marsh in relation to biophysical properties. Agricultural and Forest Meteorology 148(1):69-81. https://doi.org/10.1016/j.agrformet.2007.09.004.

Borges, A. and G. Abril. 2011. Carbon dioxide and methane dynamics in estuaries. *In*: Wolanski, E. and D.S. McLusky (Eds.). Treatise on Estuarine and Coastal Science, Vol. 5. Academic Press, Waltham. pp. 119-161.

Borges, A.V., B. Delille, and M. Frankignoulle. 2005. Budgeting sinks and sources of CO 2 in the coastal ocean: Diversity of ecosystems counts. Geophysical Research Letters 32(14):n/a-n/a. 10.1029/2005GL023053.

Bridgham, S.D., J.P. Megonigal, J.K. Keller, N.B. Bliss, and C. Trettin. 2006. The carbon balance of North American wetlands. Wetlands 26(4):889-916. 10.1672/0277-5212(2006)26[889:TCBONA]2.0.CO;2.

Butman, D., S. Stackpoole, E. Stets, C.P. McDonald, D.W. Clow, and R.G. Striegl. 2016. Aquatic carbon cycling in the conterminous United States and implications for terrestrial carbon accounting. Proceedings of the National Academy of Sciences 113(1):58. 10.1073/pnas.1512651112.

Butman, D., R. Striegl, S. Stackpoole, P.d. Giorgio, Y. Prairie, D. Pilcher, P. Raymond, F.P. Pellat, and J. Alcocer. 2018. Chapter 14: Inland waters. *In*: Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao and Z. Zhu (Eds.). Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report. U.S. Global Change Research Program, Washington, DC. pp. 568-595.

Buzby, J.C., H.F. Wells, and J. Hyman. 2014. The Estimated Amount, Value, and Calories of Postharvest Food Losses at the Retail and Consumer Levels in the United States. Economic Information Bulletin Number 121. USDA, Economic Research Service, Washington, DC. 39 pp.

Cahoon, D.R. 2006. A Review of Major Storm Impacts on Coastal Wetland Elevations. Estuaries and Coasts 29(6):889-898.

Canuel, E.A., S.S. Cammer, H.A. McIntosh, and C.R. Pondell. 2012. Climate change impacts on the organic carbon cycle at the land-ocean interface. Annual Review of Earth and Planetary Sciences 40(1):685-711. 10.1146/annurev-earth-042711-105511.

CCSP. 2007. First State of the Carbon Cycle Report (SOCCR): The North American Carbon Budget and Implications for the Global Carbon Cycle. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. King, A.W., L. Dilling, G.P. Zimmerman, D.M. Fairman, R.A. Houghton, G. Marland, A. Z. Rose and T.J. Wilbanks (Eds.). National Oceanic and Atmospheric Administration, National Climatic Data Center, Asheville, NC.

Chen, J., J. Xu, R. Jensen, and J. Kabrick. 2015. Changes in aboveground biomass following alternative harvesting in oak-hickory forests in the eastern USA. iForest - Biogeosciences and Forestry 8:e1-e9. 10.3832/ifor1349-007.

Chmura, G.L., S.C. Anisfeld, D.R. Cahoon, and J.C. Lynch. 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles 17(4). 10.1029/2002gb001917.

Chu, H., J.F. Gottgens, J. Chen, G. Sun, A.R. Desai, Z. Ouyang, C. Shao, and K. Czajkowski. 2015. Climatic variability, hydrologic anomaly, and methane emission can turn productive freshwater marshes into net carbon sources. Global Change Biology 21(3):1165-1181. 10.1111/gcb.12760.

Ciais, P., M.J. Schelhaas, S. Zaehle, S.L. Piao, A. Cescatti, J. Liski, S. Luyssaert, G. Le-Maire, E.D. Schulze, O. Bouriaud, A. Freibauer, R. Valentini, and G.J. Nabuurs. 2008. Carbon accumulation in European forests. Nature Geoscience 1:425. 10.1038/ngeo233.

Ciais, P., C. Sabine, G. Bala, L. Bopp, V. Brovkin, J. Canadell, A. Chhabra, R. DeFries, J. Galloway, M. Heimann, C. Jones, C.L. Quéré, R.B. Myneni, S. Piao, and P. Thornton. 2013. Carbon and other biogeochemical cycles. *In*: Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (Eds.). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. pp. 465-570.

Clark, K.L., N. Skowronski, and J. Hom. 2010. Invasive insects impact forest carbon dynamics. Global Change Biology 16(1):88-101. 10.1111/j.1365-2486.2009.01983.x.

Clow, D.W., S.M. Stackpoole, K.L. Verdin, D.E. Butman, Z. Zhu, D.P. Krabbenhoft, and R.G. Striegl. 2015. Organic carbon burial in lakes and reservoirs of the conterminous United States. Environmental Science & Technology 49(13):7614-7622. 10.1021/acs.est.5b00373.

Conley, A.K., E.L. White, and T.G. Howard. 2018. New York State Riparian Opportunity Assessment. New York Natural Heritage Program, State University of New York College of Environmental Science and Forestry, Albany, NY.

Coulston, J.W., D.N. Wear, and J.M. Vose. 2015. Complex forest dynamics indicate potential for slowing carbon accumulation in the southeastern United States. Scientific Reports 5, 8002. 10.1038/srep08002.

Couvillion, B.R., H. Beck, D. Schoolmaster, and M. Fischer. 2017. Land area change in coastal Louisiana 1932 to 2016. U.S. Geological Survey Scientific Investigations Map 3381 + pamphlet. 16 pp. .

Covey, K.R. and J.P. Megonigal. 2019. Methane production and emissions in trees and forests. New Phytologist 222(1):35-51. 10.1111/nph.15624.

Dahl, T.E. 1990. Wetlands Losses in the United States: 1780's to 1980's. U.S. Fish and Wildlife Service, Washington, DC. https://www.fws.gov/wetlands/Documents/Wetlands-Lossesin-the-United-States-1780s-to-1980s.pdf.

Davidson, E.A. and I.A. Janssens. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 440:165+.

de Vries, F.T., E. Thébault, M. Liiri, K. Birkhofer, M.A. Tsiafouli, L. Bjørnlund, H. Bracht Jørgensen, M.V. Brady, S. Christensen, P.C. de Ruiter, T. d'Hertefeldt, J. Frouz, K. Hedlund, L. Hemerik, W.H.G. Hol, S. Hotes, S.R. Mortimer, H. Setälä, S.P. Sgardelis, K. Uteseny, W.H. van der Putten, V. Wolters, and R.D. Bardgett. 2013. Soil food web properties explain ecosystem services across European land use systems. Proceedings of the National Academy of Sciences 110(35):14296. 10.1073/pnas.1305198110.

Domke, G., C.A. Williams, R. Birdsey, J. Coulston, A. Finzi, C. Gough, B. Haight, J. Hicke, M. Janowiak, B.d. Jong, W.A. Kurz, M. Lucash, S. Ogle, M. Olguín-Álvarez, Y. Pan, M. Skutsch, C. Smyth, C. Swanston, P. Templer, D. Wear, and C.W. Woodal. 2018. Chapter 9: Forests. *In*: Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao and Z. Zhu (Eds.). Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report. U.S. Global Change Research Program, Washington, DC, USA. pp. 365-398.

Domke, G.M., C.W. Woodall, and J.E. Smith. 2011. Accounting for density reduction and structural loss in standing dead trees: Implications for forest biomass and carbon stock estimates in the United States. Carbon Balance and Management 6:14.

Domke, G.M., C.W. Woodall, B.F. Walters, and J.E. Smith. 2013. From models to measurements: comparing downed dead wood carbon stock estimates in the U.S. forest inventory. PLoS ONE 8(3):e59949. 10.1371/journal.pone.0059949.

Domke, G.M., C.W. Woodall, J.E. Smith, J.A. Westfall, and R.E. McRoberts. 2012. Consequences of alternative tree-level biomass estimation procedures on U.S. forest carbon stock estimates. Forest Ecology and Management 270:108-116. https://doi.org/10.1016/j.foreco.2012.01.022.

Domke, G.M., C.H. Perry, B.F. Walters, C.W. Woodall, M.B. Russell, and J.E. Smith. 2016. Estimating litter carbon stocks on forest land in the United States. Science of The Total Environment 557-558:469-478. https://doi.org/10.1016/j.scitotenv.2016.03.090.

Domke, G.M., C.H. Perry, B.F. Walters, L.E. Nave, C.W. Woodall, and C.W. Swanston. 2017. Toward inventory-based estimates of soil organic carbon in forests of the United States. Ecological Applications 27(4):1223-1235. 10.1002/eap.1516.

Drexler, J.Z., C.S. de Fontaine, and S.J. Deverel. 2009. The legacy of wetland drainage on the remaining peat in the Sacramento — San Joaquin Delta, California, USA. Wetlands 29(1):372-386. 10.1672/08-97.1.

Ducey, M., J. Gunn, and A. Whitman. 2013. Late-successional and old-growth forests in the northeastern United States: Structure, dynamics, and prospects for restoration. Forests 4:1055-1086. 10.3390/f4041055.

Eidenshink, J., B. Schwind, K. Brewer, Z.L. Zhu, B. Quayle, and S. Howard. 2007. A project for monitoring trends in burn severity. Fire Ecology 31(1):3-21.

Elser, J.J., M.E.S. Bracken, E.E. Cleland, D.S. Gruner, W.S. Harpole, H. Hillebrand, J.T. Ngai, E.W. Seabloom, J.B. Shurin, and J.E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters 10(12):1135-1142. 10.1111/j.1461-0248.2007.01113.x.

Erb, K.-H., T. Kastner, S. Luyssaert, R.A. Houghton, T. Kuemmerle, P. Olofsson, and H. Haberl. 2013. Bias in the attribution of forest carbon sinks. Nature Climate Change 3:854. 10.1038/nclimate2004.

Fagherazzi, S., G. Mariotti, P. Wiberg, and K. McGlathery. 2013. Marsh collapse does not require sea level rise. Oceanography 26(3):70-77. 10.5670/oceanog.2013.47.

Fargione, J.E., S. Bassett, T. Boucher, S.D. Bridgham, R.T. Conant, S.C. Cook-Patton, P.W. Ellis, A. Falcucci, J.W. Fourqurean, T. Gopalakrishna, H. Gu, B. Henderson, M.D. Hurteau, K.D. Kroeger, T. Kroeger, T.J. Lark, S.M. Leavitt, G. Lomax, R.I. McDonald, J.P. Megonigal, D.A. Miteva, C.J. Richardson, J. Sanderman, D. Shoch, S.A. Spawn, J.W. Veldman, C.A. Williams, P.B. Woodbury, C. Zganjar, M. Baranski, P. Elias, R.A. Houghton, E. Landis, E. McGlynn, W.H. Schlesinger, J.V. Siikamaki, A.E. Sutton-Grier, and B.W. Griscom. 2018. Natural climate solutions for the United States. Science Advances 4(11):eaat1869. 10.1126/sciadv.aat1869.

Finzi, A.C. and W.H. Schlesinger. 2002. Species control variation in litter decomposition in a pine forest exposed to elevated CO_2. Global Change Biology 8(12):1217-1229. 10.1046/j.1365-2486.2002.00551.x.

Food and Agriculture Organization (FAO). 2016. Forestry for a Low-Carbon Future: Integrating Forests and Wood Products in Climate Change Strategies. FAO Forestry Paper 177. Food and Agriculture Organization of the United Nations, Rome.

Forbrich, I. and A.E. Giblin. 2015. Marsh-atmosphere CO_2 exchange in a New England salt marsh. Journal of Geophysical Research: Biogeosciences 120(9):1825-1838. 10.1002/2015jg003044.

Ford, S.E. and W.S. Keeton. 2017. Enhanced carbon storage through management for old-growth characteristics in northern hardwood-conifer forests. Ecosphere 8(4):e01721. 10.1002/ecs2.1721.

Frayer, W.E. and G.M. Furnival. 1999. Forest survey sampling designs: A history. Journal of Forestry 97(12):4-10. 10.1093/jof/97.12.4.

Friborg, T., T.R. Christensen, B.U. Hansen, C. Nordstroem, and H. Soegaard. 2000. Trace gas exchange in a high-Arctic valley: 2. Landscape CH4 fluxes measured and modeled using eddy correlation data. Global Biogeochemical Cycles 14(3):715-723. 10.1029/1999gb001136.

Galloway, J.N., J.D. Aber, J.W. Erisman, S.P. Seitzinger, R.W. Howarth, E.B. Cowling, and B.J. Cosby. 2003. The nitrogen cascade. BioScience 53(4):341-356.

Griscom, B.W., J. Adams, P.W. Ellis, R.A. Houghton, G. Lomax, D.A. Miteva, W.H. Schlesinger, D. Shoch, J.V. Siikamäki, P. Smith, P. Woodbury, C. Zganjar, A. Blackman, J. Campari, R.T. Conant, C. Delgado, P. Elias, T. Gopalakrishna, M.R. Hamsik, M. Herrero, J. Kiesecker, E. Landis, L. Laestadius, S.M. Leavitt, S. Minnemeyer, S. Polasky, P. Potapov, F.E. Putz, J. Sanderman, M. Silvius, E. Wollenberg, and J. Fargione. 2017. Natural climate solutions. Proceedings of the National Academy of Sciences 114(44):11645. 10.1073/pnas.1710465114.

Gruber, N. and J.N. Galloway. 2008. An Earth-system perspective of the global nitrogen cycle. Nature 451:293-296.

Gunn, J.S., M.J. Ducey, and A.A. Whitman. 2014. Late-successional and old-growth forest carbon temporal dynamics in the Northern Forest (northeastern USA). Forest Ecology and Management 312:40-46. https://doi.org/10.1016/j.foreco.2013.10.023.

Guo, L.B. and R.M. Gifford. 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8(4):345-360. 10.1046/j.1354-1013.2002.00486.x.

Hanson, P.J., A.L. Gill, X. Xu, J.R. Phillips, D.J. Weston, R.K. Kolka, J.S. Riggs, and L.A. Hook. 2016. Intermediate-scale community-level flux of CO_2 and CH_4 in a Minnesota peatland: putting the SPRUCE project in a global context. Biogeochemistry 129(3):255-272. 10.1007/s10533-016-0230-8.

Harmon, M.E., W.K. Ferrell, and J.F. Franklin. 1990. Effects on carbon storage of conversion of old-growth forests to young forests. Science 247(4943):699. 10.1126/science.247.4943.699.

Harmon, M.E., C.W. Woodall, B. Fasth, J. Sexton, and M. Yatkov. 2011. Differences between standing and downed dead tree wood density reduction factors: A comparison across decay classes and tree species. Res. Paper. NRS-15. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA. 40 pp.

Harriss, R.C., D.I. Sebacher, and F.P. Day. 1982. Methane flux in the Great Dismal Swamp. Nature 297(5868):673-674. 10.1038/297673a0.

Hashimoto, S., N. Carvalhais, A. Ito, M. Migliavacca, K. Nishina, and M. Reichstein. 2015. Global spatiotemporal distribution of soil respiration modeled using a global database. Biogeosciences 12:4121.

Heimann, M. and M. Reichstein. 2008. Terrestrial ecosystem carbon dynamics and climate feedbacks. Nature 451:289+.

Herrmann, M., R.G. Najjar, W.M. Kemp, R.B. Alexander, E.W. Boyer, W.-J. Cai, P.C. Griffith, K.D. Kroeger, S.L. McCallister, and R.A. Smith. 2015. Net ecosystem production and organic carbon balance of U.S. East Coast estuaries: A synthesis approach. Global Biogeochemical Cycles 29(1):96-111. 10.1002/2013gb004736.

Hirsch, P.R., A.J. Miller, and P.G. Dennis. 2013. Do root exudates exert more influence on rhizosphere bacterial community structure than other rhizodeposits? *In*: de Bruijn, F.J. (Ed.) Molecular Microbial Ecology of the Rhizosphere. John Wiley & Sons, Inc., Hoboken, NJ. pp. 229-242.

Holgerson, M.A. and P.A. Raymond. 2016. Large contribution to inland water CO_2 and CH_4 emissions from very small ponds. Nature Geoscience 9:222. 10.1038/ngeo2654.

Homer, C., J. Dewitz, Yang, L., S. Jin, P. Danielson, G. Xian, J. Coulston, N. Herold, J. Wickham, and K. Megown. 2015. Completion of the 2011 National Land Cover Database for the conterminous United States–representing a decade of land cover change information. Photogrammetric Engineering & Remote Sensing 81(5):345-354.

Hopkinson, C.S., W.-J. Cai, and X. Hu. 2012. Carbon sequestration in wetland dominated coastal systems—a global sink of rapidly diminishing magnitude. Current Opinion in Environmental Sustainability 4(2):186-194. https://doi.org/10.1016/j.cosust.2012.03.005.

Intergovernmental Panel on Climate Change (IPCC). 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventoires Programme. *In*: Eggleston, S., L. Buendia, K. Miwa, T. Ngara and K. Tanabe (Eds.). Institute for Global Environmental Strategies, Japan.

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate change 2007: the physical science basis. *In*: Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (Eds.). Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York

Intergovernmental Panel on Climate Change (IPCC). 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (Eds.). Cambridge University Press, Cambridge, UK, and New York, NY.

Janssens, I.A., W. Dieleman, S. Luyssaert, J.A. Subke, M. Reichstein, R. Ceulemans, P. Ciais, A.J. Dolman, J. Grace, G. Matteucci, D. Papale, S.L. Piao, E.D. Schulze, J. Tang, and B.E. Law. 2010. Reduction of forest soil respiration in response to nitrogen deposition. Nature Geoscience 3:315. 10.1038/ngeo844. https://www.nature.com/articles/ngeo844#supplementary-information.

Jenkins, J., D. Chojnacky, L. Heath, and R. Birdsey. 2003. National-scale biomass estimators for United States tree species. Forest Science 49(1):12.

Jobbágy, E.G. and R.B. Jackson. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10(2):423-436. 10.2307/2641104.

Johnson, D.W. and P.S. Curtis. 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management 140(2):227-238. https://doi.org/10.1016/S0378-1127(00)00282-6.

Jung, M., M. Reichstein, H.A. Margolis, A. Cescatti, A.D. Richardson, M.A. Arain, A. Arneth, C. Bernhofer, D. Bonal, J. Chen, D. Gianelle, N. Gobron, G. Kiely, W. Kutsch, G. Lasslop, B.E. Law, A. Lindroth, L. Merbold, L. Montagnani, E.J. Moors, D. Papale, M. Sottocornola, F. Vaccari, and C. Williams. 2011. Global patterns of land-atmosphere fluxes of carbon dioxide, latent heat, and sensible heat derived from eddy covariance, satellite, and meteorological observations. Journal of Geophysical Research: Biogeosciences 116(G3). 10.1029/2010jg001566.

Keeton, W.S., A.A. Whitman, G.C. McGee, and C.L. Goodale. 2011. Late-Successional Biomass Development in Northern Hardwood-Conifer Forests of the Northeastern United States. Forest Science 57(6):489-505. 10.1093/forestscience/57.6.489.

King, A.W., R.J. Andres, K.J. Davis, M. Hafer, D.J. Hayes, D.N. Huntzinger, B. de Jong, W.A. Kurz, A.D. McGuire, R. Vargas, Y. Wei, T.O. West, and C.W. Woodall. 2015. North America's net terrestrial CO_2 exchange with the atmosphere 1990–2009. Biogeosciences 12(2):399-414. 10.5194/bg-12-399-2015.

Kirschke, S., P. Bousquet, P. Ciais, M. Saunois, J.G. Canadell, E.J. Dlugokencky, P. Bergamaschi, D. Bergmann, D.R. Blake, L. Bruhwiler, P. Cameron-Smith, S. Castaldi, F. Chevallier, L. Feng, A. Fraser, M. Heimann, E.L. Hodson, S. Houweling, B. Josse, P.J. Fraser, P.B. Krummel, J.-F. Lamarque, R.L. Langenfelds, C. Le Quéré, V. Naik, S. O'Doherty, P.I. Palmer, I. Pison, D. Plummer, B. Poulter, R.G. Prinn, M. Rigby, B. Ringeval, M. Santini, M. Schmidt, D.T. Shindell, I.J. Simpson, R. Spahni, L.P. Steele, S.A. Strode, K. Sudo, S. Szopa, G.R. van der Werf, A. Voulgarakis, M. van Weele, R.F. Weiss, J.E. Williams, and G. Zeng. 2013. Three decades of global methane sources and sinks. Nature Geoscience 6:813. 10.1038/ngeo1955.

Kirwan, M.L. and J.P. Megonigal. 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504:53-60.

Kolka, R., C. Trettin, W. Tang, K.W. Krauss, S. Bansal, J.Z. Drexler, K.P. Wickland, R.A. Chimner, D.M. Hogan, E.J. Pindilli, B. Benscoter, B.A. Tangen, E.S. Kane, S.D. Bridgham, and C.J. Richardson. 2018. Chapter 13: Terrestrial wetlands. *In*: Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao and Z. Zhu (Eds.). Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report. U.S. Global Change Research Program, Washington, DC. pp. 507-567.

Kroeger, K.D., S. Crooks, S. Moseman-Valtierra, and J. Tang. 2017. Restoring tides to reduce methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention. Scientific Reports 7(1):11914. 10.1038/s41598-017-12138-4.

Lajtha, K., V.L. Bailey, K. McFarlane, K. Paustian, D. Bachelet, R. Abramoff, D. Angers, S.A. Billings, D. Cerkowniak, Y.G. Dialynas, A. Finzi, N.H.F. French, S. Frey, N.P. Gurwick, J. Harden, J.M.F. Johnson, K. Johnson, J. Lehmann, S. Liu, B. McConkey, U. Mishra, S. Ollinger, D. Paré, F.P. Pellat, D.d. Richter, S.M. Schaeffer, J. Schimel, C. Shaw, J. Tang, K. Todd-Brown, C. Trettin, M. Waldrop, T. Whitman, and K. Wickland. 2018. Soils. *In*: Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao and Z. Zhu (Eds.). Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report. U.S. Global Change Research Program, Washington, DC. pp. 469-506.

Lal, R. 2003. Soil erosion and the global carbon budget. Environment International 29(4):437-450. https://doi.org/10.1016/S0160-4120(02)00192-7.

Lane, R.R., S.K. Mack, J.W. Day, R.D. DeLaune, M.J. Madison, and P.R. Precht. 2016. Fate of soil organic carbon during wetland loss. Wetlands 36(6):1167-1181. 10.1007/s13157-016-0834-8.

Lark, T.J., J. Meghan Salmon, and H.K. Gibbs. 2015. Cropland expansion outpaces agricultural and biofuel policies in the United States. Environmental Research Letters 10(4):044003. 10.1088/1748-9326/10/4/044003.

Lawrence, G.B., T.C. McDonnell, T.J. Sullivan, M. Dovciak, S.W. Bailey, M.R. Antidormi, and M.R. Zarfos. 2018. Soil base saturation combines with beech bark disease to influence composition and structure of sugar maple-beech forests in an acid rain-impacted region. Ecosystems 21(4):795-810. 10.1007/s10021-017-0186-0.

Le Quéré, C., G.P. Peters, R.J. Andres, R.M. Andrew, T.A. Boden, P. Ciais, P. Friedlingstein, R.A. Houghton, G. Marland, R. Moriarty, S. Sitch, P. Tans, A. Arneth, A. Arvanitis, D.C.E. Bakker, L. Bopp, J.G. Canadell, L.P. Chini, S.C. Doney, A. Harper, I. Harris, J.I. House, A.K. Jain, S.D. Jones, E. Kato, R.F. Keeling, K. Klein Goldewijk, A. Körtzinger, C. Koven, N. Lefèvre, F. Maignan, A. Omar, T. Ono, G.H. Park, B. Pfeil, B. Poulter, M.R. Raupach, P. Regnier, C. Rödenbeck, S. Saito, J. Schwinger, J. Segschneider, B.D. Stocker, T. Takahashi, B. Tilbrook, S. van Heuven, N. Viovy, R. Wanninkhof, A. Wiltshire, and S. Zaehle. 2014. Global carbon budget 2013. Earth Syst. Sci. Data 6(1):235-263. 10.5194/essd-6-235-2014.

Liu, L. and T. Greaver. 2009. A review of nitrogen enrichment effects on three biogenic GHGs: The CO_2 sink may be largely offset by stimulated N_2O and CH_4 emission. Ecology Letters 12:1103-1117. 10.1111/j.1461-0248.2009.01351.x.

Liu, S., Y. Wei, W.M. Post, R.B. Cook, K. Schaefer, and M.M. Thornton. 2013. The Unified North American Soil Map and its implication on the soil organic carbon stock in North America. Biogeosciences 10(5):2915-2930. 10.5194/bg-10-2915-2013.

Loudermilk, E.L., R.M. Scheller, P.J. Weisberg, and A. Kretchun. 2017. Bending the carbon curve: fire management for carbon resilience under climate change. Landscape Ecology 32(7):1461-1472. 10.1007/s10980-016-0447-x.

Lucchese, M., J.M. Waddington, M. Poulin, R. Pouliot, L. Rochefort, and M. Strack. 2010. Organic matter accumulation in a restored peatland: Evaluating restoration success. Ecological Engineering 36(4):482-488. https://doi.org/10.1016/j.ecoleng.2009.11.017.

Malmsheimer, R.W., P. Heffernan, S. Brink, D. Crandall, F. Deneke, C. Galik, E. Gee, J.A. Helms, N. McClure, M. Mortimer, S. Ruddell, M. Smith, and J. Stewart. 2008. Forest management solutions for mitigating climate change in the United States. Journal of Forestry 106:115-171.

Manies, K.L., J.W. Harden, C.C. Fuller, and M.R. Turetsky. 2016. Decadal and long-term boreal soil carbon and nitrogen sequestration rates across a variety of ecosystems. Biogeosciences 13(15):4315. 10.5194/bg-13-4315-2016.

McKinley, D.C., M.G. Ryan, R.A. Birdsey, C.P. Giardina, M.E. Harmon, L.S. Heath, R.A. Houghton, R.B. Jackson, J.F. Morrison, B.C. Murray, D.E. Pataki, and K.E. Skog. 2011. A synthesis of current knowledge on forests and carbon storage in the United States. Ecological Applications 21(6):1902-1924. 10.1890/10-0697.1.

McLellan, E.L., K.G. Cassman, A.J. Eagle, P.B. Woodbury, S. Sela, C. Tonitto, R.D. Marjerison, and H.M. van Es. 2018. The nitrogen balancing act: tracking the environmental performance of food production. BioScience 68(3):194-203. 10.1093/biosci/bix164.

Mcleod, E., G.L. Chmura, S. Bouillon, R. Salm, M. Björk, C.M. Duarte, C.E. Lovelock, W.H. Schlesinger, and B.R. Silliman. 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. Frontiers in Ecology and the Environment 9(10):552-560. 10.1890/110004.

Melillo, J.M., S.D. Frey, K.M. DeAngelis, W.J. Werner, M.J. Bernard, F.P. Bowles, G. Pold, M.A. Knorr, and A.S. Grandy. 2017. Long-term pattern and magnitude of soil carbon feedback to the climate system in a warming world. Science 358(6359):101. 10.1126/science.aan2874.

Melillo, J.M., S. Butler, J. Johnson, J. Mohan, P. Steudler, H. Lux, E. Burrows, F. Bowles, R. Smith, L. Scott, C. Vario, T. Hill, A. Burton, Y.-M. Zhou, and J. Tang. 2011. Soil warming, carbon–nitrogen interactions, and forest carbon budgets. Proceedings of the National Academy of Sciences 108(23):9508. 10.1073/pnas.1018189108.

Miller, D.N., W.C. Ghiorse, and J.B. Yavitt. 1999. Seasonal patterns and controls on methane and carbon dioxide fluxes in forested swamp pools. Geomicrobiology Journal 16(4):325-331. 10.1080/014904599270578.

Morris, J.T., D.C. Barber, J.C. Callaway, R. Chambers, S.C. Hagen, C.S. Hopkinson, B.J. Johnson, P. Megonigal, S.C. Neubauer, T. Troxler, and C. Wigand. 2016. Contributions of organic and inorganic matter to sediment volume and accretion in tidal wetlands at steady state. Earth's Future 4(4):110-121. 10.1002/2015ef000334.

Moseman-Valtierra, S., R. Gonzalez, K.D. Kroeger, J. Tang, W.C. Chao, J. Crusius, J. Bratton, A. Green, and J. Shelton. 2011. Short-term nitrogen additions can shift a coastal wetland from a sink to a source of N2O. Atmospheric Environment 45(26):4390-4397. https://doi.org/10.1016/j.atmosenv.2011.05.046.

Moseman-Valtierra, S., O.I. Abdul-Aziz, J. Tang, K.S. Ishtiaq, K. Morkeski, J. Mora, R.K. Quinn, R.M. Martin, K. Egan, E.Q. Brannon, J. Carey, and K.D. Kroeger. 2016. Carbon dioxide fluxes reflect plant zonation and belowground biomass in a coastal marsh. Ecosphere 7(11):e01560. 10.1002/ecs2.1560.

MTBS Data Summaries. 2018. MTBS Data Access: Fire Level Geospatial Data. MTBS Project (USDA Forest Service/U.S. Geological Survey). http://mtbs.gov/direct-download.

Mulholland, P.J. and E.J. Kuenzler. 1979. Organic carbon export from upland and forested wetland watersheds1. Limnology and Oceanography 24(5):960-966. 10.4319/lo.1979.24.5.0960.

Nabuurs, G.-J., M. Lindner, P.J. Verkerk, K. Gunia, P. Deda, R. Michalak, and G. Grassi. 2013. First signs of carbon sink saturation in European forest biomass. Nature Climate Change 3:792. 10.1038/nclimate1853.

Najjar, R.G., M. Herrmann, R. Alexander, E.W. Boyer, D.J. Burdige, D. Butman, W.-J. Cai, E.A. Canuel, R.F. Chen, M.A.M. Friedrichs, R.A. Feagin, P.C. Griffith, A.L. Hinson, J.R. Holmquist, X. Hu, W.M. Kemp, K.D. Kroeger, A. Mannino, S.L. McCallister, W.R. McGillis, M.R. Mulholland, C.H. Pilskaln, J. Salisbury, S.R. Signorini, P. St-Laurent, H. Tian, M. Tzortziou, P. Vlahos, Z.A. Wang, and R.C. Zimmerman. 2018. Carbon Budget of Tidal Wetlands, Estuaries, and Shelf Waters of Eastern North America. Global Biogeochemical Cycles 32(3):389-416. 10.1002/2017gb005790.

Nave, L.E., C.W. Swanston, U. Mishra, and K.J. Nadelhoffer. 2013. Afforestationeffects on soil carbon storage in the United States: A synthesis. Soil Science Society of America Journal 77(3):1035-1047. 10.2136/sssaj2012.0236.

New York State Crop Reporting Service. 2008. Muckland Report, September 2008. 1 p.

New York State Department of Environmental Conservation (NYSDEC). 2010. Beyond Waste A Sustainable Materials Management Strategy for New York State. NYSDEC.

New York State Energy Research and Development Authority (NYSERDA). 2010. Renewable Fuels Roadmap and Sustainable Biomass Feedstock Supply for New York. Report submitted by Z. Wojnar, Project Manager, with multiple contributors. NYSERDA Report 10-05. NYSERDA, Albany, NY.

New York State Energy Research and Development Authority (NYSERDA). 2018. New York State Greenhouse Gas Inventory: 1990-2015. New York State Energy Research and Development Authority in cooperation with New York State Department of Environmental Conservation, Albany, NY.

Norby, R.J., E.H. DeLucia, B. Gielen, C. Calfapietra, C.P. Giardina, J.S. King, J. Ledford, H.R. McCarthy, D.J.P. Moore, R. Ceulemans, P. De Angelis, A.C. Finzi, D.F. Karnosky, M.E. Kubiske, M. Lukac, K.S. Pregitzer, G.E. Scarascia-Mugnozza, W.H. Schlesinger, and R. Oren. 2005. Forest response to elevated CO_2 is conserved across a broad range of productivity. Proceedings of the

National Academy of Sciences of the United States of America 102(50):18052. 10.1073/pnas.0509478102.

Nunery, J.S. and W.S. Keeton. 2010. Forest carbon storage in the northeastern United States: Net effects of harvesting frequency, post-harvest retention, and wood products. Forest Ecology and Management 259(8):1363-1375. https://doi.org/10.1016/j.foreco.2009.12.029.

Ogle, S.M., C.W. Woodall, A. Swan, J.E. Smith, and W. T. In preparation. Determining the managed land base for delineating carbon sources and sinks in the United States. Environmental Science and Policy.

Oldfield, E.E., S.A. Wood, C.A. Palm, and M.A. Bradford. 2015. How much SOM is needed for sustainable agriculture? Frontiers in Ecology and the Environment 13(10):527-527. 10.1890/1540-9295-13.10.527.

Olefeldt, D., M.R. Turetsky, P.M. Crill, and A.D. McGuire. 2013. Environmental and physical controls on northern terrestrial methane emissions across permafrost zones. Global Change Biology 19(2):589-603. 10.1111/gcb.12071.

Olson, D.M., T.J. Griffis, A. Noormets, R. Kolka, and J. Chen. 2013. Interannual, seasonal, and retrospective analysis of the methane and carbon dioxide budgets of a temperate peatland. Journal of Geophysical Research: Biogeosciences 118(1):226-238. 10.1002/jgrg.20031.

Orgiazzi, A., M.B. Dunbar, P. Panagos, G.A. de Groot, and P. Lemanceau. 2015. Soil biodiversity and DNA barcodes: opportunities and challenges. Soil Biology and Biochemistry 80:244-250. https://doi.org/10.1016/j.soilbio.2014.10.014.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2014. Forest Resources of the United States, 2012: A Technical Document Supporting the Forest Service 2010 Update of the RPA Assessment. Gen. Tech. Rep. WO-91. U.S. Department of Agriculture, Forest Service, Washington, DC. https://www.fs.usda.gov/treesearch/pubs/47322.

Pan, Y., R.A. Birdsey, J. Fang, R. Houghton, P.E. Kauppi, W.A. Kurz, O.L. Phillips, A. Shvidenko, S.L. Lewis, J.G. Canadell, P. Ciais, R.B. Jackson, S.W. Pacala, A.D. McGuire, S. Piao, A. Rautiainen, S. Sitch, and D. Hayes. 2011. A large and persistent carbon sink in the world's forests. Science 333(6045):988. 10.1126/science.1201609.

Pape, D., J. Lewandrowski, R. Steele, D. Man, M. Riley-Gilbert, K. Moffroid, and S. Kolansky. 2016. Managing Agricultural Land for Greenhouse Gas Mitigation within the United States. Report prepared by ICF International under USDA Contract No. AG-3144-D-14-0292.

Paustian, K., J. Lehmann, S. Ogle, D. Reay, G.P. Robertson, and P. Smith. 2016. Climate-smart soils. Nature 532:49-57.

Paustian, K., O. Andrén, H.H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, and P.L. Woomer. 1997. Agricultural soils as a sink to mitigate CO_2 emissions. Soil Use and Management 13(s4):230-244. 10.1111/j.1475-2743.1997.tb00594.x.

Pendleton, L., D.C. Donato, B.C. Murray, S. Crooks, W.A. Jenkins, S. Sifleet, C. Craft, J.W. Fourqurean, J.B. Kauffman, N. Marbà, P. Megonigal, E. Pidgeon, D. Herr, D. Gordon, and A. Baldera. 2012. Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLOS ONE 7(9):e43542. 10.1371/journal.pone.0043542.

Peylin, P., R.M. Law, K.R. Gurney, F. Chevallier, A.R. Jacobson, T. Maki, Y. Niwa, P.K. Patra, W. Peters, P.J. Rayner, C. Rödenbeck, I.T. van der Laan-Luijkx, and X. Zhang. 2013. Global atmospheric carbon budget: results from an ensemble of atmospheric CO_2 inversions. Biogeosciences 10(10):6699-6720. 10.5194/bg-10-6699-2013.

Piao, S., P. Ciais, P. Friedlingstein, N. de Noblet-Ducoudré, P. Cadule, N. Viovy, and T. Wang. 2009. Spatiotemporal patterns of terrestrial carbon cycle during the 20th century. Global Biogeochemical Cycles 23(4). 10.1029/2008gb003339.

Prentice, I.C., G.D. Farquhar, M.J.R. Fasham, M.L. Goulden, M. Heimann, V.J. Jaramillo, H.S. Kheshgi, C. Le Quere, R.J. Scholes, and D.W.R. Wallace. 2001. The carbon cycle and atmospheric carbon dioxide. *In*: Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell and C.A. Johnson (Eds.). Climate change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge.

Raile, G.K. 1982. Estimating Stump Volume. Res. Paper NC-224. USDA Forest Service, St. Paul, MN. 7 pp.

Ray, D.G., R.S. Seymour, N.A. Scott, and W.S. Keeton. 2009. Mitigating climate change with managed forests: balancing expectations, opportunity, and risk. Journal of Forestry 107(1):50-51. 10.1093/jof/107.1.50.

Raymond, P.A. and J.E. Bauer. 2001. Use of 14C and 13C natural abundances for evaluating riverine, estuarine, and coastal DOC and POC sources and cycling: a review and synthesis. Organic Geochemistry 32(4):469-485. https://doi.org/10.1016/S0146-6380(00)00190-X.

Reid, M.C., R. Tripathee, K.V.R. Schäfer, and P.R. Jaffé. 2013. Tidal marsh methane dynamics: Difference in seasonal lags in emissions driven by storage in vegetated versus unvegetated sediments. Journal of Geophysical Research: Biogeosciences 118(4):1802-1813. 10.1002/2013jg002438.

Ren, W., H. Tian, B. Tao, J. Yang, S. Pan, W.-J. Cai, S.E. Lohrenz, R. He, and C.S. Hopkinson. 2015. Large increase in dissolved inorganic carbon flux from the Mississippi River to Gulf of Mexico due to climatic and anthropogenic changes over the 21st century. Journal of Geophysical Research: Biogeosciences 120(4):724-736. 10.1002/2014jg002761.

Richardson, C.J., N. Flanagan, H. Wang, and M. Ho. 2014. Impacts of Peatland Ditching and Draining on Water Quality and Carbon Sequestration Benefits of Peatland Restoration. Eastern North Carolina/Southeastern Virginia Strategic Habitat Conservation Team, U.S. Fish and Wildlife Service, Region 4 and The Nature Conservancy North Carolina Chapter.

Rosenzweig, C., W. Solecki, A. DeGaetano, M. O'Grady, S. Hassol, and P. Grabhorn (Eds.). 2011. Responding to Climate Change in New York State: the ClimAID Integrated Assessment for Effective Climate Change Adaptation in New York State. Report 11-15. New York State Energy Research and Development Authority, Albany, NY.

Ruddell, S., R. Sampson, M. Smith, R. Giffen, J. Cathcart, J. Hagan, D. Sosland, J. Godbee, J. Heissenbuttel, S. Lovett, J. Helms, W. Price, and R. Simpson. 2007. The role for sustainably managed forests in climate change mitigation. Journal of Forestry 105(6):314-319. 10.1093/jof/105.6.314.

Saunois, M., P. Bousquet, B. Poulter, A. Peregon, P. Ciais, J. Canadell, E. Dlugokencky, G. Etiope, D. Bastviken, S. Houweling, G. Janssens - Maenhout, F. Tubiello, S. Castaldi, R. Jackson, M. Alexe, V. Arora, D. Beerling, P. Bergamaschi, D. Blake, G. Brailsford, V. Brovkin, L. Bruhwiler, C. Crevoisier, P. Crill, K. Covey, C. Curry, C. Frankenberg, N. Gedney, L. Höglund - Isaksson, M. Ishizawa, A. Ito, F. Joos, H.S. Kim, T. Kleinen, P. Krummel, J.F. Lamarque, R. Langenfelds, R. Locatelli, T. Machida, S. Maksyutov, K. McDonald, J. Marshall, Melton, Jr., I. Morino, V. Naik, Amp, Apos, S. Doherty, F.J. Parmentier, P. Patra, C. Peng, S. Peng, G. Peters, I. Pison, C. Prigent, R. Prinn, M. Ramonet, W. Riley, M. Saito, M. Santini, R. Schroeder, I. Simpson, R. Spahni, P. Steele, A. Takizawa, B. Thornton, H. Tian, Y. Tohjima, N. Viovy, A. Voulgarakis, M. Van Weele, G. Van Der Werf, R. Weiss, C. Wiedinmyer, D. Wilton, A. Wiltshire, D. Worthy, D. Wunch, X. Xu, Y. Yoshida, B. Zhang, Z. Zhang, and Q. Zhu. 2016. The global methane budget 2000–2012. Earth Syst. Sci. Data 8:697-751.

Schäfer, K.V.R., R. Tripathee, F. Artigas, T.H. Morin, and G. Bohrer. 2014. Carbon dioxide fluxes of an urban tidal marsh in the Hudson-Raritan estuary. Journal of Geophysical Research: Biogeosciences 119(11):2065-2081. 10.1002/2014jg002703.

Schultz, C. 2011. Lakes an important source of carbon emissions. Eos, Transactions American Geophysical Union 92(45):408-408. 10.1029/2011eo450014.

Sela, S., P.B. Woodbury, and H.M. van Es. 2018. Dynamic model-based N management reduces surplus nitrogen and improves the environmental performance of corn production. Environmental Research Letters 13(5):054010. 10.1088/1748-9326/aab908.

Seymour, R.S., A.S. White, and P.G. deMaynadier. 2002. Natural disturbance regimes in northeastern North America—evaluating silvicultural systems using natural scales and frequencies. Forest Ecology and Management 155(1):357-367. https://doi.org/10.1016/S0378-1127(01)00572-2.

Shirer, R. and C. Zimmerman. 2010. Forest Regeneration in New York State. The Nature Conservancy, New York. 25 pp.

Sitch, S., P. Friedlingstein, N. Gruber, S.D. Jones, G. Murray-Tortarolo, A. Ahlström, S.C. Doney, H. Graven, C. Heinze, C. Huntingford, S. Levis, P.E. Levy, M. Lomas, B. Poulter, N. Viovy, S. Zaehle, N. Zeng, A. Arneth, G. Bonan, L. Bopp, J.G. Canadell, F. Chevallier, P. Ciais, R. Ellis, M. Gloor, P. Peylin, S.L. Piao, C. Le Quéré, B. Smith, Z. Zhu, and R. Myneni. 2015. Recent trends and drivers of regional sources and sinks of carbon dioxide. Biogeosciences 12(3):653-679. 10.5194/bg-12-653-2015.

Smith, J.E., L.S. Heath, and J.C. Jenkins. 2003. Forest Volume-to-Biomass Models and Estimates of Mass for Live and Standing Dead Trees of U.S. Forests. Gen. Tech. Rep. NE-298. U.S. Department of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, PA.

Smith, J.E., L.S. Heath, K.E. Skog, and R.A. Birdsey. 2006. Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. General Technical Report NE-343. USDA Forest Service, Northeastern Research Station. 216 pp.

Smith, P. 2008. Land use change and soil organic carbon dynamics. Nutrient Cycling in Agroecosystems 81(2):169-178. 10.1007/s10705-007-9138-y.

Stanley, E.H., N.J. Casson, S.T. Christel, J.T. Crawford, L.C. Loken, and S.K. Oliver. 2016. The ecology of methane in streams and rivers: patterns, controls, and global significance. Ecological Monographs 86(2):146-171. 10.1890/15-1027.

Strachan, I.B., K.A. Nugent, S. Crombie, and M.-C. Bonneville. 2015. Carbon dioxide and methane exchange at a cool-temperate freshwater marsh. Environmental Research Letters 10(6):065006. 10.1088/1748-9326/10/6/065006.

Sullivan, T.J. 2015. Air Pollutant Deposition and Its Effects on Natural Resources in New York State. Cornell University Press, Ithaca, NY. 307 pp.

Sullivan, T.J., G.B. Lawrence, S.W. Bailey, T.C. McDonnell, C.M. Beier, K.C. Weathers, G.T. McPherson, and D.A. Bishop. 2013. Effects of acidic deposition and soil acidification on sugar maple in the Adirondack Mountains, New York. Environmental Science & Technology 47:12687-12694. 10.1021/es401864w.

Tang, J., L. Misson, A. Gershenson, W. Cheng, and A.H. Goldstein. 2005. Continuous measurements of soil respiration with and without roots in a ponderosa pine plantation in the Sierra Nevada Mountains. Agricultural and Forest Meteorology 132(3):212-227. https://doi.org/10.1016/j.agrformet.2005.07.011.

Tate, K.R. 2015. Soil methane oxidation and land-use change – from process to mitigation. Soil Biology and Biochemistry 80:260-272. https://doi.org/10.1016/j.soilbio.2014.10.010.

Terrer, C., S. Vicca, B.A. Hungate, R.P. Phillips, and I.C. Prentice. 2016. Mycorrhizal association as a primary control of the CO_2 fertilization effect. Science 353(6294):72. 10.1126/science.aaf4610.

Thomas, R.Q., C.D. Canham, K.C. Weathers, and C.L. Goodale. 2010. Increased tree carbon storage in response to nitrogen deposition in the US. Nature Geoscience 3:13-17.

Tian, X., B. Sohngen, J.B. Kim, S. Ohrel, and J. Cole. 2016. Global climate change impacts on forests and markets. Environmental Research Letters 11(3):035011. 10.1088/1748-9326/11/3/035011.

Tranvik, L.J., J.A. Downing, J.B. Cotner, S.A. Loiselle, R.G. Striegl, T.J. Ballatore, P. Dillon, K. Finlay, K. Fortino, L.B. Knoll, P.L. Kortelainen, T. Kutser, S. Larsen, I. Laurion, D.M. Leech, S.L. McCallister, D.M. McKnight, J.M. Melack, E. Overholt, J.A. Porter, Y. Prairie, W.H. Renwick, F. Roland, B.S. Sherman, D.W. Schindler, S. Sobek, A. Tremblay, M.J. Vanni, A.M. Verschoor, E. von Wachenfeldt, and G.A. Weyhenmeyer. 2009. Lakes and reservoirs as regulators of carbon cycling and climate. Limnology and Oceanography 54(6part2):2298-2314. 10.4319/lo.2009.54.6_part_2.2298.

Trettin, C., J. Drexler, R. Kolka, S. Bridgham, S. Bansal, B. Tangen, B. Bescoter, W. Tang, and S. Campbell. 2018. Appendix 13-B. Terrestrial Wetland– Atmosphere Exchange of Carbon Dioxide and Methane. *In*: Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao and Z. Zhu (Eds.). Second State of the Carbon Cycle Report. U.S. Global Change Research Program, Washington, DC. pp. 557-567.

Turetsky, M.R., E.S. Kane, J.W. Harden, R.D. Ottmar, K.L. Manies, E. Hoy, and E.S. Kasischke. 2010. Recent acceleration of biomass burning and carbon losses in Alaskan forests and peatlands. Nature Geoscience 4:27. 10.1038/ngeo1027.

U.S. Department of State. 2016. Second Biennial Report of the United States of America under the United Nations Framework Convention on Climate Change. U.S. Department of State. http://unfccc.int/national_reports/biennial_reports_and_iar/submitted_biennial_reports/items/7550.php.

U.S. Environmental Protection Agency. 2015. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2013. EPA 430-R-15-004. U.S. Environmental Protection Agency. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2013.

U.S. Environmental Protection Agency. 2016a. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2014. EPA 430-R-16-002. U. S. Environmental Protection Agency, Office of Atmospheric Programs, Washington, DC. https://www.epa.gov/sites/production/files/2017-04/documents/us-ghg-inventory-2016-main-text.pdf.

U.S. Environmental Protection Agency. 2016b. National Wetland Condition Assessment 2011: A Collaborative Survey of the Nation's Wetlands. EPA-843-R-15-005. U.S. Environmental Protection Agency, Office of Wetlands, Oceans and Watersheds, Office of Research and Development, Washington, DC. https://www.epa.gov/national-aquatic-resource-surveys/nwca.

U.S. Environmental Protection Agency. 2017. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2015. EPA 430-P-17-001. U. S. Environmental Protection Agency. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2015.

U.S. Environmental Protection Agency. 2018. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2016. EPA 430-P-18-003. U.S. EPA. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2016.

U.S. Environmental Protection Agency. 2019. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2017. EPA 430-R-19-001. U.S. Environmental Protection Agency. https://www.epa.gov/sites/production/files/2019-04/documents/us-ghg-inventory-2019-main-text.pdf.

U.S. Fish and Wildlife Service. 2011. Status and Trends of Wetlands in the Conterminous United States 2004 to 2009. Report to Congress. https://www.fws.gov/wetlands/status-and-trends/.

U.S. Fish and Wildlife Service. 2019. National Wetlands Inventory website. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Available at: http://www.fws.gov/wetlands/].

USDA Forest Service. 2018a. Forest Inventory and Analysis National Program: Program Features. U.S. Department of Agriculture Forest Service, Washington, D.C. Available at: http://fia.fs.fed.us/program-features/].

USDA Forest Service. 2018b. Forest Inventory and Analysis National Program, FIA library: Database Documentation. U.S. Department of Agriculture, Forest Service, Washington Office. Available at: http://fia.fs.fed.us/library/database-documentation/].

USDA Forest Service. 2018c. Forest Inventory and Analysis National Program: FIA Data Mart. U.S. Department of Agriculture Forest Service, Washington, D.C. Available at: http://apps.fs.fed.us/fiadb-downloads/datamart.html].

USDA National Agricultural Statistics Service (USDA-NASS). 2019. Quickstat (online database). Available at: https://www.nass.usda.gov/Quick_Stats/CDQT/chapter/2/table/11/state/NY [accessed May, 2019].

van Groenigen, K.J., X. Qi, C.W. Osenberg, Y. Luo, and B.A. Hungate. 2014. Faster decomposition under increased atmospheric CO_2 limits soil carbon storage. Science 344(6183):508. 10.1126/science.1249534.

Veltman, K., C.A. Rotz, L. Chase, J. Cooper, P. Ingraham, R.C. Izaurralde, C.D. Jones, R. Gaillard, R.A. Larson, M. Ruark, W. Salas, G. Thoma, and O. Jolliet. 2018. A quantitative assessment of Beneficial Management Practices to reduce carbon and reactive nitrogen footprints and phosphorus losses on dairy farms in the US Great Lakes region. Agricultural Systems 166:10-25. https://doi.org/10.1016/j.agsy.2018.07.005.

Waddington, J.M., P.J. Morris, N. Kettridge, G. Granath, D.K. Thompson, and P.A. Moore. 2015. Hydrological feedbacks in northern peatlands. Ecohydrology 8(1):113-127. 10.1002/eco.1493.

Wang, Y.P., J. Jiang, B. Chen-Charpentier, F.B. Agusto, A. Hastings, F. Hoffman, M. Rasmussen, M.J. Smith, K. Todd-Brown, Y. Wang, X. Xu, and Y.Q. Luo. 2016. Responses of two nonlinear microbial models to warming and increased carbon input. Biogeosciences 13(4):887-902. 10.5194/bg-13-887-2016.

Ward, N.D., T.S. Bianchi, P.M. Medeiros, M. Seidel, J.E. Richey, R.G. Keil, and H.O. Sawakuchi. 2017. Where carbon goes when water flows: carbon cycling across the aquatic continuum. Frontiers in Marine Science 4(7). 10.3389/fmars.2017.00007.

Wear, D.N. and J.W. Coulston. 2015. From sink to source: Regional variation in U.S. forest carbon futures. Scientific Reports 5:16518. 10.1038/srep16518.

Werner, C., K. Davis, P. Bakwin, C. Yi, D. Hurst, and L. Lock. 2003. Regional-scale measurements of CH_4 exchange from a tall tower over a mixed temperate/boreal lowland and wetland forest. Global Change Biology 9(9):1251-1261. 10.1046/j.1365-2486.2003.00670.x.

Weston, N.B., S.C. Neubauer, D.J. Velinsky, and M.A. Vile. 2014. Net ecosystem carbon exchange and the greenhouse gas balance of tidal marshes along an estuarine salinity gradient. Biogeochemistry 120:163-189.

Wightman, J.L. and P.B. Woodbury. 2016. New York dairy manure management greenhouse gas emissions and mitigation costs (1992-2022). J Environ Qual 45(1):266. 10.2134/jeq2014.06.0269.

Wightman, J.L., J.M. Duxbury, and P.B. Woodbury. 2015a. Land quality and management practices strongly affect greenhouse gas emissions of bioenergy feedstocks. BioEnergy Research 8(4):1681-1690. 10.1007/s12155-015-9620-3.

Wightman, J.L., Z.U. Ahmed, T.A. Volk, P.J. Castellano, C.J. Peters, S.D. DeGloria, J.M. Duxbury, and P.B. Woodbury. 2015b. Assessing sustainable bioenergy feedstock production potential by integrated geospatial analysis of land use and land quality. BioEnergy Research 8(4):1671-1680. 10.1007/s12155-015-9618-x.

Wik, M., R.K. Varner, K.W. Anthony, S. MacIntyre, and D. Bastviken. 2016. Climate-sensitive northern lakes and ponds are critical components of methane release. Nature Geoscience 9:99. 10.1038/ngeo2578.

Williams, C.A., G.J. Collatz, J. Masek, and S.N. Goward. 2012. Carbon consequences of forest disturbance and recovery across the conterminous United States. Global Biogeochemical Cycles 26(1). 10.1029/2010gb003947.

Williams, C.A., H. Gu, R. MacLean, J.G. Masek, and G.J. Collatz. 2016. Disturbance and the carbon balance of US forests: A quantitative review of impacts from harvests, fires, insects, and droughts. Global and Planetary Change 143:66-80. https://doi.org/10.1016/j.gloplacha.2016.06.002.

Wilson, J.O., P.M. Crill, K.B. Bartlett, D.I. Sebacher, R.C. Harriss, and R.L. Sass. 1989. Seasonal variation of methane emissions from a temperate swamp. Biogeochemistry 8(1):55-71.

Windham-Myers, L., W.-J. Cai, S.R. Alin, A. Andersson, J. Crosswell, K.H. Dunton, J.M. Hernandez-Ayon, M. Herrmann, A.L. Hinson, C.S. Hopkinson, J. Howard, X. Hu, S.H. Knox, K. Kroeger, D. Lagomasino, P. Megonigal, R.G. Najjar, M.-L. Paulsen, D. Peteet, E. Pidgeon, K.V.R. Schäfer, M. Tzortziou, Z.A. Wang, and E.B. Watson. 2018. Tidal wetlands and estuaries. *In*: Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao and Z. Zhu (Eds.). Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report. U.S. Global Change Research Program, Washington, DC. pp. 596-648.

Wolfe, D.W., A.T. DeGaetano, G.M. Peck, M. Carey, L.H. Ziska, J. Lea-Cox, A.R. Kemanian, M.P. Hoffmann, and D.Y. Hollinger. 2018. Unique challenges and opportunities for northeastern US crop production in a changing climate. Climatic Change 146(1):231-245. 10.1007/s10584-017-2109-7.

Woodall, C.W., L.S. Heath, G.M. Domke, and M.C. Nichols. 2011. Methods and Equations for Estimating Aboveground Volume, Biomass, and Carbon for Trees in the U.S. Forest Inventory,2010. Gen. Tech. Rep. NRS-88. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA. https://doi.org/10.2737/NRS-GTR-88.

Woodall, C.W., J.W. Coulston, G.M. Domke, B.F. Walters, D.N. Wear, J.E. Smith, H.-E. Andersen, B.J. Clough, W.B. Cohen, D.M. Griffith, S.C. Hagen, I.S. Hanou, M.C. Nichols, C.H. Perry, M.B. Russell, J.A. Westfall, and B.T. Wilson. 2015. The U.S. Forest Carbon Accounting Framework: Stocks and Stock Change, 1990-2016. Gen. Tech. Rep. NRS-154. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA. https://doi.org/10.2737/NRS-GTR-154.

Woodbury, P., T. Volk, R.H. Germain, P. Castellano, T. Buchholz, J. Wightman, J. MeIkonian, H. Mayton, Z. Ahmed, and C. Peters. 2010. Appendix E: Analysis of sustainable feedstock production potential in New York State *In*: Wojnar, Z. (Ed.) Renewable fuels roadmap and sustainable biomass feedstock supply for New York. New York State Energy Research and Development Authority, Albany, NY.

Woodbury, P.B. 2018. Agriculture can mitigate climate change at low cost to help meet Paris Climate Agreement goals. BioScience 68(7):485-486. 10.1093/biosci/biy053.

Woodbury, P.B., L.S. Heath, and J.E. Smith. 2007. Effects of land use change on soil carbon cycling in the conterminous United States from 1900 to 2050. Global Biogeochemical Cycles 21(3). 10.1029/2007gb002950.

Yan, Z., C. Liu, K.E. Todd-Brown, Y. Liu, B. Bond-Lamberty, and V.L. Bailey. 2016. Pore-scale investigation on the response of heterotrophic respiration to moisture conditions in heterogeneous soils. Biogeochemistry 131(1):121-134. 10.1007/s10533-016-0270-0.

Yavitt, J.B., J.A. Simmons, and T.J. Fahey. 1993. Methane fluxes in a northern hardwood forest ecosystem in relation to acid precipitation. Chemosphere 26(1):721-730. 10.1016/0045-6535(93)90456-F.

Yavitt, J.B., C.J. Williams, and R.K. Wieder. 1997. Production of methane and carbon dioxide in peatland ecosystems across North America: Effects of temperature, aeration, and organic chemistry of peat. Geomicrobiology Journal 14(4):299-316. 10.1080/01490459709378054.

Zhang, F., J.M. Chen, Y. Pan, R.A. Birdsey, S. Shen, W. Ju, and L. He. 2012. Attributing carbon changes in conterminous U.S. forests to disturbance and non-disturbance factors from 1901 to 2010. Journal of Geophysical Research: Biogeosciences 117(G2). 10.1029/2011jg001930.

Endnotes

1 http://nadp.slh.wisc.edu/

2 https://iscn.fluxdata.org/

3 https://nature4climate.org/u-s-carbon-mapper/

4 https://www.usda.gov/foodlossandwaste; https://www.epa.gov/sustainable-management-food/united-states-2030-food-loss-and-waste-reduction-goal#goal

5 https://nature4climate.org/u-s-carbon-mapper/

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