Agriculture and Forestry

I. Introduction

The contributions of the Agriculture and Forestry sectors to greenhouse gas ("GHG") emissions are often overlooked in the discussion on climate change. Agricultural activities and forest management for commercial products are both major sources of greenhouse gases, and state governments have numerous tools at their disposal to transform these sectors to help mitigate catastrophic climate change.

The Agriculture and Forestry sectors provide unique opportunities for climate change mitigation because they impact GHG sources *and* sinks. New York's 18.6 million acres of forest play a critical role in sequestering carbon, supporting biodiversity, and providing numerous other ecosystem services.[1] The conservation and restoration of the state's existing forests, as well as reforestation of previously forested areas, are critical to achieving New York's climate targets. However, many of the strategies in the Draft Scoping Plan ("DSP") related to forestry seek to protect the profitability of the forestry industry rather than maximizing climate benefits. The

[1] N.Y. Climate Action Council, *Draft Scoping Plan* ("DSP"), 194 (2021), https://climate.ny.gov/-/media/Project/Climate/Files/Draft-Scoping-Plan.pdf.

Final Scoping Plan ("FSP") should ensure that pressures from the forestry industry to harvest and generate forest crops do not threaten the preservation of New York's forests. Additionally, the FSP should not offer New York's forests as an excuse for delaying action on reducing fossil fuel emissions through offset programs, which are scientifically unsound and strongly opposed by many environmental justice groups.

The State Department of Environmental Conservation ("DEC") indicates that agriculture is responsible for 6% of total state GHG emissions, and that 92% of those emissions come from livestock.[2] Unlike other sectors in New York where emissions have already decreased, livestock management emissions have increased 44% since 1990.[3] And unlike the energy sector, whose contributions to climate change are largely in the form of carbon dioxide, agricultural emissions include methane, nitrous oxide, and carbon dioxide. Over 20 years, methane has a global warming potential about 84 times greater than carbon dioxide, and nitrous oxide has a global warming potential about 264 times greater than carbon dioxide.[4] Despite the impact of these emissions, the DSP fails to include any mandatory strategies to regulate methane emissions from livestock. The FSP should include greater accountability and transparency across all strategies related to reducing emissions from livestock and croplands and strategies related to increasing soil carbon storage.[5]

The DSP acknowledges the significance of the Agriculture and Forestry sectors to climate change mitigation and proposes many effective approaches to curbing their greenhouse gas contributions. However, the urgency of achieving the Climate Leadership and Community Protection Act ("CLCPA")'s ambitious goals calls for more transformative and creative approaches. Below, we provide recommendations to maximize the efficacy of the Plan's agriculture and forestry strategy.

II. Sustainable Forest Management

A. The Final Scoping Plan Should Prioritize Reforestation and Forest Preservation Efforts, Which Provide the Maximum Climate Benefit, Rather Than Promoting Strategies Designed to Profit the Forestry Industry

Several strategies under the Sustainable Forest Management section of the DSP are based on a mischaracterization of forest carbon cycling in New York. These strategies are designed to support the forestry industry rather than to maximize climate benefits. The FSP must revisit these assumptions and only make recommendations based on accurate climate impact accounting,

---

[2] N.Y. Dep't of Env't Conservation ("DEC"), *Agriculture Forestry, and Other Land Use: 2021 NYS Greenhouse Gas Emissions Report*, at 3, tbl. SR3.3, https://www.dec.ny.gov/docs/administration_pdf/ghgafolu21.pdf.

[3] *Id.*

[4] Intergovernmental Panel on Climate Change Working Groups I, II and III, *Climate Change 2014: Synthesis Report* 87 box 3.2 tbl.1 (2014), https://www.ipcc.ch/site/assets/uploads/2018/02/SYR_AR5_FINAL_full.pdf; *see also* Eastern Rsch. Grp. Inc., *Technical Documentation: Estimating Energy Sector Greenhouse Gas Emission Under New York State's Climate Leadership and Community Protection Act* 65 app. E (2021), https://www.dec.ny.gov/docs/administration_pdf/energyghgerg.pdf.

[5] *See generally* Peter H. Lehner & Nathan A. Rosenberg, *Advancing Climate-Neutral Agriculture in New York, Viewpoint*, 33 Env't Law in N.Y. (2022) [attached as Exhibit A].

rather than relying on biased accounting promoted by industry to suggest that harvesting provides a climate benefit.

The DSP claims that "[t]o maximize New York forests carbon sequestration potential, it is critical that forest management activities *increase* statewide," because the "carbon sequestration rate has slowed" in New York's forests.[6] This flawed framing is used to justify removals from forests, despite clear scientific evidence that allowing New York forests to remain intact will generally provide the maximum climate benefit. There are several reasons why this is so.

First, most forest stands in New York are predicted to have positive growth increments for several decades absent accelerated harvesting intensities. The majority of forest stands in the northeast are relatively young and are dominated by growth following the abandonment of agricultural fields in the region in the mid-1800s.[7] The mean age of forest stands in New York is between 60–70 years old, with most forest stands comprised of younger trees.[8] This transition from agricultural activities has allowed northeastern forests to play a unique, ongoing role in mitigating climate change. While global anthropogenic activities have dramatically increased atmospheric carbon dioxide concentrations, northeastern forests continue to help counteract these emissions by sequestering more than a megaton of carbon per hectare annually through photosynthesis.[9] Forest stands with trees between 70–100 years hold the greatest densities of carbon in the state, and these older stands also continue to sequester significant quantities of carbon.[10] Protected from harvest, New York forests have the potential to continue to sequester carbon at increasing or stable rates for several decades. However, harvesting reduces the capacity of these forests to continue functioning as a carbon sink.

Second, losses in carbon stocks following harvest are not compensated by new growth in timescales relevant to New York state's climate action planning. In northeastern forests, it takes several decades to recover from the debt of carbon removals following harvest to arrive back at pre-harvest carbon stocks.

Third, this period of regrowth represents a lost opportunity for existing forest growth to continue to accrue carbon, as would have occurred in the absence of disturbance. The continued

---

[6] DSP at 198–199 (emphasis added).
[7] *See* Jana E. Compton & Richard D. Boone, *Long-term Impacts of Agriculture on Soil Carbon and Nitrogen in New England Forests*, 81 Ecology 8 (2000) [attached as Exhibit B]; *see also* Charles V. Cogbill et al., *The Forests of Presettlement New England, USA: Spatial and Compositional Patterns Based on Town Proprietor Surveys*, 29 J. Biogeography 1279 (2002) [attached as Exhibit C].
[8] *See* Yude Pan et al., *Age Structure and Disturbance Legacy of North American Forests*, 8 Biogeosciences 715 (2011); *see also* Richard H. Widmann et al., U.S. Dep't of Agric., *New York Forests*, at 97, fig.70 (2012), https://www.fs.fed.us/nrs/pubs/rb/rb_nrs98.pdf.
[9] *See* Xiaoliang Lu et al., *Land Carbon Sequestration within the Conterminous United States: Regional- and State-Level Analyses*, 120 J. Geophysical Rsch.; Biogeosciences 379 (2015); *see also* Thomas Buchholz et al., Cary Institute of Ecosystem Studies, *Forest Biomass and Bioenergy: Opportunities and Constraints in the Northeastern United States* (2011), https://www.caryinstitute.org/sites/default/files/public/downloads/report_biomass.pdf.
[10] *See Forest Inventory & One Click Factsheet*, U.S. Dep't of Agric. ("USDA"), https://public.tableau.com/views/FIA_OneClick_V1_2/StateSelection?:showVizHome=no; *see also* Forest Res. Ass'n. Forest Carbon Report: New York (2021), https://live-forest-resources-association.pantheonsite.io/wp-content/uploads/2021/12/New-York.pdf.

harvesting of these forests as they mature not only reduces stored carbon but also eliminates the sequestration that continued growth would otherwise provide. Accounting for this opportunity cost is often left out of assessments on the sustainability of bioenergy harvesting and other evaluations of forest management planning.

Fourth, harvesting these forests, including the removal of biomass for bioenergy, leads to additional emissions from harvesting activities, burning, transportation, and manufacture of wood products.[11]

The forest industry's claims and the strategies in this section of the DSP wholly overlook these critical facts. Harvesting biomass sets the clock back on carbon sequestration and weakens one of our strongest defenses against increasing atmospheric greenhouse gases. The FSP must recognize the fundamental benefits of leaving forests intact and carefully account for this potential for continued carbon sequestration in any forest management proposals that suggest harvesting as a climate mitigation strategy.

Despite their important functions, only 6% of forestland in the northeastern U.S. is legally preserved from harvest.[12] While logging efforts may not consume a large proportion of the landscape, these removals consume over 50% of *net* growth in New York State (i.e., the change in biomass that remains in undisturbed forests following natural causes of tree mortality), already significantly reducing the potential of these systems to sequester carbon.[13]

The Land Use chapter of the DSP provides a more accurate account of the climate benefits of allowing New York's forests to remain as forests than what is contained in the Sustainable Forest Management section:

> New York has 18.6 million acres of forests, which hold an estimated 1,911 MMT of carbon. In addition to carbon sequestration and storage, New York's forests provide wildlife habitat, forest products, flood mitigation, recreational opportunities, and mental health benefits, and protect the State's air and water quality. *Forestlands in many parts of the State are under pressure from development and forest conversion, which is causing a steady decline in the amount of CO2 being absorbed each year. Keeping forests as forests is critical to maintaining and increasing levels of carbon sequestration and storage and preventing emissions, as forests sequester and store much more carbon than any other land use in New York.* State and municipal land acquisition provide the most reliable long-term protection of forested areas from land conversion. There are currently 4.8 million acres of forestland owned by the State, local municipalities, or land trusts in New York. In 2020, 6,005 acres of land were protected through acquisition by DEC and OPRHP and 14 grants were awarded to protect forests

[11] *See* Tara W. Hudiburg et al., *Regional Carbon Dioxide Implications of Forest Bioenergy Production,* 1 Nature Climate Change 419 (2011) [attached as Exhibit D].

[12] Thomas Buchholz et al., Cary Institute of Ecosystem Studies, *Forest Biomass and Bioenergy: Opportunities and Constraints in the Northeastern United States* 14 (2011), https://www.caryinstitute.org/sites/default/files/public/downloads/report_biomass.pdf.

[13] Buchholz et al., *supra* note 12, at 19; *see also* Widmann et al., *supra* note 8.

> through the Conservation Partnership Program. *To maintain the State's carbon storage and sequestration levels, additional protection is needed, which can be accomplished through land acquisition and conservation easements.*[14]

Thus, the FSP should ensure recommendations related to forests are internally consistent and it should focus on strategies to incentivize forest conservation, protection, and afforestation and reforestation efforts as laid out in the Land Use chapter rather than conflicting incentives to manage forests for forest products as described in the Agriculture and Forestry chapter.

The FSP must accurately reflect the climate benefits of allowing New York's forests to remain intact and continue to sequester carbon as they age. As described in greater detail below, the Climate Action Council should re-evaluate the strategies currently within AF1-8 and eliminate those that incentivize removals from New York's forests, including, for example, tax breaks for the development of forest management plans to produce and harvest forest crops. The DSP offers potential remedies to level the playing field and encourage private landowners to keep forest land intact. However, the FSP must ensure that these new programs are at least equally attractive as existing harvesting incentives. Additionally, the FSP should eliminate recommendations that offer forest carbon sequestration as an opportunity to purchase offsets—rather than actually reduce—fossil fuel emissions. Finally, the FSP should include mechanisms for close oversight of any funding directed towards forest harvesting equipment.

B. The Final Scoping Plan Should Ensure That Benefits for Private Forest Landowners Who Manage for Carbon Sequestration or Conserve Their Forests in Natural Conditions are *At Least* Equal to Benefits for Private Forest Landowners Who Manage for Wood Products

The DSP includes recommendations for amending Real Property Tax Law 480a and enacting new legislation to include tax incentives for private forest landowners to manage for multiple benefits including wildlife habitat and carbon sequestration or to conserve their forests in natural conditions. These recommendations will help reduce the incentive in Real Property Tax Law 480a to harvest forests. However, the DSP states that "[i]nitial benefits" of these new tax incentives—which will be contained in a new section 480b—"should start at a lower level than 480a and 480c with up to 100% reimbursement to municipalities."[15] While these amendments will help incentivize landowners to keep forested land intact, the FSP should ensure that abatement rates for forest landowners managing their forests for wildlife habitat or carbon sequestration or conserving their forests in natural conditions are offered benefits *at least* equal to those available to forest landowners managing for wood products or other harvesting activities. Absent a level playing field for these outcomes with clear climate benefits, the FSP will not go far enough to protect New York forests from harvest.

---

[14] DSP at 276. (emphasis added).
[15] DSP at 204.

C. The Final Scoping Plan Should Not Include AF6, Which Relies on Dangerous and Ineffective Offsetting of Fossil Fuel Emissions Through Forest Carbon Sequestration

The FSP should not include AF6, which suggests that carbon sequestration in New York State forests may be used to offset emissions from other sectors. Forest carbon sequestration should not be used to allow fossil fuel emissions from other sectors to persist. Fossil fuel polluters should not be allowed to circumvent their responsibility to curb direct emissions by claiming to offset them by purchasing impermanent carbon gains elsewhere.

Such offset schemes seek to avoid accountability for direct emissions of greenhouse gases with uncertain, imprecise and difficult-to-monitor supposed increases in carbon stocks elsewhere. These offset schemes are premised on a scientific fallacy that equates increases in carbon stocks in soil and vegetation with past and ongoing losses of fossil carbon. However, these are not at all equivalent. It is critical to note that climate change is primarily attributed to the removal of large amounts of *fossil* carbon, which would have remained sequestered in the absence of anthropogenic activities. In contrast to these slow-cycling fossil stocks, carbon in biogenic pools including vegetation and soils in New York forests is inherently impermanent and perpetually vulnerable to decomposition. Thus, offsets should not be allowed to delay irreversible losses of fossil carbon. Carbon sequestration rates in New York state should be restored and accelerated (for example, through strategies to incentivize reforestation described in the Land Use chapter) in parallel with independent reductions in fossil fuel emissions.

The FSP should also take heed of the failures of past market-based approaches to regulating pollution that allow for offsets. As noted in our comments on economy-wide mechanisms and by the Climate Justice Working Group ("CJWG"), environmental justice communities have historically not benefited from—and indeed have often been harmed by—offset market-based policies though they are the most burdened by pollution-generating facilities. For example, one leading study found that California's cap-and-trade policy, which represents a market scheme that permits offsets, has exacerbated environmental injustice. An analysis of the program found that (1) regulated facilities were disproportionately sited in environmental justice neighborhoods, (2) most of the regulated facilities increased emissions of both GHGs and co-pollutants during the time period studied, and (3) neighborhoods that experienced increases in both annual average GHGs and annual average co-pollutants were more likely to be environmental justice neighborhoods.[16] This study also concluded that the use of offsets allowed regulated facilities to keep polluting (and degrading local air quality) by purchasing offsets from projects largely out-of-state that provided no benefit to frontline communities.[17] To avoid replicating these type of harms, the FSP must consider non-GHG co-pollutants and local environmental impacts to environmental justice communities and thus avoid offering New York forests as an opportunity to offset fossil fuel emissions.

There is simply no substitute for directly reducing fossil fuel emissions. Such reductions are critical to achieving climate targets as well as environmental justice goals as pollution

---

[16] *See* Lara Cushing et al., *Carbon Trading, Co-pollutants, and Environmental Equity: Evidence from California's Cap-and-Trade Program (2011–2015)*, 15 PLOS Med. e1002604 (2018).
[17] *See id*.

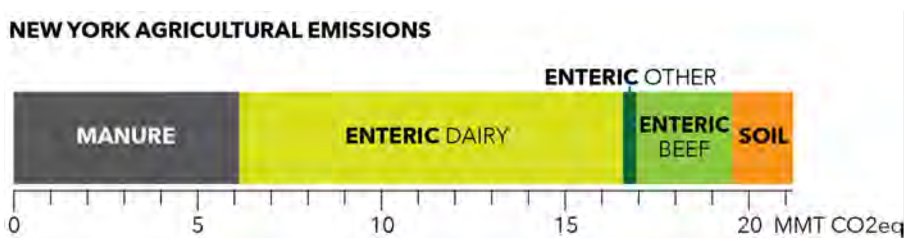
hotspots disproportionately burden low-income communities and communities of color. The FSP should not support accounting that allows avoidable ongoing fossil fuel emissions to persist based on offsets.

D. The Final Scoping Plan Should Require Close Oversight of Any Grants for Logging and Manufacturing Equipment to Ensure These Programs Do Not Inadvertently Support Increased Harvesting at the Expense of Conservation

Under AF3, the DSP recommends investments in logging and manufacturing equipment as a strategy to reduce site impacts associated with harvesting activities. While foresters should be required to adopt strategies to reduce site impacts, the FSP should ensure that funding such equipment does not further incentivize forest harvests over allowing New York forests to remain intact. If DEC provides foresters with funding for adopting such technology, it should include close oversight of grants to ensure forest management planning accurately accounts for the climate benefits of avoiding harvesting (as described above).

III. Livestock Management

New York ranks third for the number of milk cows in farms across the state, and is among the top five largest dairy-producing states in the country.[18] Its scale of production is associated with large, concentrated emissions of methane. Manure management and enteric fermentation from livestock account for 92% of New York's agricultural greenhouse gas emissions.[19] In 2019, manure management released over 6 million metric tons of CO2 equivalents (MMT CO2eq), and enteric fermentation released over 13 MMT CO2eq as methane (*see figure below*).[20]

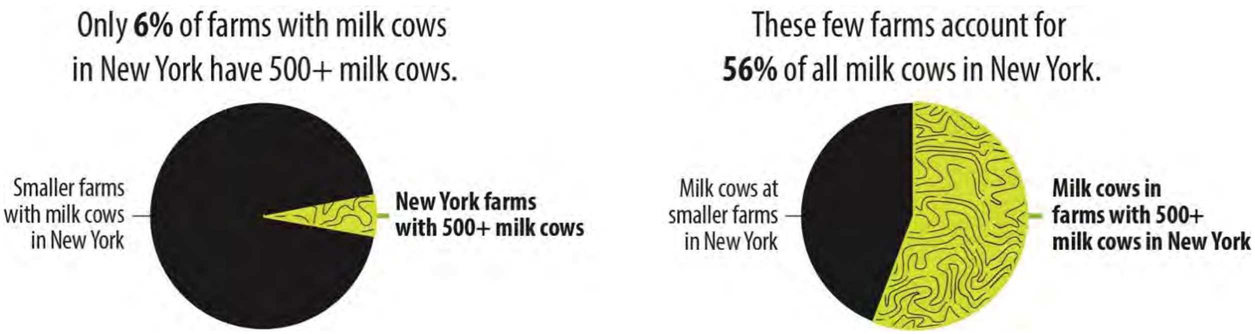


Livestock emissions in New York are heavily concentrated in the largest concentrated animal feeding operations (CAFOs). In 2017, out of over 4,600 dairy farms in New York, only 142 farms—3% of all dairy farms in New York—had herd sizes over 1,000 milk cows, and only an additional 141 farms had herd sizes between 500 and 999 milk cows.[21] Just 6% of New York

[18] USDA, Statistical Bull. No. 1055, *Milk Cows and Production Final Estimates 2013-2017* 7, 9 (2019), https://downloads.usda.library.cornell.edu/usda-esmis/files/cz30ps66x/jd473517g/bk128k88x/mcprsb19.pdf; *see also Farm Milk Production*, USDA, Econ. Rsch. Serv., https://www.ers.usda.gov/topics/animal-products/dairy/background/ (last updated Apr. 27, 2022).

[19] N.Y. DEC, *supra* note 2 at 3, tbl.SR3.3.
[20] *Id.*
[21] USDA, AC-17-A-32, *Census of Agriculture*, 23 tbl.17 (2019), https://www.nass.usda.gov/Publications/AgCensus/2017/Full_Report/Volume_1,_Chapter_1_State_Level/New_York/nyv1.pdf.

dairies account for 56% of New York's dairy cow population (*see figure below*), and are thus responsible for the majority of methane emissions from both enteric fermentation and manure management.[22] This provides an opportunity to tailor policies for reducing livestock methane based on farm size. New York's limited funding should be prioritized to support small-and mid-size farms in controlling their emissions. The largest producers, on the other hand, should be required to control their emissions without financial support (or with much lower financial support) from the state. Focusing climate mitigation efforts on these large operations should be a priority for reducing New York's total greenhouse gas emissions.



A. The Final Scoping Plan Should Include Regulatory Options, as Authorized Under the Environmental Conservation Law ("ECL") and Consistent with the CLCPA, for Reducing Methane Emissions

1. DEC Has a Mandate and Authority to Regulate Methane Emissions from New York's Largest CAFOs

Under New York law, "[i]t *shall* be the responsibility of the department, in accordance with such existing provisions and limitations as may be elsewhere set forth in law, by and through the commissioner to carry out the environmental policy of the state set forth in section 1-0101 of this chapter." ECL § 3-0301(1) (emphasis added). That environmental policy, in turn, is "to conserve, improve and protect [New York's] natural resources and environment and to prevent, abate and control water, land and air pollution, in order to enhance the health, safety and welfare of the people of the state and their overall economic and social well being." ECL § 1-0101(1). New York's laws specific to air pollution additionally mandate that DEC "require the use of all available practical and reasonable methods to prevent and control air pollution in the state of New York." ECL § 19-0103.

There is no question that methane is considered "air pollution" under the statute, as the term is broadly defined as:

> the presence in the outdoor atmosphere of one or more air contaminants in quantities, of characteristics and of a duration which are injurious to human, plant or animal life or to property or which unreasonably interfere with the comfortable

[22] *Id.*

enjoyment of life and property throughout the state or throughout such areas of the state as shall be affected thereby. . .

ECL § 19-0107(3). Given methane's tremendous global warming potential, and the New York legislature's finding in the CLCPA that "climate change is adversely affecting economic well-being, public health, natural resources, and the environment of New York," 2019 NY Senate-Assembly Bill S6599, A8429 § 1 ("CLCPA"), DEC is required by statute to abate this pollution. And under New York law, it must use "all available practical and reasonable methods to prevent and control" this air pollution—i.e., methane emissions—in the state. ECL § 19-0103.

DEC is also empowered to "[f]ormulate, adopt and promulgate, amend and repeal codes and rules and regulations for preventing, controlling or prohibiting air pollution in such areas of the state as shall or may be affected by air pollution," including requiring permits or certificates. ECL § 19-0301(1)(a). DEC is given explicit authority to "[i]nclude in any such codes and rules and regulations provisions establishing areas of the state and prescribing for such areas (1) the degree of air pollution or air contamination that may be permitted therein, [and] (2) the extent to which air contaminants may be emitted to the air by any air contamination source."[23] "Air contamination source" is defined as "any source at, from or by reason of which there is emitted into the atmosphere any air contaminant" and clearly encompasses livestock. ECL § 19-0107(5). DEC can thus use its authority under this provision to regulate methane emissions from CAFOs.

2. The CLCPA Does Not Limit DEC's Authority to Regulate Livestock Emissions

The CLCPA requires DEC to promulgate regulations "to ensure compliance with" the CLCPA's new greenhouse gas emission targets. CLCPA § 2 (amending ECL § 75-0109(1)). These regulations must "include legally enforceable emissions limits, performance standards, or measures or other requirements to control emissions from greenhouse gas emission sources, with the exception of agricultural emissions from livestock." *Id*. (amending ECL § 75-0109(2)(b)). This does not limit the authority DEC already had under the ECL to regulate methane emissions from livestock, for several reasons.

First, emissions that emanate from manure and grazing lands are not "from" the livestock, but rather are a result of how manure and grazing lands are managed by farmers and ranchers. (Emissions from municipal sewage treatment plants, similarly, are not considered to be "from humans".) Thus, the plain language makes clear the legislature's intent that DEC retain authority to promulgate legally enforceable emissions limits or performance standards relating to manure GHG emissions.

Second, the exception occurs in the paragraph imposing a mandate on DEC that it *must* regulate certain sources: "The regulations promulgated by [DEC] *shall*: . . . Include legally enforceable emissions limits, performance standards, or measures or other requirements to control emissions from greenhouse gas emissions sources, with the exception of agricultural

[23] ECL § 19-0301(1)(b); *see also* ECL §§ 3-0301(1)(a)–(b); §§ 3-0301 (2)(a), (m) (stating DEC's authority to issue rules and regulations to carry out state's general environmental policy).

emissions from livestock." CLCPA § 2 (amending ECL § 75-0109(2)(b)) (emphasis added). A close reading suggests that DEC *may* impose enforceable emission limits; it is only that the CLCPA does not *require* DEC to do so under the aegis of the CLCPA.

3. Regulating Methane Emissions from CAFOs Would Be Feasible and Come at Reasonable Cost to CAFOs

Not only *can* DEC regulate methane from CAFOs, but the costs of such regulation are reasonable and would be easily borne by the industry's largest operations. Several existing practices and mitigation strategies can curb these emissions at reasonable cost. CAFO operators can reduce methane generation by shifting more production to pasture-based systems or implementing dry manure management and greater solid/liquid separation at reasonable cost. As described below, these and other transformative shifts should be a priority in the FSP as they achieve greater emission reductions along with many other environmental and social co-benefits.

However, we recognize that at best it will take significant time to transition New York dairies from the current CAFOs structure. CAFOs with liquid manure management can currently adopt technology to cover existing lagoons and flare methane emissions. While cover and flare systems do not address the large share of enteric methane emissions upstream of manure production or emissions from land application of liquid manure, they are preferable to open liquid manure lagoons. Recent studies show this practice is cost-effective and financially feasible in the context of large New York dairies. For example, researchers at Cornell University found that these systems cost about $13 per megagram of carbon dioxide equivalent, or $0.005 per liter milk.[24] A separate Cornell University study of 128 farms in New York found that net farm income among the top 20% of dairies with an average of 1,515 cows was $1,112,949 (or $735/cow) in 2017.[25] This cost of adoption is similar to costs borne by producers in other sectors to mitigate greenhouse gas pollution.

B. The Final Scoping Plan Should Include More Transformative Strategies for Reducing Manure Outside of Cover and Flare Systems and Digesters, Including Strategies to Reduce Manure Generation and Reducing Wet Storage

The FSP should focus much more on reducing methane generation upstream of emissions, unlike the DSP's focus on methane destruction following production. This approach would be similar to the framework guiding waste management, where there is a primary preference for strategies leading to source reduction and reuse rather than simply treating produced waste.[26] While the DSP includes alternative manure management strategies in AF9, these strategies focus largely on end-of-lifecycle strategies to reduce emissions from manure

---

[24] Jennifer L. Wightman & Peter B. Woodbury, N*ew York Dairy Manure Management Greenhouse Gas Emissions and Mitigation Costs (1992-2022)*, 45 J. Env't Quality 1 (2016).

[25] John Karszes, Cornell Univ., EB 2018-08, *Six Year Trend Analysis New York State Dairy Farms Selected Financial and Production Factors* (2018), https://dyson.cornell.edu/wp-content/uploads/sites/5/2019/02/Cornell-Dyson-eb1808.pdf.

[26] *See Sustainable Materials Management: Non-Hazardous Materials and Waste Management Hierarchy*, EPA, https://www.epa.gov/smm/sustainable-materials-management-non-hazardous-materials-and-waste-management-hierarchy (last updated Dec. 15, 2021).

storage rather than more transformative strategies focused on reducing manure generation and accumulation in the first place. We are especially concerned that any further public investment in the largest CAFOs in the state (as opposed to enacting regulations limiting methane emissions as discussed above) will only further exacerbate consolidation, concentration, and harm to our rural communities.

While cover and flare systems and other strategies listed in AF9 can reduce methane emissions, the FSP should also recommend more transformative practices upstream of manure storage and incentivize practices that smaller producers can adopt. The first priority in manure management should be generating less methane to begin with. For example, using dry manure management and transitioning to managed-pasture-based and lower-density farming reduces the concentration and quantity of stored manure, and thus the generation of methane, while also improving soil health. Additionally, best practices during the spreading of manure—such as spreading only the amount that plants need and can use and avoiding spreading on frozen or saturated soils—can prevent unnecessary emissions.[27] These practices also have significant air and water quality co-benefits.

Relying on end-of-process systems also is less certain as engineered systems often fail, leak, or are operated sub-optimally. Digesters have been found to have leakage rates of about 3–6%,[28] which largely undercuts their climate benefits, and which can even cause them to be net sources of methane.[29] Furthermore, biodigesters release additional pollutants such as NOx, sulfur oxide, and particulate matter.[30] If the gas is then transported—through pipelines that also tend to have high leakage rates—the climate benefit is further reduced. These leaks not only increase the climate change impact, but they also endanger local communities. In addition, these systems do nothing to address enteric emissions or emissions from the land application of the liquid manure after digestion or flaring.

In allocating resources for these emissions reduction strategies, the FSP should prioritize financial support to small and mid-sized livestock operations, rather than the state's largest CAFOs. We suggest a cap on total funding awarded to large industrial CAFOs. Reducing methane emissions from large operations is essential to meet GHG emission targets, but these emissions can and should be controlled through regulatory safeguards (as described above) rather than through voluntary incentive mechanisms. New York's limited financial resources should support smaller operations, which often have thinner profit margins and face larger obstacles to

---

[27] *See* Adam Kotin et al., Cal. Climate & Agric. Network, *Diversified Strategies for Reducing Methane Emissions from Dairy Operations* (2015), https://calclimateag.org/wp-content/uploads/2015/11/Diversified-Strategies-for-Methane-in-Dairies-Oct.-2015.pdf; *see also*, Olga Gavrilova et al., Emissions From Livestock and Manure Management, 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories 67 tbl.10.17 (2019), https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch10_Livestock.pdf.
[28] Lehner & Rosenberg, *supra* note 5 at 99 [attached as Exhibit A].
[29] *See* Felipe Montes et al., *Mitigation of Methane and Nitrous Oxide Emissions from Animal Operations: II. A Review of Manure Management Mitigation Options*, 91 J. Animal Sci. 5070 (2013); *see also* Mathieu Dumont et al., *Methane Emissions in Biogas Production*, Biogas Handbook (2013); *See also* Thomas Flesch, et al., *Fugitive Methane Emissions from an Agricultural Biodigester*, 34 Biomass & Bioenergy 3927 (2011); *see also* Jessica Fu, *Is California Giving Its Methane Digesters Too Much Credit?*, The Counter (May 19, 2022), https://thecounter.org/is-california-giving-its-methane-digesters-too-much-credit/.
[30] *See* Nicole G. Di Camillo, *Methane Digesters and Biogas Recovery - Masking the Environmental Consequences of Industrial Concentrated Livestock Production*, 29 UCLA J. Env't Law & Policy 367 (2011).

implementing sustainability practices.

In addition, the FSP should have explicit recommendations to increase the number and portion of organic operations in the state, aiming to at least double them by 2030 and double them again by 2040. For example, California calls for a doubling of organic agriculture by 2045 as a climate-smart strategy.[31] These operations generally use systems that generate far less methane from manure—and through better manure, compost and soil management, less cropland GHG emissions as well—and thus reliably reduce GHG emissions. Increasing support and incentives for certified (and perhaps non-certified) organic operations through direct and market support (including State procurement and certification) can also increase the profitability and viability of these operations.

C. The Final Scoping Plan Should Include Strategies for Increasing Oversight and Data Transparency Related to Emissions and Practices at Large Industrial CAFOs.

The FSP should recommend strategies for greater oversight of manure management planning and reporting to quantify emissions. Currently, the EPA's Greenhouse Gas Reporting Program requires livestock operations with manure management systems that have animal populations over a set threshold to report emissions of methane. *See* 40 C.F.R. § 98.360. The regulation applies to facilities using manure management systems including uncovered anaerobic lagoons, liquid/slurry systems with and without crust covers, storage pits, digesters, solid manure storage, dry lots (including feedlots), high-rise houses for poultry production, poultry production with litter, deep bedding systems for cattle and swine, manure composting, and aerobic treatment.[32]

However, EPA is currently prevented from implementing or enforcing this regulation due to restrictions placed on it in legislative riders. For example, section 437 of the Consolidated Appropriations Act states: "Notwithstanding any other provision of law, none of the funds made available in this or any other Act may be used to implement any provision in a rule, if that provision requires mandatory reporting of greenhouse gas emissions from manure management systems."[33] Thus, the full extent of emissions from manure management is not quantified well. In order to fully understand the scope of impact from this sector and achieve maximum emissions reduction, New York must gather these data from the largest of the state's livestock operations. The FSP should include strategies to require such data reporting from the largest facilities, particularly those receiving public funding through state programs, and should make these data publicly available.

IV. The Final Scoping Plan Should Include More Transformative Strategies for Reducing Enteric Methane Emissions from Livestock, Such as Feed Additives and Reductions in Livestock Antibiotic Use

[31] *See* Cal. Air Resources Bd., *Draft 2022 Scoping Plan Update* 65 (2022), https://ww2.arb.ca.gov/sites/default/files/2022-05/2022-draft-sp.pdf.
[32] *Id*.
[33] Consolidated Appropriations Act, Public Law No. 116–260, 116th Cong. § 437 (2021).

In addition to manure management, New York's meat and dairy operations release significant emissions directly from livestock as part of animals' digestive processes. The DSP proposes several effective strategies to reduce enteric methane emissions under AF10, including precision feed and forage management. While we support the promotion and expansion of these methods, the FSP should also explore strategies to accelerate the adoption of feed additives and integrate strategies to promote reductions in antibiotic use.

As noted under AF10, numerous feed additives have demonstrated promising results in decreasing methane emissions from livestock, at least in the short-term. One study documented a 30% decrease in enteric methane emissions over 12 weeks with the addition of 3-nitrooxypropanol, a chemical compound that blocks an enzyme critical to methane formation.[34] Another promising study found that supplementing livestock feed with red seaweed resulted in an 80% reduction in enteric emissions from cattle over 5 months.[35] Scientists continue to develop new additives that may have even more promising results. The FSP should include recommendations to accelerate the adoption of feed additives (through nudges, incentives, fees, and possibly mandates) as a potential approach to achieving significant emissions reductions and should explore opportunities to fund accelerated research and outreach on the development of novel strategies to reduce enteric emissions.

Finally, eliminating nontherapeutic uses of antibiotics in livestock could also be an effective approach to reducing emissions. Studies indicate that antibiotics may alter microbial activities and have cascading consequences that lead to increased methane emissions.[36] This research indicates that—in addition to mitigating the public health risks of increasing antibiotic resistance—minimizing antibiotic use could also be an effective method for decreasing emissions. The FSP should examine the prohibition or restriction of unnecessary antibiotic use (to the extent that it's still employed in New York), as well as any other emerging, science-based strategies for reducing enteric methane.

V. The Final Scoping Plan Should Focus on Strategies to Reduce Herd Size Which Could Accelerate Reductions in Both Manure and Enteric Emissions

The additions to AF9 and AF10 that we recommend above will help strengthen strategies to reduce emissions from manure and enteric fermentation from existing livestock. However, the DSP overlooks one key strategy entirely, which would reduce both enteric emissions and emissions from manure: the reduction in the number of livestock animals in New York State. As a long-term strategy, with appropriate support for a just transition for current producers, promoting dietary changes to reduce demand for dairy and beef products and thus ruminant livestock may be one of the strongest tools we have for reducing agricultural emissions. Given

[34] *See* Alexander Hristov et al., *An Inhibitor Persistently Decreased Enteric Methane Emission From Dairy Cows With No Negative Effect on Milk Production*, 112 Proc. Nat'l Acad. Sci. U.S. Am. 10663 (2015) (finding a 30% decrease in enteric methane emissions over 12 weeks with the addition of 3-nitrooxypropanol); *see also* J. Dijkstra et al., *Short Communication: Antimethanogenic Effects of 3-Nitrooxypropanol Depend on Supplementation Dose, Dietary Fiber Content, and Cattle Type*, 101 J. Dairy Sci. 9041 (2018) (A subsequent study to Hristov's).
[35] *See* Breanna M. Roque et al., *Red Seaweed* (Asparagopsis Taxiformis) *Supplementation Reduces Enteric Methane by Over 80 Percent in Beef Steers* 16 PLoS ONE (2021).
[36] *See* Tobin J. Hammer et al., *Treating Cattle with Antibiotics Affects Greenhouse Gas Emissions, and Microbiota in Dung and Dung Beetles*, 283 Proceedings Royal Soc'y Biological Sci. (2016).

that current and chronic over-production of dairy products also creates pressure for prices below production costs, a careful effort to reduce supply could have significant producer benefits (as federal farm policy did before 1980).

In addition, meat and dairy alternatives also present an enormous business opportunity. Indeed, while demand for dairy and beef products is falling,[37] demand for plant-based alternatives is skyrocketing.[38] The plant-based meat and dairy products market was already an over $29.4 billion industry in 2020—and is projected to reach $162 billion by 2030.[39] There is also growing interest in "cultured meat" products, given recent technological innovations and an influx of public and private funding for research and development.[40] These trends have prompted New York-based producers to reassess their operations—and, in some cases, have inspired rapid changes in operations to meet shifts in consumer demand. For example, in 2017, Elmhurst—a former dairy operation that was founded in 1925—responded to the "steady decline in dairy consumption and the changing American diet" by reopening as a plant-based milk operation in Buffalo, New York.[41] Like other efforts by producers to reimagine their operations, Elmhurst's transition indicates that the growing plant-based sector offers New York producers an enormous market—especially if they get ahead of the curve.[42]

Many studies have found lower GHG emissions throughout the full life cycle of both more plant-based diets and plant-based dairy and meat alternatives when compared to animal-based products.[43] In a carbon footprinting analysis of the USDA Foods Program based on one year of purchasing data, Friends of the Earth found that animal products were responsible for 98% of GHG emissions associated with the $1.3 billion of food purchasing for this program.[44] The same analysis found that replacing 25% of USDA's beef, pork, chicken, and cheese purchases with plant-based sources of protein would save 4 million metric tons of carbon dioxide

---

[37] *See, e.g.*, Hayden Stewart et al., USDA Econ. Research Serv., *Examining the Decline in U.S. Per Capita Consumption of Fluid Cow's Milk, 2003–18* (2021), https://www.ers.usda.gov/webdocs/publications/102447/err-300.pdf?v=5705.9.

[38] *See* Blake Byrne & Ryan Dowdy, *Demand for Plant-Based Meat is Growing. We Must Ensure Our Supply Chain Can Keep Up*, Good Food Institute (Jan. 21, 2022), https://gfi.org/blog/meeting-plant-based-meat-demand/ (noting that "[i]n 2020, retail sales for plant-based alternatives grew twice as fast as overall food sales in the US" and that "[s]ales for plant-based meat in particular grew 45 percent.").

[39] Bloomberg Intelligence, *Plant-Based Foods Poised for Explosive Growth* (2020), https://assets.bbhub.io/professional/sites/10/1102795_PlantBasedFoods.pdf.

[40] *See* Kate Aronoff, *Lab to Table*, New Republic (Sept. 29, 2021), https://newrepublic.com/article/163554/lab-meat-save-planet; Isaac Nicholas & Mike Silver, *Tufts Receives $10 Million Grant to Help Develop Cultivated Meat*, TuftsNow (Oct. 15, 2021), https://now.tufts.edu/articles/tufts-receives-10-million-grant-help-develop-cultivated-meat (describing USDA funding for interdisciplinary research about cultured meat products).

[41] *The Dairy That Gave Up Dairy*, Elmhurst, https://elmhurst1925.com/pages/our-story (last visited Mar. 29, 2022).

[42] *See* Liz Susman Karp, *Farmers Trial Climate-Friendly Chickpeas in Upstate New York*, Civil Eats (May 3, 2022), https://civileats.com/2022/05/03/farmers-trial-climate-friendly-chickpeas-in-upstate-new-york/ (describing a farm in the Finger Lakes region that transitioned to chickpea farming to meet demand spurred by "the popularity of plant-based products").

[43] *See* Peter Newton & Daniel Blaustein-Rejto, *Social and Economic Opportunities and Challenges of Plant-Based and Cultured Meat for Rural Producers in the US*, Frontiers Sustainable Food Sys., (2021); *see also* Martin C. Heller et al., *Greenhouse Gas Emissions and Energy Use Associated with Production of Individual Self-Selected US Diets*, 13 Env't Rsch. Letters (2018); *see also* World Resources Institute, *Creating a Sustainable Food Future* (2019), https://files.wri.org/d8/s3fs-public/wrr-food-full-report.pdf.

[44] Friends of the Earth, *USDA Foods: How A $1.3 Billion Program Can Be Transformed to Create a More Just and Healthy Food System* (2021), https://foe.org/usda-foods.

equivalent (CO2eq) annually, which is equivalent to taking every registered automobile in Mississippi off the road.[45]

There are also numerous studies showing that plant-based diets are better for public health.[46] New York should dramatically increase efforts to promote these healthier, lower-emission diets—through the state's food procurement policies, public education programs, and increasing access to plant-based foods. The FSP should explore these and other strategies for decreasing meat and dairy consumption, and in turn reducing both manure management-related and enteric methane emissions.

VI. Soil Health, Nutrient Management, and Agroforestry

Improving soil health has the potential to restore soil carbon stocks lost due to poor land use practices. These efforts can help store carbon and reduce losses of nutrients from soil. The FSP should increase the focus of soil health, nutrient management and agroforestry programs to optimize climate-benefits, improve equity, ensure accountability, and reduce pesticide use, while advancing goals to improve water quality and protect biodiversity. As discussed below, the FSP may do so by setting targets for adoption rates of improved practices, setting targets for funding to disadvantaged communities, and recommending strategies to shift funds away from practices that further entrench polluting systems from industrial animal agriculture. The FSP should also include clear plans for measurement of outcomes, integrate strategies to reduce pesticide use, and impose fees on excess fertilizer use to promote improved nutrient management.

A. The Final Scoping Plan Should Set Statewide Goals Based on Targets Proposed at The National Scale for the Adoption of Climate-Friendly Practices to Track Progress and Increase Accountability.

The DSP includes several strategies to incentivize and promote the voluntary adoption of climate-friendly soil health practices. The FSP should go one step further and recommend mechanisms to ensure these strategies effectively drive shifts in practice adoption. The FSP should include targets for the adoption of climate-friendly practices to track progress and increase accountability. It can model such targets after those proposed at the federal level. For example, The Agriculture Resilience Act, introduced by U.S. Representative Chellie Pingree (ME) in 2021, offers several possible targets for 2040 that could be adopted in New York:

- Expand soil health practices to restore at least half of soil organic carbon that has been lost in the last 300 years;

---

[45] *Id.* at 4.
[46] *See* Am. College of Lifestyle Med., *The Benefits of Plant-Based Nutrition* (2021), https://lifestylemedicine.org/common/Uploaded%20files/PDFs/News%20Room/ACLM-2021_Plant-based-Nutrition.pdf; *see also More Evidence for Replacing Red Meat with Other Protein Sources*, Harvard T.H. Chan Sch. Pub. Health , https://www.hsph.harvard.edu/news/hsph-in-the-news/replacing-red-meat-with-other-protein/ - :~:text=Reducing red meat consumption while,School of Public Health researchers (last visited June 13, 2022); Press Release, World Health Org., Int'l Agency Rsch. Cancer, *IARC Monographs Evaluate Consumption of Red Meat and Processed Meat* (Oct. 26, 2015), https://www.iarc.who.int/wp-content/uploads/2018/07/pr240_E.pdf.

- Maintain year-round cover on at least 75% of cropland acres;
- Establish advanced grazing management on 100% of existing grazing land;
- Reduce GHG emissions related to the feeding of ruminants by at least 50% by reducing non-grazing of ruminants, growing feed grains and forages with soil health and nutrient practices that minimize net GHG emissions from cropland, and utilizing livestock feed mixtures and supplements to mitigate enteric methane emissions;
- Increase crop-livestock integration by at least 100% over 2017 levels; and convert at least two thirds of wet manure handling and storage to alternative management.[47]

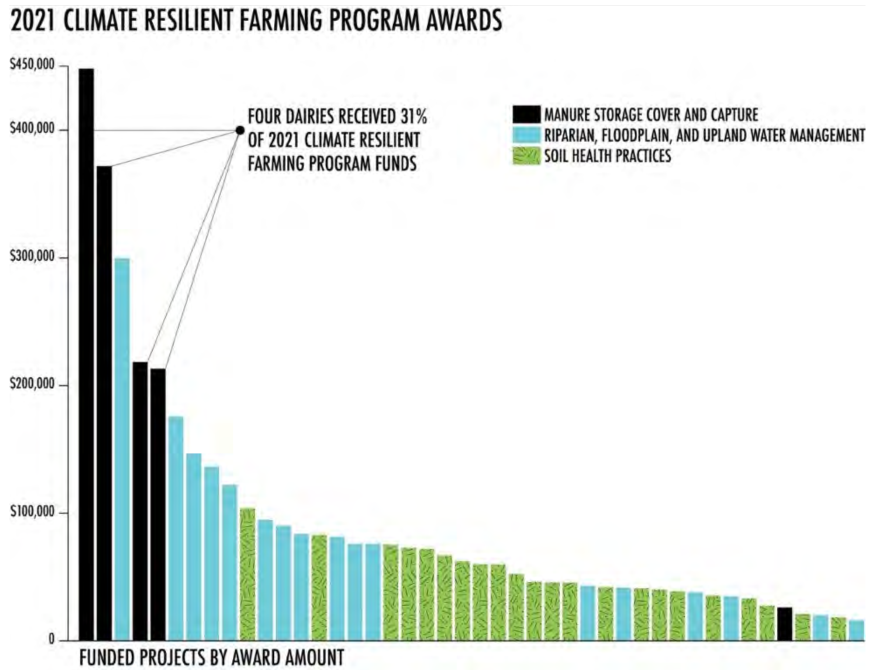
Additionally, the FSP could include a target for reductions in the use of fossil-fuel based synthetic inputs, such as a 25% reduction in total fertilizer use by 2040, consistent with data on current excess application, and a 50% reduction of synthetic fertilizer use by 2040 due to its much greater climate impacts. Including specific targets, such as those listed above, will be necessary to drive progress toward climate targets for the voluntary strategies listed in the DSP.

B. The Final Scoping Plan Should Focus Strategies and Soil Health Funding on Climate-Friendly Perennials, Rather Than Practices That Further Entrench Polluting Systems from Animal Agriculture

The DSP includes expanded support for existing programs as a strategy to incentivize adoption of soil health practices (AF12). However, the FSP should ensure that these expanded programs focus funding exclusively on soil health practices with clear climate benefits, rather than practices that entrench polluting systems, as they have in the past. For example, in Climate Resilient Farming ("CRF") Program awards announced for 2021, a few large dairies received significant grants up to $448,000 to install cover and flare systems, while soil health practices accounted for smaller allocations of funds (see figure below).[48] Over 30% of total program funds went to just four large dairies to adopt this practice.[49] Manure management practices at these large operations are already eligible for support through the Department of Agriculture and Markets Agricultural Nonpoint Source Pollution Abatement and Control Program, which in 2021 allocated $8.9 million (or 55% of the total program budget) to projects including manure storage and management practices.[50] Thus, for improved equity and efficacy, the FSP must include guidelines to ensure that expanded programs are tailored towards soil health practices with climate benefits rather than simply channeling funding towards large industrial animal facilities.

---

[47] Agriculture Resilience Act, H.R. 2803, 117th Cong. (2021).
[48] N.Y. Dep't of Agric. & Markets, *Climate Resilient Farming Round* 5 (2021), https://agriculture.ny.gov/system/files/documents/2021/02/crf_round5_projectdescriptions_0.pdf.
[49] *Id.* Note that this is an improvement from 2018 awards, in which 78% of funds went to just four dairies and one swine farm to implement cover and flares. *See also* N.Y. Dep't of Agric. & Markets, *Climate Resilient Farming* (2018), https://agriculture.ny.gov/soil-and-water/climate-resilient-farming.
[50] *See* N.Y. Dep't of Agric. & Markets, *Agricultural Nonpoint Source Pollution Abatement and Control Program Round* (2021), https://agriculture.ny.gov/soil-and-water/agricultural-non-point-source-abatement-and-control.



C. The Final Scoping Plan Should Ensure 40% of the Benefits from Soil Health Programs Accrue to Disadvantaged Communities

Given the legacy of discrimination that prevented farmers of color from gaining equal resources to support ownership of agricultural operations, it is essential that farmers and ranchers of color benefit from the resources provided through CRF and other programs. As noted by the CJWG, Black, Indigenous, and people of color ("BIPOC") producers represent a small fraction of total producers in New York state and an even smaller proportion of producers on the largest farms. For example, only 0.24% of farmers in New York State are Black, and government subsidies and support per Black farm are 60% less than average payments per farm for all of New York's farms.[51]

The CLCPA directs that disadvantaged communities receive at least 35% of overall benefits of spending on certain key climate, energy, and environmental investments. *See* ECL § 75-0117. The Biden Administration has similarly committed to direct 40% of certain climate and environmental federal investments to disadvantaged communities.[52] The CLCPA is not entirely clear whether this mandate applies to agricultural expenditures. We urge that the FSP include guidelines to extend this guarantee to the benefits of funding from soil health programs. Many soil health programs will benefit both the farmers and/or those downstream or downwind, and the FSP should ensure that disadvantaged communities, including previously underserved farmers, in New York benefit from soil health programs. The FSP must recognize that existing

---

[51] *Rising and Organizing in New York State*, Black Farmers United NYS, https://www.blackfarmersunited.org/statements/rising-and-organizing-in-new-york-state (last updated Apr. 6, 2022); *see also* USDA, Nat'l Agric. Stat. Serv., *Census of Agriculture* (2017), https://www.nass.usda.gov/Publications/AgCensus/2017/Online_Resources/Race,_Ethnicity_and_Gender_Profiles/New_York/cpd36000.pdf.

[52] *See* Tackling the Climate Crisis at Home and Abroad, Exec. Order No. 14,008, 86 Fed. Reg. 7619 (Jan. 27, 2021).

incentive programs that benefit the largest farms will further entrench these disparities and must therefore ensure that disadvantaged communities have access to relevant forms of support.

D. The Final Scoping Plan Should Include a Plan for Measurement of Outcomes.

The DSP largely relies on voluntary programs to incentivize the adoption of soil health practices. While many soil health practices, including cover crops, improved nutrient management, perennial crops, conservation crop rotations, and agroforestry have demonstrated climate benefits compared to conventional cropping systems,[53] the FSP should include plans for measurement, monitoring and verification of outcomes to ensure accountability and track soil health progress within New York. This accountability is necessary for ensuring that funding results in climate benefits—either through increases in carbon sequestration or reductions in emissions—and for shaping state programs towards practices with maximal climate benefits. Measurement, monitoring and verification can also help guide research efforts and provide valuable information for outreach and education specific to producers in New York.

While many soil health practices have been well documented and proven, they are still not widely adopted in New York and there is always room for additional improvement in their design and implementation with respect to specific crops, regions, and contexts. Moreover, as farmers themselves are in an excellent position to share information with others, both about implementation and impact, there should be a strong push—and funding—for gathering detailed documentation about project implementation and environmental outcomes with each funded project. In addition, if taxpayers can be assured that their money is being put to good use and achieving the goals it is being allocated for, it is more likely the program will be able to continue and grow. Industrial-scale producers receiving these sources of funding should be required to submit detailed documentation of implemented activities and data on outcomes and key environmental indicators to DEC. This will allow the agency, legislature, and the public to measure the progress of the program, help quantify its environmental benefits, provide data to help refine and improve the program, and give farmers the information they need to make sound business and conservation decisions.

E. The Final Scoping Plan Should Include Strategies to Reduce Reliance on Pesticides and Herbicides

[53] *See*, e.g., Amy Swan et al., USDA & Colo. State Univ., *COMET-Planner: Carbon and Greenhouse Gas Evaluation for NRCS Conservation Practice Planning*, http://bfuels.nrel.colostate.edu/health/COMET-Planner_Report_Final.pdf; *see also* Christopher Poeplaua & Axel Don, *Carbon Sequestration in Agricultural Soils via Cultivation of Cover crops – A Meta-analysis*, 200 Agric., Ecosystems & Env't 33 (2015) [attached as Exhibit E]; Jinshi Jian et al., *A Meta-analysis of Global Cropland Soil Carbon Changes Due to Cover Cropping*, 143 Soil Biology & Biochemistry 107,735 (2020) [attached as Exhibit F]; Shibu Jose & Sougata Bardhan, *Agroforestry for Biomass Production and Carbon Sequestration: An Overview*, 86 Agroforestry Systems 105 (2012) [attached as Exhibit G]; Joseph E. Fargione et al., *Natural Climate Solutions for the United States* 4 Sci. Advances (2018); *AgEvidence*, The Nature Conservancy, https://www.agevidence.org/ (last visited June 13, 2022); Xiongxiong Bai et al., *Responses of Soil Carbon Sequestration to Climate-smart Agriculture Practices: A Meta-analysis*, 25 Global Change Biology 2591 (2019) [attached as Exhibit H].

Reducing pesticide and herbicide use is critical to building soil health and preventing harm to non-target organisms and surrounding communities. Healthy soil depends on the presence of billions of soil microorganisms, including bacteria and fungi. Pesticide use by its very nature kills beneficial as well as harmful life in soil and thus often impairs soil health and fertility, with the potential to impact soil carbon and nutrient cycling and climate. Pesticides and herbicides can alter the composition, diversity, and functioning of soil organisms. Ultimately, pesticides and herbicides can harm and alter soil communities that play a major role in carbon sequestration and create a thriving agricultural system.[54] We thus urge the FSP to explicitly include strategies to reduce synthetic pesticide and herbicide use and to promote integrated pest management (including alternative strategies to suppress pests through conservation crop rotations, cover crops and other agroecological practices[55]) as key goals.

F. The Final Scoping Plan Should Include Additional Strategies to Reduce Nitrous Oxide Emissions from Excess Fertilizer Use, Including Outreach and Consideration of a Graduated Fertilizer Fee

The FSP should more directly address excess application of fertilizer, a common practice that has several harmful environmental and climate impacts, including the release of nitrous oxide, which is both a potent greenhouse gas and a major ozone depleting substance. This gas is emitted almost entirely by agricultural soil management and accounts for about 10% of the state's agricultural GHG emissions.[56] Farmers routinely apply fertilizer at higher rates than crops require for a variety of reasons: as a form of insurance or risk avoidance, hope for a great year, over-focus on yield over return, habit, and misinformation.[57] Due to losses to the atmosphere, retention in soil, and runoff to waterways, only a proportion of the nitrogen applied as fertilizer to annual grains is removed at harvest.[58] In addition, in New York, application of manure from CAFOs in the winter or on saturated ground is allowed, even though plants do not take up any nutrients at those times. These practices result in large losses of nutrients, leading to nitrous oxide emissions among other negative consequences.

The DSP recognizes that "[e]fficient use of nitrogen fertilizer can reduce nitrous oxide emissions from cropland, improve water quality, and can save the farmer money."[59] The efficient use of fertilizer includes applying it at the right time and place and can be advanced by practices such as split application and slow-release fertilizers. We support certain strategies in the DSP, including increasing outreach and support for improved nutrient management, especially to and

---

[54] *See* Kendra Klein, Friends of the Earth, *Pesticides and Soil Health* (2019), https://foe.org/wp-content/uploads/2019/08/PecticidesSoilHealth_Final-1.pdf.
[55] *See Integrated Pest Management*, USDA Nat'l Res. Conservation Serv., https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/about/?cid=nrcs144p2_027181 (last visited June 13, 2022).
[56] *See* N.Y. DEC, *supra* note 2.
[57] G. Philip Robertson & Peter M. Vitousek, *Nitrogen in Agriculture: Balancing the Cost of an Essential Resource*, 34 Ann. Rev. Env't & Res. 97, 117 (2009) (Finding that farmers often apply excess fertilizer "in the hopes that 'this year will be the one in ten' when extra N will pay off.").
[58] G. Philip Robertson, *Nitrogen Use Efficiency in Row-Crop Agriculture: Crop Nitrogen Use and Soil Nitrogen Loss*, Ecology in Agriculture 351 (Louise E. Jackson ed., Academic Press 1997).
[59] DSP at 213.

for previously underserved farmers. We support the use of all existing support programs including the Agricultural Nonpoint Source Abatement and Control program and the new CRF program. However, the FSP should incorporate at least two additional measures to improve nutrient management on farms in New York.

First, since one of the most important things a farm can do is apply fertilizer no earlier than the planting season,[60] the FSP should recommend that DEC revise its CAFO general permits—applicable to the several hundred large dairies in New York accounting for nearly 70% of New York's dairy cow population—to prohibit winter manure spreading. Such a provision would reduce both nitrous oxide and methane emissions because fertilizer left unutilized in the soil over winter is vulnerable to environmental loss, including as nitrous oxide.[61]

Second, the FSP should recommend more aggressive efforts to incentivize improved fertilizer management, including phased-in institution of a fertilizer fee. While the DSP includes discussion on imposing some form of price on carbon dioxide emissions in its Economy-Wide Strategies Chapter (which we address elsewhere in these comments), the FSP should also develop a similar approach for nitrous oxide. The FSP should include consideration of imposing a fertilizer fee that could directly encourage and fund assistance for farmers' enhancing fertilizer use efficiency. This should be structured to apply only to excess fertilizer, such as applying over the per acre amounts that represent the plants' nutritional needs. More sophisticated fee schemes could provide a base rate with discounts for enhanced-efficiency fertilizers that emit less nitrous oxide. To assist in the transition, such a fee could be phased in, with significant outreach and technical assistance beforehand to enable farmers to adopt precision and other improved fertilizer management regimes. All revenue from the fee should be directed to farmer support. Improvements in fertilizer management are possible—and profitable—with similar reductions in nitrous oxide emissions.

VII. Climate-Focused Bioeconomy

A. The Final Scoping Plan Should Ensure That Strategies Listed Under the Climate-Focused Bioeconomy are Founded on Accurate Accounting of the Climate Impact of Harvesting and That They Do Not Undermine Strategies Listed in the Land Use Chapter

The DSP includes a number of strategies under the Climate Focused Bioeconomy section which prioritize growth of the forestry industry over climate mitigation. The FSP should include guidance to relevant agencies to ensure that any funding for forestry training as described in

---

[60] *See* Marc Ribaudo et al., USDA Econ. Rsch. Serv., ERR-127, *Nitrogen in Agricultural Systems: Implications for Conservation Policy* 11 (2011), https://www.ers.usda.gov/publications/pub-details/?pubid=44919; *see also* Terry L. Roberts, Int'l Plant Nutrition Inst., *Right Product, Right Rate, Right Time, and Right Place . . . the Foundation of Best Management Practices for Fertilizer*, Fertilizer Best Management Practices, 29–32, (1st ed. 2007); G. Philip Robertson et al., *Nitrogen-Climate Interactions in U.S. Agriculture*, 114 Biogeochemistry 41, 55–56 (2013).

[61] *See* Xiaojing Hao et al., *Nitrous Oxide Emissions From an Irrigated Soil as Affected by Fertilizer and Straw Management*, 60 Nutrient Cycling Agroecosystems 1, 5 (2001) [attached as Exhibit I]; Claudia Wagner-Riddle & G.W. Thurtell, *Nitrous Oxide Emissions From Agricultural Fields During Winter and Spring Thaw as Affected by Management Practices*, 52 Nutrient Cycling Agroecosystems 151, 162 (1998) [attached as Exhibit J].

AF18, efforts to expand wood product markets as described in AF19, or promotion of wood products as described in AF8, reflect accurate accounting of the impact of harvesting on both (1) existing carbon stocks in New York forests, and (2) the lost potential for sequestration resulting from removals of these carbon stocks. As described above, this is necessary to ensure that these recommendations do not undermine strategies laid out in the Land Use chapter, which rightly prioritize the climate benefits of keeping forests as forests rather than managing for commercial products. The FSP should provide guidance to relevant agencies to ensure that any funded education, outreach and product promotion reflect unbiased accounting of the potential negative impacts of forest harvesting on climate, and the FSP should direct relevant agencies to ensure that all claims and educational materials reflect sound science.

B. The Final Scoping Plan Should Not Include AF20, Which Calls for the Expansion of the Use of Biomass Feedstocks and Bioenergy Products

Biomass harvesting and bioenergy are false climate solutions and should have no place in the FSP. Biomass harvests reduce the capacity for New York's forests to continue functioning as a carbon sink. Not only do these activities reduce the magnitude of the carbon sink, they also lead to additional emissions from the harvest, burning, transportation and manufacture of wood products.[62] Harvesting biomass results in a lost opportunity for forest stands to continue to sequester carbon, as they would if left undisturbed. The FSP must recognize the fundamental benefits of leaving forests intact and carefully account for this potential for continued carbon sequestration in any proposals that suggest harvesting as a climate mitigation strategy.

As noted in the DSP, the CJWG has "expressed concerns about the combustion of biomass and biofuels due to their release of emissions."[63] The DSP fails to address these concerns, and should not include AF20, which calls for an expansion of biomass and bioenergy.

VIII. Conclusion

Reducing emissions from livestock and dairy production in New York, rebuilding soil organic carbon stocks on croplands, and restoring and protecting forests must all be a part of New York's climate action plan. The DSP includes several strategies that have the potential to reduce emissions from agriculture and forestry; however, there are a number of areas in which the DSP may be improved to avoid false solutions and increase accountability and impact. This includes the following:

Forestry

The FSP must revisit the currently proposed forestry strategies to avoid incentivizing removals from forests. The FSP must prioritize forest preservation and restoration efforts, which provide the maximum climate benefit, over managing forests to produce forest crops. The FSP must also not offer forests in New York as an excuse to delay action on reducing fossil fuel emissions through offset markets.

---

[62] *See* Tara W. Hudiburg et al., *supra* note 11 at 419 [attached as Exhibit D].
[63] DSP at 227.

1. The FSP should ensure that benefits for private forest landowners who manage for carbon sequestration or conserve their forests in natural conditions are *at least* equal to benefits for private forest landowners who manage for wood products.
2. The FSP should not include AF6, which relies on dangerous and ineffective offsetting of fossil fuel emissions through forest carbon sequestration.

Livestock

Successfully reducing methane emissions from livestock will require strategies that extend beyond the voluntary and limited suggestions in the DSP. The FSP must include regulatory options to mandate reductions in methane emissions from large operations, and the FSP should explore an additional suite of more transformative strategies to reduce methane from enteric fermentation and manure management.

3. The FSP should include regulatory options, as authorized under the ECL and consistent with the CLCPA, for reducing methane emissions.
4. The FSP should focus on more transformative strategies for reducing manure methane outside of cover and flare and digesters, including strategies to reduce manure generation and reducing wet storage, and more transformative strategies for reducing enteric methane emissions from livestock, such as feed additives and reductions in livestock antibiotic use.
5. The FSP should focus on strategies to reduce herd size which could accelerate reductions in both manure and enteric emissions.

Soil Health

While the DSP provides useful suggestions for increasing the adoption of soil health practices on croplands in New York, the FSP should incorporate additional strategies to ensure that soil health programs result in real climate benefits and that these funds support disadvantaged communities.

6. The FSP should set statewide goals based on targets proposed at the national scale for the adoption of climate-friendly practices and climate-smart systems such as organic practices, and should include a plan to track progress and increase accountability.
7. The FSP should focus on strategies to incentivize climate-friendly cropping practices, rather than practices that further entrench polluting systems from animal agriculture.
8. The FSP should ensure 40% of the benefits from soil health programs accrue to disadvantaged communities.
9. The FSP should urge revision of the CAFO general permit and development of a phased-in tiered fertilizer fee to incentivize enhanced fertilizer management.

Climate-focused Bioeconomy

The FSP should focus on preserving the climate benefits of keeping forests as forests. The FSP must revisit forestry strategies in the context of accurate carbon accounting unbiased by the forestry industry to avoid false solutions like bioenergy from forest carbon stocks.

10. The FSP should ensure that strategies listed under the Climate-Focused Bioeconomy are founded on accurate accounting of the climate impact of harvesting.
11. The FSP should not include AF20, which calls for the expansion of the use of biomass feedstocks and bioenergy products.

Respectfully submitted,

Acadia Center
Alliance for a Green Economy
Brookhaven Landfill Action and Remediation Group
Catskill Mountainkeeper
Clean Air Coalition of WNY
Climate Reality Project, Capital Region NY Chapter
Climate Reality Project, Finger Lakes Greater Region NY Chapter
Climate Reality Project, Hudson Valley and Catskills Chapter
Climate Reality Project, Long Island Chapter
Climate Reality Project, NYC
Climate Reality Project, Westchester NY Chapter
Climate Reality Project, Western New York Chapter
Committee to Preserve the Finger Lakes
Community Food Advocates
CUNY Urban Food Policy Institute
Earthjustice
Environmental Advocates NY
Friends of the Earth
Fossil Free Tompkins
Gas Free Seneca
Green Education and Legal Fund
HabitatMap
Hotshot Hotwires
Long Island Progressive Coalition
Nassau Hiking & Outdoor Club
Natural Resources Defense Council (NRDC)
Network for a Sustainable Tomorrow
New Clinicians for Climate Action
North Brooklyn Neighbors
Northeast Organic Farming Association of New York, Inc. (NOFA-NY)
NY Renews
People of Albany United for Safe Energy
Riverkeeper Inc.
Roctricity
Seneca Lake Guardian
Sierra Club
South Shore Audubon Society
Sustainable Finger Lakes
University Network for Human Rights
UPROSE
WE ACT for Environmental Justice

Exhibit A

Developments in Federal and State Law

Michael B. Gerrard
Editor

ENVIRONMENTAL LAW IN NEW YORK

ARNOLD & PORTER

LexisNexis®

Volume 33, No. 5 May 2022

Advancing Climate-Neutral Agriculture in New York
Viewpoint

Peter H. Lehner and Nathan A. Rosenberg

IN THIS ISSUE

In every state, New York included, the realities of climate change are becoming ever more tangible—more frequent floods, droughts, heat waves, and more. Agricultural producers are facing some of the most immediate impacts as these extreme and changed weather events, as well as increased pests and more challenging working conditions, profoundly affect producers' ability to grow plants and raise animals. As a result, federal and state officials are now looking at how we can reform our agriculture system both to reduce its contribution to climate change and to be more resilient to climate change's impacts. Our new book, *Farming for Our Future: The Science, Law, and Policy of Climate-Neutral Agriculture*,[1] aims to provide the foundation that policymakers need for making sound decisions.

Fortunately, New York is ahead of the pack and enacted a strong climate change law in 2019. The Climate Leadership and Community Protection Act (CLCPA) requires New York to reduce overall emissions by 40% below 1990 emissions by 2030 and by 85% by 2050.[2] The law also requires the state to achieve 100% emission-free electricity by 2040, but has no goals or targets for agriculture or other sectors.

Agriculture's contribution to climate change differs from what is normally discussed—the emission of carbon dioxide from the burning of fossil fuels. Agriculture's contributions, by contrast, come largely from methane from the exhalations of dairy and beef cows (ruminant animals whose guts produce methane) and liquid manure management; nitrous oxide that derives primarily from excess fertilization of crops; and carbon stripped from soils by agricultural activities. Over 20 years, methane is about 85 times stronger than carbon dioxide in global warming, and nitrous oxide is about 265 times stronger.[3]

In New York, the 2021 greenhouse gas (GHG) emissions report by the State Department of Environmental Conservation (DEC) indicates that agriculture is responsible for 6% of total statewide GHG emissions, and that 92% of those emissions come from livestock.[4] Unlike other sectors in New York where emissions have already decreased, livestock management emissions have increased 44% since 1990. Overall, about two-thirds of these emissions statewide are from cows exhaling methane and

[1] Peter H. Lehner & Nathan A. Rosenberg, Farming for Our Future: The Science, Law, and Policy of Climate-Neutral Agriculture (2021).

[2] N.Y. Env't Conserv. Law § 75-0107(1).

[3] *Understanding Global Warming Potentials*, EPA, https://www.epa.gov/ghgemissions/understanding-global-warming-potentials (last updated Oct. 18, 2021); *see also* Lehner & Rosenberg, *supra* note 1, at 47–57.

[4] *Statewide Greenhouse Gas Emissions Report*, N.Y. State Dept. Of Env't Conserv., https://www.dec.ny.gov/energy/99223.html#Report (last visited Mar. 10, 2022).

one-third from liquid manure management (although actual measurements of methane emissions suggest that the models on which these emission figures are based significantly understate actual manure emissions).[5]

This statewide figure is highly concentrated, with a small number of large facilities contributing the vast majority of emissions.[6] According to the U.S. Department of Agriculture (USDA) Census of Agriculture (the most recent from 2017), New York has about 625,000 dairy cows and a total of approximately 1.5 million cows and calves, including the beef herd.[7] All release enteric methane, although dairy cows release more per cow than beef cows. The state also has about 4,500 dairies and 7,300 beef operations. Most are small with cows on pasture, but New York also has about 500 "concentrated animal feeding operations" (CAFOs)—which in New York are mostly dairies with an average of over 800 cows where the animals are kept in confined spaces and feed is brought to them. These relatively few facilities hold about 70% of the state's dairy cows. These CAFOs generate a large amount of waste and under State law must have manure storage systems, mostly large liquid lagoons, that hold the waste until it can be spread on fields. The anaerobic conditions in these lagoons generate (per unit of waste) about 20 times or more methane than manure deposited on pasture. Thus, these facilities are not only responsible for the bulk of enteric emissions in New York, but also the majority of manure emissions which are exacerbated by storage conditions—an important consideration for State policymakers.

DEC's report considers only the direct emissions of agricultural activities and thus somewhat understates the real impact of agricultural operations on climate change—and thus the potential of farm policy reform. For example, while the CLCPA requires inclusion of the embedded GHG gas emissions of energy and fuel produced out of state and imported into New York (about 17% of total state GHG emissions),[8] it does not similarly require inclusion of the GHG emissions from the production of other energy-intensive projects such as fertilizer or pesticide imported into the state. (Nor does it require consideration of the embedded GHG from the food imported into the state, which is likely significant.) Thus, for example, policies that could reduce fertilizer use in the state would have a larger climate impact than analyses focused entirely on New York would suggest, and policies that reduce on-farm energy would help the State meet its overall goals, if not agriculture sector goals. Moreover, DEC assigns all energy-related emissions to the energy sector rather than to where the energy is used and thus does not include the impact of on-farm energy in the agriculture climate footprint. Perhaps for this reason, the draft scoping plan has no discussion of eliminating emissions from farm vehicles and equipment.

Perhaps more important, DEC's report discusses separately the climate impact of annual land conversions (which in New York is largely a function of the 19 million acres of forest), and does not include at all the lost carbon sequestration potential of prior land conversion of New York's almost 7 million acres in farms.[9] This farmland takes the place of native vegetation, which generally would otherwise sequester more carbon. However, this land offers great potential to regain and restore its lost carbon through more climate-friendly agricultural practices.

Supplementing DEC's report with all these considerations indicates that agriculture's true contribution to climate change in New York far exceeds the modest 6% figure given. Indeed, many analyses find that agriculture and food systems (which often include emissions from food processing and other after-farm steps) contribute about one-third of all U.S. (or global) emissions[10] and that countries cannot reach their climate goals without addressing the agriculture and food production sector.[11] Policymakers must recognize the importance and opportunities of this sector.

Unfortunately, thus far, ambition for the agriculture sector has been overly modest. The CLCPA created a Climate Action Council (CAC) to develop a specific plan for climate change mitigation. The statute also established several advisory committees to the CAC, including one on agriculture and forestry. That advisory committee recommended measures that would achieve only a 30% reduction from current levels by 2050 in agriculture's contribution to climate change (i.e., a return to 1990 levels), even though the CLCPA's overall goal is an 85% reduction below 1990 levels of emissions statewide.[12] The draft scoping plan adopts these recommendations and proposes actions that would result in a 15% GHG reduction by 2030 and a 30% reduction by

[5] *See, e.g.*, Matthew N. Hayek & Scot M. Miller, *Underestimates of Methane from Intensively Raised Animals Could Undermine Goals of Sustainable Development*, ENV'T RSCH. LETTERS 16 063006 (2021), https://doi.org/10.1088/1748-9326/ac02ef.

[6] *See* Earthjustice, Visualizing USDA 2017 Census of Agriculture Livestock Inventory Concentrations, https://sustainablefoodfarming.shinyapps.io/CAFOINVENTORY/ (last visited Mar. 28, 2022).

[7] *See Census of Agriculture*, U.S. DEPT. OF AGRIC., https://www.nass.usda.gov/Publications/AgCensus/2017/index.php (last modified Jan. 12, 2022).

[8] *See* N.Y. STATE CLIMATE ACTION COUNCIL, DRAFT SCOPING PLAN 25–26 (Dec. 30, 2021) [hereinafter CLCPA DRAFT SCOPING PLAN], https://climate.ny.gov/-/media/Project/Climate/Files/Draft-Scoping-Plan.ashx.

[9] *See, e.g.*, Matthew N. Hayek et al., *The Carbon Opportunity Cost of Animal-Sourced Food Production on Land*, 4 NATURE SUSTAINABILITY 21 (2021), https://doi.org/10.1038/s41893-020-00603-4.

[10] *See, e.g.*, M. Crippa et al., *Food Systems Are Responsible for a Third of Global Anthropogenic GHG Emissions*, 2 NATURE FOOD 198 (2021), https://doi.org/10.1038/s43016-021-00225-9.

[11] Michael A. Clark et al., *Global Food System Emissions Could Preclude Achieving the 1.5° and 2°C Climate Change Targets*, 370 SCI. 705 (2020), https://doi.org/10.1126/science.aba7357.

[12] *Appendix A: Advisory Panel Recommendations*, *in* CLCPA DRAFT SCOPING PLAN, *supra* note 8, https://climate.ny.gov/-/media/Project/Climate/Files/Draft-Scopping-Plan-Appendix-A.ashx.

2050.[13] The CAC should explore and develop additional and more aggressive abatement opportunities. To date, however, the CAC and the agencies drafting the scoping plan appear to be continuing to downplay agriculture's role, even noting that "Climate Act policies typically affect only combustion sources."[14]

There is still time for the CAC to develop a more ambitious plan. There are many opportunities to change agricultural practices that would significantly reduce GHG emissions. It has been well demonstrated that farmers and ranchers can reduce net GHG emissions through improved grazing and animal feeding practices, better manure management, more efficient irrigation, and climate-friendly crop and soil management.[15] For instance, aggressive agricultural tillage of soil and pesticide application reduce soil organic carbon and nutrients. Farmers seek to compensate for this with heavy doses of fertilizer—often synthetic fertilizer, the manufacture of which itself creates significant GHG emissions. Indeed, studies show that on average about twice as much fertilizer is applied to U.S. cropland than the plants can use. The fertilizer not taken up by plants runs off into waterways (contributing to harmful algae blooms) or converts into nitrous oxide driving climate change. Yet soil can be restored to biological health and greater natural fertility. Reduced tillage that disturbs the soil less and cover crops that keep soil covered all year can restore nutrients to the soil. This reduces the need for fertilizers—saving farmers money—and the amount of nitrous oxide released into the atmosphere, as well as increasing the carbon stored in the soil and plants.

The challenge is that these climate-friendly practices are employed on just a small fraction of New York agricultural land. In 2017, cover crops were used on only about 14% of New York cropland, no-till on 18%, and certified organic practices on a mere 4%. There is no accessible data on how widespread managed rotational grazing practices are; these practices can significantly reduce erosion and increase soil carbon sequestration. Liquid manure management practices that increase methane emissions have actually increased in recent years, causing the emissions growth noted above. Policy change is needed to accelerate the development and adoption of climate-friendly practices. Widespread implementation of these practices could make New York's agriculture close to or even beyond climate neutral.

New York has several tools in addition to the CLCPA to accelerate adoption of these policies. Much of the State's technical outreach to farmers and ranchers is through the Agricultural Environmental Management (AEM) program overseen by the Department of Agriculture and Markets and the New York Soil and Water Conservation Committee and Districts. The draft scoping plan appropriately proposes to expand this program, providing site-specific climate-specific assistance.[16] However, much of this work is not shared and is held confidential, inhibiting the ability of farmers to learn from others and to build greater public support for these changes on farms. In addition to expanding this program, it would be helpful to increase its transparency.

Most of these climate-friendly practices will also reduce nutrient and pesticide runoff from fields and animal operations and can make farms more resilient to extreme weather. Thus, the state Agricultural Non-Point Source program, which funds projects to reduce farm runoff, can be expanded and focused on particularly climate-friendly practices. The Climate Resilient Farming program has provided about $20 million since 2015, some for soil management practices and most to cover manure lagoons at dairies and flaring the methane to convert it to carbon dioxide, thereby reducing the warming potential of these emissions (but not addressing many other polluting aspects of CAFOs). In 2021, the Legislature passed and Governor Hochul signed the Soil Health and Climate Resiliency Act,[17] which codifies the Climate Resilient Farming program; creates a soil health initiative that will, among other things, establish soil health standards; and increases research and development funding authority. In her proposed budget (S. 8004/A. 9004), Governor Hochul proposed to increase funding to $15 million for Soil and Water Conservation Districts and $17.5 million for the Climate Resilient Farming program, tripling past funding. Viewing all of agriculture and its impacts holistically opens the door to more system-level approaches that can benefit both farmers and the landscape.

As the State implements the CLCPA, it can augment and accelerate these efforts. One of its main tools to drive policy is to set targets that agencies must then figure out how to meet. As noted, the law has specific targets for electricity generation but none for the agriculture or land use sector. Before it is finalized, the scoping plan must include additional specific practice-based targets to be achieved by 2030 and 2050 to enable the sector to achieve more significant reductions and to guide further policy. Even trying to develop such goals—or other adaptive management system of accountable continuous improvement—would lead to important policy advances. (Indeed, one of the specific recommendations is to monitor and benchmark agricultural GHG emissions, noting that it is "central to the success of all other agricultural mitigation efforts."[18]) There is currently a woeful lack of accurate information about agricultural practices employed in New York (and elsewhere). New York does not collect its own data but relies

[13] *See* CLCPA Draft Scoping Plan, *supra* note 8. Appendix G of the scoping plan shows different reductions associated with different scenarios in the integration analysis for agriculture (see page 54). This shows about a 13–15% reduction by 2030 and a 30–34% reduction by 2050 under the primary scenario. This also shows a greater 20% reduction by 2030 for all other scenarios, and a 38–63% reduction by 2050 under other scenarios.

[14] N.Y. State Climate Action Council, Integration Analysis: Benefits and Costs with Sensitivity Analysis (updated Nov. 18, 2021), https://climate.ny.gov/-/media/Project/Climate/Files/2021-11-18-Integration-Analysis-Benefits-Costs-Presentation.ashx.

[15] *See, e.g.*, Natalie D. Hunt et al., *Fossil Energy Use, Climate Change Impacts, and Air Quality-Related Human Health Damages of Conventional and Diversified Cropping Systems in Iowa, USA*, 54 Env't Sci. & Tech. 10977 (2020), https://doi.org/10.1021/acs.est.9b06929.

[16] *See* CLCPA Draft Scoping Plan, *supra* note 8, at 195, 220–21.

[17] 2021 N.Y. Laws ch. 735.

[18] CLCPA Draft Scoping Plan, *supra* note 8, at 221.

on the Census of Agriculture done by the USDA every five years. For several reasons, this data is incomplete and often inaccurate. At the federal level, there has been great resistance to increasing reporting of GHG emissions from CAFOs; New York policy makers should not succumb to similar pressure here. The effort to set targets and monitor progress toward them would itself provide valuable information to New York policymakers and better allow farmers to learn from each other.

The Agriculture Resilience Act, first introduced by U.S. Representative Chellie Pingree (ME) in 2020,[19] offers several possible targets that could be adopted in New York:

- Overall reduce agriculture's GHG footprint 50% below 2010 levels by 2030 and achieve net-zero emissions by 2040 (a far more ambitious target than proposed in the draft scoping plan);
- Triple agricultural research and development funding by 2030 and quadruple it by 2040 (offsetting a long-term reduction in such funding and critical to achieving the other goals);
- Maintain year-round living plant cover on at least 50% of cropland acres by 2030 and 75% by 2040;
- Expand soil health practices to increase soil carbon by at least 0.4% per year (consistent with an international effort to offset much of the world's fossil fuel GHG emissions);
- Establish advanced rotational grazing management (which increases soil carbon and decreases manure emissions) on 50% of grazing land by 2030 and 100% by 2040;
- Increase agroforestry and silvicultural practices onto 15% of applicable land by 2030 and 30% by 2040;
- Increase crop-livestock integration by at least 100% over 2017 levels by 2040;
- Convert at least half of wet manure handling and storage to dry or alternative management systems that generate far less methane by 2040; and
- Reduce GHG emissions related to the feeding of ruminants by at least 50% below 2010 levels by reducing non-grazing of ruminants, growing feed grains and forages with soil health and nutrient practices that minimize net GHG from cropland, and utilizing livestock feed mixtures and supplements to mitigate enteric methane emissions.

New York could of course also establish additional targets, such as dramatically increasing the amount of organic farmland, both certified and non-certified, in the state (rough estimates suggest that total organically managed farmland in the state may be double the certified acreage). Similarly, it should add equity-focused goals, such as significantly increasing the amount of farmland owned and operated by farmers of color (now less than one percent of New York farmland) and ensuring that these farmers get a major share of the State's financial and technical support.

Livestock methane comprises the bulk of the state's agricultural GHG emissions, and the draft scoping plan includes two strategies for livestock emissions—improved manure management and feed management.[20] However, the draft scoping plan integration analysis shows only partial reduction of manure methane (far less than the statewide 85% goal) and little reduction in enteric methane in all but the last (less likely) scenario. But the plan notes that "additional innovation in methane mitigation" is necessary to attain the CLCPA goals and even acknowledges that this "will require transformative solutions."[21]

Despite this acknowledgment, the current recommendations are modest. They focus largely on technological measures such as destroying manure methane and should focus more on reducing methane generation. Known feed additives that slightly reduce methane emissions are used on much of the dairy herd, but relatively little research or effort is yet underway to tackle enteric emissions[22] or find alternatives to products with unavoidably high emissions. The recommendations could also put more emphasis on shifting to dry management and pasture systems, which generate a tiny fraction of the methane generated by liquid systems, or reducing herd size.[23] The draft scoping plan does not consider alternative plant-based milk products as an option, even though this could significantly decrease methane emissions and be a major growth industry for New York agriculture.[24] This omission is akin to looking only at reducing emissions from coal-fired power plants but not looking at energy efficiency or renewable energy.

The draft scoping plan then offers several soil health and nutrient management recommendations.[25] These generally seek to increase education and outreach, as well as financial and technical assistance for better soil health practices and to ensure programs are reaching all—and especially previously

[19] H.R. 5861, 116th Cong. (2020). Representative Pingree reintroduced the bill in 2021. *See* H.R. 2803, 117th Cong. (2021).

[20] CLCPA DRAFT SCOPING PLAN, *supra* note 8, at 208–13.

[21] *See* Climate Action Council, Draft Scoping Plan Overview 8, 12 (Jan. 2022) [hereinafter Draft Scoping Plan Overview], https://climate.ny.gov/-/media/Project/Climate/Files/Draft-Scoping-Plan-Overview.ashx; CLCPA DRAFT SCOPING PLAN, *supra* note 8, at 74.

[22] The draft scoping plan acknowledges the need for research to address enteric emissions. CLCPA DRAFT SCOPING PLAN, *supra* note 8, at 75. The various State budget proposals included some additional research and development (R&D) funds. Along with more R&D, there should be a commitment to transparency about the studies and results.

[23] *See* LEHNER & ROSENBERG, *supra* note 1, at 94–102.

[24] *See, e.g.*, *The Dairy That Gave Up Dairy*, ELMHURST, https://elmhurst1925.com/pages/our-story (last visited Mar. 29, 2022).

[25] *See* CLCPA DRAFT SCOPING PLAN, *supra* note 8, at 213–23.

underserved—producers.[26] These sound voluntary measures would be well supported by some of the other measures suggested here.

One of the most powerful tools to reduce net GHG emissions is shifting to more perennial crops,[27] and the draft scoping plan proposes increasing agroforestry.[28] Perennial plants produce more biomass than shorter-lived annuals, and this additional biomass stores dramatically more carbon. This effectively pulls carbon dioxide from the atmosphere and helps restore lost and declining soil carbon stocks. Even conservative estimates find that perennial practices sequester two to five times more carbon per acre than the most effective annual practices.[29] New York is already one of the largest producers of apples, grapes, and other perennials. We can further expand the number of trees, shrubs, and other perennial plants on farms and ranches by intercropping annual plants between rows of tree crops, introducing riparian and field buffers of trees with marketable crops, and adding trees to pasture. The plan recommends programs to expand these practices through incentives, financial support, and technical assistance; it would do well to also consider making adoption of some of these practices (e.g., riparian buffers) a "best management practice" that must be followed in order to receive general financial support.[30]

While the CAC is considering imposition of some form of price on carbon dioxide emissions (or "polluter pays"), the State could also develop a similar approach regarding agricultural GHGs. Nitrous oxide is a powerful GHG that also depletes the ozone layer, and is emitted almost entirely by agricultural soil management. It accounts for about 10% of the state's agricultural GHG emissions. New York could impose a modest fee on synthetic fertilizer that could both directly encourage fertilizer use efficiency and shifts to organic fertilizer and provide funds to assist this transition (although it may not be appropriate until the current price spike driven by Russia's invasion of the Ukraine abates). More sophisticated fee schemes could provide a base rate with discounts for enhanced-efficiency fertilizers that convert less to nitrous oxide. Recent studies by Cornell and others suggest that great improvements in fertilizer management are possible—and profitable—with similar reductions in nitrous oxide emissions.[31] Similarly a fee on methane would internalize the pollution cost (primarily pollution cost associated with beef and dairy along with oil and gas and food waste), and could fund efforts to accelerate feed and manure management changes.

In addition to policies directed at farmers, governments at all levels can use their enormous procurement power to support the production of climate-friendly food.[32] The scoping plan appropriately encourages purchasing of and education about climate-friendly products, but so far without many specifics.[33] Several bills pending in the State Legislature (S. 740, S. 7534) would mandate agencies or encourage municipalities to use their purchasing power for foods with a lower climate footprint. Governor Hochul in her State of the State address proposed to create green procurement standards for State agencies and authorities. The Cool Food Pledge, which establishes a methodology for calculating the GHG impact of foods and proposes a reduction target of at least 25% by 2030 relative to a 2015 baseline,[34] and the Good Food Purchasing Program, which commits governments to purchase food that meets five values including environmental sustainability,[35] offer good models for this approach. New York City has recently adopted these programs; other cities in New York State as well as the State itself could do the same.

While the draft scoping plan notes that "consumer … decision-making is key," it then only mentions the purchase of "new passenger vehicles and heating systems."[36] Thus, the scoping plan unfortunately misses an important opportunity to include consumer dietary decisions among actions individuals can make to help curb climate change. These can be some of the most significant opportunities for individuals to reduce their personal carbon footprint.[37]

Finally, New York should consider combining financial and other incentives with regulation. As noted above, a small percentage of New York dairies account for nearly 70% of New York's dairy cow population and are responsible for the vast majority of associated methane emissions. Most of these CAFOs

[26] *See* CLCPA Draft Scoping Plan, *supra* note 8, at 214–22 (Strategies AF11, AF12, AF14, AF16).

[27] Ranjith P. Udawatta & Shibu Jose, *Agroforestry Strategies to Sequester Carbon in Temperate North America*, 86 Agroforestry Sys. 225 (2012), https://doi.org/10.1007/s10457-012-9561-1.

[28] *See* CLCPA Draft Scoping Plan, *supra* note 8, at 218–20 (Strategy AF13).

[29] *See* Lehner & Rosenberg, *supra* note 1, at 67–68.

[30] *See* Lehner & Rosenberg, *supra* note 1, at 156–59.

[31] *See, e.g.*, Fred Magdoff & Harold Van Es, Building Soils for Better Crops: Ecological Management for Healthy Soils (4th ed. 2021), https://www.sare.org/wp-content/uploads/Building-Soils-for-Better-Crops.pdf; Lehner & Rosenberg, *supra* note 1, at 84–87.

[32] *See* Lehner & Rosenberg, *supra* note 1, at 229–39; *see also, e.g.*, Press Release, U.S. Dept. of Agric., USDA Announces $3 Billion Investment in Agriculture, Animal Health, and Nutrition; Unveils New Climate Partnership Initiative, Requests Public Input (Sept. 29, 2021), https://www.usda.gov/media/press-releases/2021/09/29/usda-announces-3-billion-investment-agriculture-animal-health-and.

[33] CLCPA Draft Scoping Plan, *supra* note 8, at 222–23 (Strategy AF17).

[34] *The Cool Food Pledge*, Cool Food, https://coolfood.org/ (last visited Mar. 11, 2022).

[35] Ctr. for Good Food Purchasing, https://goodfoodpurchasing.org/ (last visited Mar. 11, 2022).

[36] Draft Scoping Plan Overview, *supra* note 21, at 11.

[37] *See* Lehner & Rosenberg, *supra* note 1, at 237–39; World Res. Inst., Creating a Sustainable Food Future: A Menu of Solutions to Feed Nearly 10 Billion People by 2050 (July 2019), https://research.wri.org/sites/default/files/2019-07/WRR_Food_Full_Report_0.pdf; J. Poore & T Nemecek, *Reducing Food's Environmental Impacts Through Producers and Consumers*, 360 Science 987 (2018), https://doi.org/10.1126/science.aaq0216.

already operate under a water pollution permit issued by DEC under the Environmental Conservation Law; this permit could include provisions regarding manure handling that could reduce methane as well as water pollution. DEC also has authority to address air pollution under the Environmental Conservation Law. Manure lagoons at large dairies, in addition to releasing methane, generally emit two toxic gases, ammonia and hydrogen sulfide, over the standard air toxics regulatory threshold of 10 tons per year. The State could focus its financial support on the smaller dairies and use its regulatory authority to speed change in these few largest facilities. (The CLCPA provides that regulations must "include legally enforceable emissions limits, performance standards, or measures or other requirements to control emissions from greenhouse gas emission sources, with the exception of agricultural emissions from livestock," but this only means that DEC is not required to impose such limits under the CLCPA; it does not apply to GHG emissions from manure or limit DEC's pre-existing air pollution mitigation authority.)

The food system affects many other aspects of our economy and GHG emissions. In our homes, restaurants, and stores, we waste about one-third of the food produced, and most of that is dumped in landfills where it rots and releases methane. DEC indicates that landfill methane accounts for about 9% of New York's total GHG emissions. The State's new Food Donation and Food Scraps Recycling Law, effective January 2022, requires that businesses and institutions that generate an annual average of two tons of wasted food per week donate excess edible food to the maximum extent practicable and recycle all remaining food scraps if they are within 25 miles of an organics recycler.[38] The effectiveness of this law is compromised by the unreasonably low distance limit (garbage is often trucked far further to landfills) and dearth of organics recycling facilities (which seems to be in part a result of the small catchment areas created by the distance limit). It also exempts several large food waste generators and it does not apply in New York City, where Mayor Adams recently proposed to suspend the expansion of the City's composting program.[39] By increasing the number of composting and other organics recycling facilities around the state, especially near cities where the food waste is largely generated, and increasing efforts to reduce food waste, the State can make additional progress in reducing methane emissions and, indeed, the plan relies on the complete diversion of organic waste from landfills and incinerators to achieve its goals.

At a time when the President's efforts to tackle the climate crisis are becoming bogged down in politics and may be undercut by the Supreme Court, and Congress's ability to act is thwarted by entrenched interests of industrial agriculture and agrochemical companies, all Americans—not just those in New York—could benefit from New York's leadership enacting policies to help make agriculture more just and a climate solution. New York's policymakers must answer that call.

Peter Lehner is the Managing Attorney of the Sustainable Food & Farming Program at Earthjustice. Nathan Rosenberg is a Visiting Scholar at the Harvard Food Law and Policy Clinic.

LEGAL DEVELOPMENTS

ENERGY

State Supreme Court Rejected Challenge to Easement for Offshore Wind Transmission Cable

The Supreme Court, Suffolk County dismissed a challenge to an easement granted by the Town of East Hampton Town Board for the landing site of a transmission cable from a proposed offshore wind facility. The petitioners argued that the Town Board acted arbitrarily and capriciously by granting the easement prior to Public Service Commission (PSC) approval of the Certificate of Environmental Compatibility and Public Need under Article VII of the Public Service Law in order to "do something" about climate change. The court first concluded that three of the individual petitioners established standing because they were residents along the route of the easement who alleged "dangers and disruptions" from use of the easement. Other petitioners, however, failed to establish standing, including a petitioner described as "a local community organization devoted to preserving the natural beauty and bucolic character of Wainscott." The court found that the organization's "speculative statements about disruptions to the neighborhood" did not constitute an injury different from that experienced by the public at large. In addition, individuals who did not reside on property near the easement did not have standing; the court found that their alleged harm—that digging for the transmission cable could cause a known contaminant plume to migrate toward their property and wells—was not connected simply to use of the easement but to the way the project was connected, an issue considered by the PSC that should have been raised in those proceedings. The court also found that a property owners association and individual residents did not establish standing because alleged harms related to a new substation were conclusory and speculative. After addressing standing, the court proceeded to conclude that granting the easement did not constitute site preparation activity that could only be undertaken after the PSC approved the certificate. The court also rejected the claim that the Town Board was required to comply with New York's State Environmental Quality Review Act (SEQRA) before granting the easement. The court concluded that SEQRA did not apply to the project because the project was subject to environmental review pursuant to the Article VII process. In addition, the court found that the petitioners failed to state a claim under New York's General Municipal Law § 51 for fraudulent

[38] N.Y. ENV'T CONSERV. LAW §§ 27-2201–27-2219.

[39] Anne Barnard, *New York Was Set to Expand Composting. Now It's on the Chopping Block.*, N.Y. TIMES (Feb. 23, 2022), https://www.nytimes.com/2022/02/17/nyregion/nyc-budget-composting-adams.html.

or illegal use of public property. *Citizens for the Preservation of Wainscott, Inc. v. Town Board of East Hampton*, No. 601847/2021 (Sup. Ct. Suffolk County Feb. 24, 2022).

State Supreme Court Upheld Schoharie Town Board's Determination That Solar Energy System Was Inconsistent with Comprehensive Plan

The Supreme Court, Schoharie County upheld the Town of Schoharie Town Board's denial of a special use permit for construction of a solar energy system. The court described its review of the Town Board's determination as "quite deferential," given that the Town's Solar Energy Systems law gave the Board discretion to grant or deny the permit for the type of solar energy system at issue and did not limit the Town Board to consideration of certain factors. The court also rejected, as a preliminary matter, the petitioner's contention that the Town Board erred by considering the project's consistency with the Comprehensive Plan. The court noted that consideration of the Comprehensive Plan was required by both the Solar Energy Systems law and the Town's general zoning law. The court then found that the record "firmly" supported the finding that the location, size, and character of the proposed project was inconsistent with the Comprehensive Plan. The court cited the Comprehensive Plan's goal to maintain the Town's "rural, small town character"; its emphasis on the Town's "scenic beauty," including its "agricultural landscape"; and its specific identification of the road on which the project would be located as an important scenic location. The court further found that record evidence supported the Board's finding that solar panels would be visible from some locations, even with mitigation, and found that under the circumstances there was no error in the finding that "this new, large-scale commercial development would be out of place and inconsistent with the character of the area where it is proposed to be located." The court also found that petitioner did not establish that the Town Board was "unduly influenced by, or relied entirely on, community pressure." *Matter of Bliss Solar 1, LLC v. Town of Schoharie Town Board*, Index No. 2021-320 (Sup. Ct. Schoharie County Feb. 2, 2022).

HAZARDOUS SUBSTANCES

Federal Court Rejected Counterclaims Against Town of Islip in Park Waste Dumping Case

The federal district court for the Eastern District of New York dismissed counterclaims asserted by four defendants in the Town of Islip's lawsuit seeking to recover costs for the cleanup of Roberto Clemente Park in connection with illegal dumping of hazardous materials. The Town asserted claims under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA); the Racketeer Influenced and Corrupt Organizations Act (RICO); and State law. The four defendants—who allegedly transported construction waste to the park—asserted First Amendment retaliation, Fourteenth Amendment substantive and procedural due process, and conspiracy claims under 42 U.S.C. § 1983, as well as claims for indemnification/contribution. They alleged that the Town named them as defendants "solely to punish them for filing and pursuing" a lawsuit against the Town. The court found that the *Noerr-Pennington* doctrine protected the Town from the alleged violations of the First and Fourteenth Amendments because no reasonable jury could find that the Town's lawsuit fell within the "sham exception" to the doctrine since the Town had "an objectively reasonable reason to commence litigation," given CERCLA's broad imposition of strict liability and evidence connecting the four defendants with the person they asserted was responsible for the waste dumping (the son of two of the four defendants). The court also found that the absence of an underlying constitutional violation prevented a finding of conspiracy or, alternatively, that the intracorporate conspiracy doctrine would bar the conspiracy counterclaims since the individual counterclaim defendants were part of, or agents of, the Town and were not "motivated by an independent personal stake." The court concluded that the four defendants were barred by CERCLA from seeking indemnification or contribution for CERCLA liability because the Town had resolved its liability to New York State in an administrative settlement. The court also concluded the four defendants could not seek indemnification and contribution against the Town for non-CERCLA liability because they did not have liability exposure to other co-defendants and alleged tortfeasors may not seek contribution or indemnification from a claimant in the absence of a direct tort claim against the tortfeasor by another party. *Town of Islip v. Datre*, 2022 U.S. Dist. LEXIS 26044 (E.D.N.Y. Feb. 14, 2022). [*Editor's Note:* This case was previously covered in the June 2017 issue of *Environmental Law in New York*.]

LAND USE

Court of Appeals Said State Law Authorizing Town Regulation of Firearm Discharges Did Not Authorize Bow Restrictions

The New York Court of Appeals ruled that Town Law § 130(27)—which authorizes certain towns to regulate discharge of "firearms"—did not authorize the Town of Smithtown to regulate discharge of bows. The Court of Appeals concluded that when construed in accordance with its usual and commonly understood meaning, the term "firearm" did not encompass a "bow." The court noted that it was not considering the question of whether a town would have other authority to regulate the setback distance for discharge of bows under its municipal home rule authority or whether the Environmental Conservation Law would preempt such regulations. (The court said the Town of Smithtown had conceded that its Town Code provision would be invalid absent specific authority under Town Law § 130(27) and could not assert a contrary argument at this point.) *Hunters for Deer, Inc. v. Town of Smithtown*, 2022 N.Y. LEXIS 66 (N.Y. Feb. 10, 2022). [*Editor's Note:* This case was previously covered in the December 2020 issue of *Environmental Law in New York*.]

Appellate Division Found That Adjacent Property Owners Had Standing to Challenge Zoning Determination That Allowed Continued Concrete Manufacturing

Reversing the Supreme Court, Putnam County, the Appellate Division, Second Department held that owners of property adjacent to property where a concrete batch plant had operated had standing to challenge a Town of Kent Zoning Board of Appeals (ZBA) determination that vacated the Town Building Inspector's determination that the property could no longer be used for concrete manufacturing. The Building Inspector issued the determination in May 2017, finding that a 1948 variance permitting employment of more than five people in operating a concrete manufacturing plant was not a use variance that ran with the land and that concrete manufacturing was no longer permitted due to an extended discontinuance of the nonconforming use after an accident during renovations at the plant. The Second Department found that all petitioners had standing to challenge the ZBA decision vacating this determination. In one proceeding, the petitioners had alleged environmental injuries to a private lake owned by the lead petitioner that was directly across from the subject property. They also alleged other injuries different from those suffered by the public at large and within the zone of interests of zoning laws such as interference with recreational activities in and around the lake and property impacts such as noise, truck traffic, dust, and pollutants. The Second Department said the other set of petitioners similarly had sufficiently alleged injuries within the zone of interests protected by zoning laws. The court also found that the lead petitioner did not have to establish organizational standing, given its ownership of the lake. The Second Department affirmed the Supreme Court's rejection of petitioners' motion to compel the ZBA to include in the record documents from appeals of earlier determinations by the Building Inspector. The Second Department found that those earlier appeals were not part of the "proceedings under consideration" and that materials concerning the earlier appeals had not been presented to the ZBA in connection with the current proceeding. *Matter of Veteri v. Zoning Board of Appeals of Town of Kent*, 2022 N.Y. App. Div. LEXIS 1004 (2d Dept. Feb. 16, 2022).

Appellate Division Affirmed Decision That Town Supervisor's Conflict of Interest Rendered Concrete Manufacturing Restrictions Invalid

In a related proceeding concerning the continued operation of a concrete batch plant in the Town of Kent, the Appellate Division, Second Department agreed with the Supreme Court, Putnam County that the Town Supervisor's conflict of interest and incomplete recusal invalidated a local law restricting concrete manufacturing in the Town. The law at issue prohibited production and manufacture of concrete and operation of a concrete products plant in all but one zoning district and required such activities in other zoning districts to terminate by amortization within two years, with the possibility of limited extensions. The Town Supervisor was a member of the lead petitioner (the owner of the private lake) in the related proceeding. She recused herself from voting on the local law but continued to preside over public hearings, engaged in discussions with the concrete manufacturer's representative and the public, participated in executive session, and voted on certain motions related to the law. The Second Department found that as a member of a petitioner in a proceeding seeking the plant's closure in its current location, the Town Supervisor "had an interest in the local law which went beyond mere expressions of personal opinion or an interest shared by the majority of property owners in the town." The Second Department further found that her limited recusal was "insufficient to remedy the appearance of impropriety which arose from her participation in the public hearings." The Second Department also noted that the Town Supervisor had not obtained an advisory opinion from a local ethics board concluding that her participation was appropriate. *Matter of Titan Concrete, Inc. v. Town of Kent*, 2022 N.Y. App. Div. LEXIS 991 (2d Dept. Feb. 16, 2022). [*Editor's Note:* This case was previously covered in the June 2019 issue of *Environmental Law in New York*.]

Appellate Division Reinstated Private and Public Nuisance Claims in Connection with Daytime Beach Driving Regulation by Village and Town of Southampton

The Appellate Division, Second Department found that the Supreme Court, Suffolk County should not have granted summary judgment to the Village of Southampton on private nuisance and public nuisance claims brought by a residential property owner in connection with regulation by the Village of Southampton that allowed exceptions to the general prohibition on summer daytime beach driving and parking. The Second Department also reinstated a private nuisance claim for damages against the Trustees of the Freeholders and Commonalty of the Town of Southampton (Town Trustees) in connection with a similar (but now repealed) regulation. The Second Department found that photographs of the beach area and the affidavits of the plaintiff and her family raised triable issues of fact as to whether the driving and parking "was of an unreasonable character" so as to constitute a private nuisance. In addition, the Second Department found there were triable issues of fact as to whether these activities endangered the health and safety of the public or the beach itself so as to constitute a public nuisance. (The Second Department concluded, however, that the Town Trustees' repeal of their regulation rendered the public nuisance claim and certain other claims against the Town Trustees academic.) In addition, the Second Department found that the Supreme Court "improvidently exercised its discretion" to search the record and award the Village summary judgment on the plaintiff's claim that the Village Code provision was void because it violated State regulations on coastal erosion management. The Second Department affirmed other aspects of the Supreme Court's decision, including the dismissal of a mandamus claim that improperly sought to direct how the New York State Department of Environmental Conservation regulated coastal areas. In addition, the Second Department found that equal protection claims against the Village and an unconstitutional taking claim against the Village and Town Trustees were barred by the doctrine of res judicata based on decisions in lawsuits filed in 1988 and 2005. *Thomas v. Trustees of Freeholders & Commonalty of Town of Southampton*, 2022 N.Y. App. Div. LEXIS 872 (2d Dept. Feb. 9, 2022).

OIL SPILLS & STORAGE

Boy Scout Camp Found to Be in Violation of Petroleum Bulk Storage Regulations

New York State Department of Environmental Conservation (DEC) Commissioner Basil Seggos granted DEC staff's motion for default judgment and found that a Boy Scouts of America council had violated the petroleum bulk storage regulations at a camp owned by the council in the Town of Long Lake, Hamilton County. The violations included failures to comply with registration requirements, failure to have a surface coating on two aboveground storage tanks to prevent corrosion and deterioration, failures to mark tanks properly, and failures to conduct monthly inspections and maintain inspection records. The Commissioner assessed a civil penalty of $16,200 but suspended $11,200 of the penalty contingent upon the respondent's compliance with the order. Among the steps the respondent must take is to submit documentation of its compliance with the regulations, including by either submitting photographs showing that corrosion areas on the tanks had been coated or by providing documentation that the tanks had been closed in accordance with DEC regulations. *In re Patriots' Path Council, Inc.*, 2022 N.Y. ENV LEXIS 11 (DEC Feb. 10, 2022).

SEQRA/NEPA

Third Department Found Impermissible Segmentation in SEQRA Review of Zoning Map Amendments

The Appellate Division, Third Department found that the City of Saratoga Springs improperly segmented its environmental review of a zoning map amendment from a hospital's future plans to develop the affected parcel with a medical office building and parking. The court noted that the zoning map amendment "would be the green light to reignite development plans" and found that the hospital's potential redevelopment of the parcel "was not so attenuated from the zoning map amendment" that segmentation would be permissible. The Third Department therefore reversed the portion of judgment of the Supreme Court, Saratoga County that dismissed the petitioners' cause of action asserting SEQRA violations. The Third Department affirmed, however, the Supreme Court's determinations that the zoning map amendments were aligned with the comprehensive plan and did not constitute illegal spot zoning. The Third Department also rejected the petitioners' contention that City Council members' receipt of campaign contributions from representatives of the hospital constituted a conflict of interest requiring annulment of the zoning map amendments. *Matter of Evans v. City of Saratoga Springs*, 2022 N.Y. App. Div. LEXIS 1086 (3d Dept. Feb. 17, 2022).

Appellate Division Dismissed Fire District's Challenges to Assisted Living Facility Zoning Approvals

The Appellate Division, Second Department affirmed the dismissal of two Article 78 proceedings brought by the Greenville Fire District to challenge area variances, a special permit, and a SEQRA conditioned negative declaration for an assisted living facility in the Town of Greenburgh. In the first proceeding, the Second Department agreed with the Supreme Court, Westchester County that claims challenging the area variances and conditioned negative declaration were time-barred. The appellate court found that the Supreme Court properly determined that the challenge to the conditioned negative declaration became ripe for review when the Town of Greenburgh Zoning Board of Appeals (ZBA) granted the area variances and that the 30-day limitations period began to run upon the filing of the ZBA meeting results with the Town Clerk (and not on a later date when the ZBA filed a certification of the decision setting forth its findings). In the second proceeding, the Second Department found that the Fire District's second challenge to the conditioned negative declaration was barred by the doctrine of res judicata. The Second Department also found that the Fire District did not have standing to challenge the special permit granted by the Town Board for the assisted living facility because the alleged injury—the need for additional personnel and equipment due to an anticipated increase in the number of emergency calls related to the facility—was outside the zone of interests protected by the zoning ordinance. The Second Department further found that allegations regarding road safety hazards for emergency vehicles were conclusory and speculative and thus insufficient to establish standing. *Matter of Greenville Fire District v. Zoning Board of Appeals of Town of Greenburgh*, 2022 N.Y. App. Div. LEXIS 1040 (2d Dept. Feb. 16, 2022); *Matter of Greenville Fire District v. Town Board of Greenburgh*, 2022 N.Y. App. Div. LEXIS 1031 (2d Dept. Feb. 16, 2022).

Organizations and Nearby Residents Lacked Standing to Challenge Assisted Living Facility Special Permit

In a separate proceeding brought by the Council of Greenburgh Civic Associations, Edgemont Community Council, and individual petitioners to challenge the special permit for the assisted living facility, the Appellate Division, Second Department agreed that the petitioners failed to establish standing. The Second Department found that the individual petitioners failed to establish that their properties were sufficiently close to the proposed development to give rise to an inference of damage or injury. The appellate court also found that the petitioners' "generalized allegations" of a potential public safety hazard "failed to set forth an actual injury distinct from that suffered by the public at large." The court further found that the organizational petitioners lacked standing because their standing was dependent on the standing of the individual petitioners. *Matter of Council of Greenburgh Civic Associations v. Town Board of Greenburgh*, 2022 N.Y. App. Div. LEXIS 1015 (2d Dept. Feb. 16, 2022).

State Supreme Court Said Challenge to Negative Declaration for Open Restaurant Program Was Ripe

The Supreme Court, New York County denied New York City's motion to dismiss an Article 78 proceeding challenging the negative declaration issued under SEQRA and City Environmental Quality Review for a proposed permanent version of the program that allowed restaurants and bars to expand to certain outdoor areas during the COVID-19 pandemic. The court rejected the City's contention that the petition was not ripe for review. The court found that the petitioners alleged "concrete injuries, environmental impacts including vermin, noise, and garbage, currently occurring," and concluded that while the City Council might review the permanent open restaurant proposal, there was no guarantee that the Council would act and the temporary open restaurant program was continuing in the interim. The court found that in this case, it was the SEQRA process, and not the zoning action, that inflicted the alleged injuries. The court therefore concluded that dismissal on ripeness grounds was not appropriate. *Matter of Arntzen v. City of New York*, 2022 N.Y. Misc. LEXIS 461 (Sup. Ct. New York County Feb. 1, 2022).

SOLID WASTE

Solid Waste Management Company and President/CEO Ordered to Pay $25,000 Penalty

DEC Commissioner Seggos found that the operator of a solid waste management facility in Mount Vernon and its president and chief executive officer (CEO) had violated the facility's permit by failing to submit annual monitor fees in the amount of $37,800 for fiscal year 2020-2021. A processing and transfer station for acceptance and storage of yellow grease was operated at the facility. The Commissioner ordered that the respondents were jointly and severally liable for a civil penalty of $25,000. The Commissioner noted that it was "well-settled law" that officers of corporations could be held liable for environmental violations without piercing the corporate veil where their actions or omissions result in or contribute to the violations. In this case, the president and CEO "had direct responsibility for compliance with the monitor fee requirement" of the solid waste management facility permit. *In re NYC Oil Corp.*, 2022 N.Y. ENV LEXIS 5 (DEC Jan. 31, 2022).

DEC Commissioner Ordered Trucking Company to Pay $50,000 Penalty, Denying Staff Request for Higher Penalty

DEC Commissioner Basil Seggos imposed a $50,000 civil penalty on a freight shipping and trucking company that disposed of solid waste at an unauthorized facility in Earlton, Greene County. The respondent hauled 161 loads of "overs"—topsoil, rocks, bricks, processed paint wood, electrical wiring, piping, foam, tiles, metals, and plastics—at the site over the course of five months in 2017. The respondent did not answer or appear in the proceeding, but the Commissioner agreed with the administrative law judge (ALJ) that the pleadings and papers provided sufficient facts to enable a determination that the DEC staff had a viable claim and that default judgment was warranted. The Commissioner declined, however, to impose the $122,000 civil penalty that DEC staff requested in their motion papers. The Commissioner noted that the complaint itself sought a civil penalty of "no less than" $50,000 and that the penalty assessed in a default judgment could not exceed the amount demanded in a complaint, absent notice to the respondent. The Commissioner concluded that the use of the phrase "no less than" in the complaint did not provide adequate notice since the complaint could refer to any penalty amount between $50,000 and the maximum penalty of more than $1 million calculated by the ALJ. *In re Merrick Trucking Corp.*, 2022 N.Y. ENV LEXIS 2 (DEC Jan. 31, 2022).

TOXIC TORTS

Federal Court Approved Settlement in Hoosick Falls PFOA Class Action

The federal district court for the Northern District of New York granted final approval of a settlement agreement in a class action in which plaintiffs asserted claims based on the alleged presence of perfluorooctanoic acid (PFOA) in the Village of Hoosick Falls municipal water system, in private wells, and/or in the plaintiffs' blood. The settlement provided that the three settling defendants would pay $65,250,000 into a settlement fund, with the funds allocated to four classes: (1) a Municipal Water Property Settlement Class; (2) a Private Well Water Property Settlement Class; (3) a Nuisance Settlement Class; and (4) a Medical Monitoring Settlement Class. The court found the settlement agreement to be fair, reasonable, and adequate pursuant to Federal Rules of Civil Procedure (FRCP) 23(e), noting that "the reaction of the Settlement Classes has been overwhelmingly positive, which weighs strongly in favor of the Settlement's fairness." In reaching its determination that the settlement's method of distributing benefits to the settlement classes was fair, reasonable, adequate, and efficient, the court said more than 2300 claims had been filed during the enrollment period; the court found that this "substantial" claims rate was "a testament to the fairness" of the settlement. The court also certified the four settlement classes for purposes of settlement only, finding that they met the requirements for class certification under FRCP 23(a) and (b)(3). In addition, the court approved the requested attorneys' fee award of $12,397,500 and reimbursement of $1,040,817 for expenses incurred in connection with the litigation. The court found that the attorneys' fees were reasonable considering the magnitude and complexity of the litigation, the risk of the litigation, and the quality of representation on both sides. The court also found that the 19% fee award compared favorably with class counsel fees in other Second Circuit cases. In addition, the court awarded each of 10 class representative plaintiffs $25,000 for their "commendable work" in achieving the settlement. The court found that the settlement agreement's release was valid and enforceable and enjoined the assertion of contribution or similar claims against the released parties. The court dismissed all released claims with prejudice, including all claims asserted against the settling defendants in the Second Amended Complaint. *Baker v. Saint-Gobain Performance Plastics Corp.*, No. 1:16-cv-00917 (N.D.N.Y. Feb. 4, 2022). [*Editor's Note:* This case was previously covered in the May 2017 and October 2021 issues of *Environmental Law in New York*.]

WATERS

Federal Magistrate Again Recommended Dismissal of Safe Drinking Water Act Endangerment Claim in Connection with Septic Systems in Long Island Parks

A magistrate judge in the federal district court for the Eastern District of New York recommended denial of reconsideration of the court's decision to dismiss a Safe Drinking Water Act (SDWA) claim in a citizen suit alleging that septic systems operated by the New York State Office of Parks, Recreation, and Historic Preservation at parks on Long Island violated the endangerment standard under the SDWA and the U.S. Environmental Protection Agency's (EPA's) implementing regulations. The magistrate issued reports and recommendations (R&Rs) in 2019 and 2021 recommending the SDWA claim be dismissed; after the 2021 R&R, EPA filed a Statement of Interest that objected to the R&Rs' interpretation that the SDWA's endangerment standard did not apply to the "Upper Glacial Aquifer" underlying Long Island because the aquifer was not a drinking water system since no drinking water wells existed downgradient from the parks' septic systems. The district court construed EPA's Statement of Interest as a motion for reconsideration. In its report and recommendation on the motion for reconsideration, the magistrate found that EPA's position did not meet the standard for reconsideration because it was the same interpretation that the plaintiffs articulated, and EPA did not present controlling case law or new evidence the court overlooked. The magistrate noted that EPA conceded the SDWA might leave "'some ambiguity' as to whether the endangerment requires a showing of a possible effect on a public water system." The magistrate also found that EPA's argument regarding EPA's implementing regulations did not meet the standard for reconsideration. *Peconic Baykeeper, Inc. v. Kulleseid*, 2022 U.S. Dist. LEXIS 18384 (E.D.N.Y. Feb. 1, 2022). [*Editor's Note:* This case was previously covered in the July 2019 and January 2022 issues of *Environmental Law in New York*.]

$6,840 Penalty Assessed for Failure to Submit Discharge Monitoring Reports for Mid-2019 Through End of 2020

DEC Commissioner Seggos granted DEC staff's motion for default judgment and found that a respondent violated DEC regulations and the State Pollutant Discharge Elimination System (SPDES) Multi-Sector General Permit (MSGP) for Stormwater Discharges Associated with Industrial Activity (GP-0-17-004) by failing to file complete semiannual discharge monitoring reports for three reporting periods extending from July 1, 2019 through December 31, 2020. The respondent obtained coverage under the SPDES MSGP in 2018 for its land transportation and/or warehousing facility in Bay Shore on Long Island, where it conducted repair, maintenance, and storage of vehicles and trailers. The Commissioner assessed a $6,840 civil penalty, declining to assess a higher penalty requested by DEC staff in their motion papers. (The Commissioner noted that a penalty assessed in a default judgment cannot exceed the amount demanded in the complaint, absent notice to the respondent that a greater penalty would be sought.) *In re XPO Logistics Freight, Inc.*, 2022 N.Y. ENV LEXIS 4 (DEC Feb. 9, 2022).

NEW YORK NEWSNOTES

Chapter Amendments Made Changes to Environmental Laws Enacted in 2021

In January and February 2022, Governor Kathy Hochul signed chapter amendments modifying a number of environmental laws enacted in 2021:

- **Bioheating fuel mandate:** An Environmental Conservation Law provision establishing statewide mandates for use of bioheating fuel was amended to provide that "Renewable Hydrocarbon Diesel" could be produced anywhere in North America, not just domestically. The amendments also eliminated a requirement for DEC to promulgate regulations to enforce assurances that bioheating fuel will not void manufacturers' warranties unless the bioheat provider provides its own warranty. (Chapter 5)
- **Effluent limitations in Nassau and Suffolk Counties:** Chapter amendments modified the standard applied to establishing effluent limitations in special ground water protection areas or in Nassau or Suffolk Counties where discharges will impact marine waters within 10 years. Amendments in 2021 required application of "best available technology," and the chapter amendments added an economic feasibility and cost effectiveness requirement. (Chapter 8)
- **Flame retardant chemicals in furniture, mattresses, and electronic casings:** In December 2021, New York enacted the Family and Fire Fighter Protection Act, which will prohibit sales of upholstered furniture, mattresses, and electronic displays containing certain flame retardant chemicals that are halogenated, organophosphorus, organonitrogen, or nanoscale chemicals. The law also will bar custom upholsterers from using components that contain the regulated chemicals. The chapter amendments extended the sales prohibitions' effective date from January 1, 2024 to December 1, 2024 and the upholsterer prohibition

effective date from January 1, 2023 to December 1, 2025. As amended, the prohibitions apply where the regulated chemicals are present "[a]t or above levels set by [DEC] in regulation" or are "intentionally added." The amendments also add a new section that provides that sellers of covered products will not be found to have violated the law if they can show good-faith reliance on a manufacturer's certificate of compliance. (Chapter 11)

- **Pesticide data compilation:** A 2022 law further amended requirements for DEC to compile data on pesticide applications. DEC must compile and report data by EPA registration number and active ingredient and post annual reports on its website by December 31. (Chapter 30)
- **State geological trail:** Amendments to a 2021 law regarding the establishment of a State geological trail gave the DEC Commissioner more discretion regarding the designation of sites of geological significance and the establishment of the State geological trail. The amendments also pushed back the effective date of the law by one year. (Chapter 31)
- **Glass recycling study:** A 2022 law expanded the scope of a study on alternative municipal uses for recycled glass to include glass collection, processing, and recycling. The law also gave DEC additional time—until December 31, 2023—to complete the study. (Chapter 32)
- **Small plastic bottle hospitality personal care products:** The compliance dates for the prohibition on large hotels providing small plastic bottles containing a hospitality personal care product was extended from January 1, 2024 to January 1, 2025. The deadline for small hotels was extended from January 1, 2025 to January 1, 2026. (Chapter 33)
- **Seaweed cultivation in Suffolk County:** A 2021 law establishing kelp cultivation in Gardiner's and Peconic Bays in Suffolk County was amended to expand the authorized cultivation to include other types of seaweed. The permitting process was also revised. (Chapter 34)
- **Lead warnings for seasonal and decorative lighting cord casing:** The chapter amendments moved the warning requirement from Section 1376-b of the Public Health Law to Section 389-s of the General Business Law. The amendments also reduced the penalty to $500 for a violation (from $1,000) and limited the penalty provision to manufacturers. (Chapter 40)
- **Supermarket excess food donation:** The amendments to the Food Donation and Food Scraps Recycling Law clarified that large supermarkets (more than 10,000 square feet) that do not meet the scrap generation weight threshold to qualify as "food scrap generators" are subject to a separate provision that provides that they "shall from time to time make excess food available to food relief organizations."
- **Storm hardening and system resiliency plans for utilities:** A 2021 law required that electric utilities submit climate change vulnerability studies to the Public Service Commission. The chapter amendments modified and clarified the process for submission and review of the plans, as well as the cost recovery provisions. (Chapter 45)
- **Emerging contaminants:** The chapter amendments extended by 90 days the required timeframe for adopting a "first list" of emerging contaminants. The amendments also removed 17 substances from the list of substances required to be on the "first list" of emerging contaminants. Instead, the law established a longer timeline (to January 1, 2024) for the Health Commissioner to determine whether to include 14 of the chemicals as emerging contaminants. The amendments also provided that the Commissioner could determine not to include two of the substances as emerging contaminants (but without a specified timeframe) and removed a substance (chlorate) from the law's scope. (Chapter 69)
- **Pesticide use at children's camps:** The amendments modified the 2021 law prohibiting application of pesticides on playgrounds or athletic or playing fields at overnight and day camps. The changes included eliminating the authority of the Health Commissioner to exempt camps for which use of a pesticide alternative is not practicable and eliminating a provision to allow an emergency application of a pesticide if a camp does not receive a response to a request for emergency application within 24 hours. (Chapter 71)
- **Airport health impacts study:** The chapter amendments gave the New York State Department of Health (instead of DEC) the lead role in conducting a study describing the impacts of JFK and LaGuardia airports on noise and human health. The amendments also modified the law to focus on noise and human health—asthma, in particular—and added a requirement that the report on the study's findings include all raw and aggregate data. (Chapter 92)
- **Soil Health and Climate Resiliency Act:** The chapter amendments grant the Commissioner of the Department of Agriculture and Markets and the Soil and Water Conservation Committee the authority to adopt regulations to implement the 2021 Soil Health and Climate Resiliency Act. (Chapter 102)
- **Zero-emission passenger cars and trucks:** The 2021 law set a goal for 100% of new passenger cars and trucks to be zero-emissions by 2035. The amendments made the New York State Energy Research and Development Authority (instead of DEC) responsible for development of a market development strategy. (Chapter 109)
- **State Administrative Procedure Act public hearings:** The amendments increased the number of persons that must join a petition to trigger a requirement for a public hearing

on a proposed rule. For DEC, the number was increased from 125 to 750. Amendments also exempted rules adopted on an emergency basis from the public hearing requirement and made the hearing optional if the petition is received less than 30 days before the comment deadline. In addition, the law will take effect January 1, 2023 (instead of January 1, 2022), and agencies may require that agency-generated forms be used to submit petitions for hearings. (Chapter 116)

- **Access to locally produced, healthy foods:** In response to the governor's concerns about the potentially duplicative role of a task force created by a 2021 law to make recommendations on improving urban and rural consumer access to locally produced, healthy foods, the chapter amendments modified the law to establish instead an "advisory group" that will identify strategies and opportunities to expand access for underserved, nutritionally deficient urban and rural communities to healthy, locally produced food in New York State. (Chapter 124)
- **MTA strategic action plan for improving bicycle and pedestrian access:** Changes included requiring the Metro-North Rail Commuter Council, the Long Island Rail Road Commuter's Council, and the New York City Transit Advisory Council to submit their recommendations on improving bicycle and pedestrian access to the Metropolitan Transportation Authority by June 1, 2022 and every five years thereafter. (Chapter 125)
- **Aquatic invasive species inspection stations in the Adirondack Park:** Changes to the law authorizing DEC to establish the inspection stations included clarification that owners or operators of motorized watercraft are required to obtain a certification or certificate to ensure compliance with requirements to prevent spread of aquatic invasive species only when the stations are open for operation. The amendments also require DEC to annually review data from the inspection stations to identify improvements to reduce the spread of aquatic invasive species. (Chapter 126)
- **Potable water testing for lead in schools:** The amendments authorize the Health Commissioner to grant waivers from lead testing requirements for school buildings if the school district "has substantially complied with the testing requirements and has been found to be below lead levels as determined by regulations … for such buildings." The amendments also require that schools use other state and federal funding sources for testing and remediation prior to using funds allocated to a school district by the Board of Cooperative Educational Services under Education Law § 1950 or apportionments to school districts for school building purposes and testing and filtering of potable water systems under Education Law § 3602. (Chapter 130)

Requirements for Conditional Adult-Use Cannabis Licenses Include Environmental Sustainability

On February 22, 2022, Governor Hochul signed legislation establishing temporary conditional licenses for adult-use cannabis cultivators and processors (Chapter 18). The law directed the Office of Cannabis Management to set out terms and conditions for the awarding and maintenance of such licenses, which must include a requirement that licensees agree to participate in an environmental sustainability program. The provisions establishing the conditional licenses are codified at Sections 68-c and 69-a of the Cannabis Law.

DEC Issued Guidance on Operational Requirements for L3 Landfills Permitted Before November 2017

In the February 23, 2022 issue of the *Environmental Notice Bulletin*, DEC announced that it had finalized a program policy on "Guidance for Renewing or Modifying an Existing Part 360 Permit for Construction and Demolition Debris Landfills Three Acres or Less in Area." The guidance applies to existing construction and demolition debris landfills that are three acres or less in area and were permitted under the 6 N.Y.C.R.R. Part 360 regulations in effect before November 4, 2017. The policy sets forth options for such "L3 Landfills" at the time of permit expiration, renewal, or modification. Until an L3 Landfill's current permit expires or is renewed or modified, the landfill can operate with the existing low permeability liner system. There are three options for an L3 Landfill at the time of permit expiration, renewal, or modification: (1) it may continue to operate with the existing low permeability liner system if waste accepted for disposal is limited to tree debris; uncontaminated soil and rock from land clearing, utility line maintenance, and seasonal or storm-related cleanups; and recognizable uncontaminated concrete and concrete products, asphalt pavement, brick, glass, soil, and rock; (2) the facility can construct a liner system that meets the requirements of the current Part 363 and can continue to accept a mixed construction and demolition debris waste stream; or (3) the facility can initiate closure of the facility. The policy is available at https://www.dec.ny.gov/regulations/124062.html.

DEC Finalized Methane and VOC Regulations for Oil and Gas Sector

In the February 16, 2022 issue of the *NYS Register*, DEC published notice of its adoption of regulations to control emissions of methane and volatile organic compounds (VOCs) from the oil and natural gas sector. The regulations are primarily codified in a new Part 203 of DEC's regulations. The regulations—which have a compliance date of January 1, 2023—impose requirements on new and existing oil and natural gas well activities, natural gas gathering lines, natural gas transmission pipelines and compressor stations, natural gas underground storage facilities, and the "city gate" (a transfer point between a natural gas transmission system pipeline company/operator and a distribution system company/operator). The new Part 203 does not apply to distributing gas utilities or equipment and components located downstream of a city gate. The regulations prohibit venting natural gas to the atmosphere from some components and include leak detection and repair (LDAR) requirements, with the frequency of required inspections varying from bimonthly to semiannually depending on the type of activity, with less frequent inspections required if an alternative method offering continuous monitoring is used. The regulations generally require that leaks be repaired within 30 days and reinspected within 15 days of repair. The regulations provide, however, for delays of repairs and replacements if certain conditions are met. Blowdown activity (releases of natural gas from pipelines or compressor stations) requires notification if the amount is greater than 10,000 cubic feet, whether the blowdown is planned or unplanned. Baseline component inventories from regulated sources are due to DEC by March 31, 2023, or by March 31 of the year following initiation of operation. Records must be maintained for five years. DEC said the regulations would serve three functions: reducing methane emissions in support of the goals and requirements of the Climate Leadership and Community Protection Act; reducing VOC emissions, an ozone precursor; and fulfilling the requirements of the EPA's 2016 Control Techniques Guidelines for the oil and gas industry.

New York State Common Retirement Fund to Divest from Certain Shale Oil and Gas Companies

On February 9, 2022, New York State Comptroller Thomas P. DiNapoli announced that the New York State Common Retirement Fund would restrict investments in 21 shale oil- and gas-producing companies (out of 42 companies reviewed) that the Fund determined had failed to show viable strategies to transition to a low-carbon company. The fund will sell more than $238 million in public equity and debt securities issued by the 21 companies in a manner consistent with fiduciary obligations. The Fund previously divested from 34 oil sands and coal companies that the Fund determined had failed to demonstrate transition readiness. The Fund next will evaluate integrated oil and gas companies. The announcement of the divestment from the shale oil and gas companies said the restrictions were part of the Comptroller's Climate Action Plan and commitment to transition the Fund's investment portfolio to net zero greenhouse gas emissions by 2040.

Audit Found that NYPA Electric Vehicle Programs Failed to Achieve Goals

On February 4, 2022, Comptroller DiNapoli released an audit of the Charge NY, Charge NY 2.0 and Evolve NY programs that found that the New York Power Authority (NYPA) had failed to install electric vehicle (EV) chargers where they were most needed. The audit found that NYPA placed public charging ports in only 32 of the State's 62 counties, and that counties with a high number of EVs have relatively few public charging ports. For instance, Suffolk County has 17% of registered EVs in New York State but only 1.1% of the public charging ports. The audit also found that NEPA did not review and analyze usage data for charger placement or use outreach efforts to encourage its customers to install EV chargers. In addition, NEPA had only installed 29 high-speed chargers as of March 5, 2021, though the EVolve NY program had a deadline of the end of 2019 for stalling 200 high-speed chargers. The report said installation of such chargers was as much as two years behind schedule. The audit report's recommendations included development of a formal process for evaluating new initiatives and programs; development of a formal marketing strategy to increase awareness of the features and benefits of EVs; incorporation of analysis of usage data into NYPA's current EV program; and work with the NYPA customer base to roll out EV charging stations. The audit report is available at https://bit.ly/3tZp9if.

Amendments to New York City Stormwater Rule Established Citywide Construction/ Post-Construction Permitting Requirements

Amendments to New York City's stormwater rule took effect on February 15, 2022. The amended rules apply to discharges from industrial stormwater sources within portions of New York City served by the municipal separate storm sewer system (MS4) as well as discharges of stormwater from "covered development projects" regardless of whether they are in the MS4 area. Covered development projects encompass private or public development activity involving or resulting in an amount of soil disturbance of at least 20,000 square feet or creation of at least 5,000 square feet of impervious surface or roadway maintenance involving 20,000 square feet or more. The New York City Department of Environmental Protection (DEP) described the amendments as part of a unified citywide stormwater policy that will align the construction/post-construction permitting program water quality requirements (15 R.C.N.Y. Chapter 19.1) with stormwater quantity and flow rate requirements in the sewer connection regulations (15 R.C.N.Y. Chapter 31). DEC said the amended stormwater rule would allow for reduction in combined sewer overflows and flooding, increase in green space, greater consistency across stormwater programs, flexibility in design options, and improvements in water quality. DEP also added a NYC Stormwater Manual as an appendix to Chapter 19.1 to provide additional procedural and technical guidance to owners, developers, and applicants.

WORTH READING

Michael B. Gerrard & Edward McTiernan, *Regulation of Polyfluoroalkyl Chemicals in New York*, N.Y.L.J. (Mar. 9, 2022), https://bit.ly/3CvIVG0

Meredith Mandell, *Placing an Emphasis on the "S" in ESG*, NYSBA Journal, Mar./Apr. 2022, at 24, https://nysba.org/nysba-journal-march-april-2022/

Office of the N.Y. State Comptroller, *Green and Growing: Employment Opportunities in New York's Sustainable Economy* (Feb. 2022), https://www.osc.state.ny.us/files/reports/pdf/green-jobs-in-new-york.pdf

Jack Rusk, *Across Currents*, Urb. Omnibus (Feb. 3, 2022), https://urbanomnibus.net/2022/02/across-currents/

David E. Schwartz & Emily D. Safko, *Employment Law Considerations for ESG in Upcoming Proxy Season*, N.Y.L.J. (Feb. 18, 2022), https://bit.ly/36BjB5F

Jared Woollacott et al., State of New York, U.S. Climate Alliance, & RTI Int'l, *Economic Impacts of Investing in Climate Mitigation in New York State Forests and Agriculture: Afforestation, Reforestation, and Manure Methane Capture* (Feb. 2022), https://on.ny.gov/3CHUQ3K

UPCOMING EVENTS

May 2–6, 2022

The RPA Centennial Assembly. For information, see https://rpa.org/events/assembly/2022.

May 2–8, 2022

Circular City Week New York. For information, see https://www.circularcityweek.com/.

May 18, 2022

Independent Power Producers of New York Annual Conference, Albany Capital Center, Albany. For information, see https://www.ippny.org/page/events-3.html.

Matthew Bender & Co., Inc.

Sara Kolb — Product Manager
Linda J. Folkman — Legal Editor

For editorial questions contact Linda Folkman:
by phone at (908) 673-1548, or
by e-mail to *Linda.Folkman@lexisnexis.com*.
For all other questions call 1-800-833-9844.

POSTMASTER: Send address changes to:
Environmental Law in New York
4810 Williamsburg Road, Unit 2, Hurlock, MD 21643.

Arnold & Porter

Environmental Practice Group

Washington
601 Massachusetts Ave., N.W.
Washington, D.C.
20001
(202) 942-5000
Contact:
Brian D. Israel
brian.israel@arnoldporter.com

Denver
370 Seventeenth St., Suite 4400
Denver, CO
80202
(303) 863-1000
Contact:
Matthew J. Douglas
matthew.douglas@arnoldporter.com

Los Angeles
777 South Figueroa St.
Los Angeles, CA
90017-5844
(213) 243-4000
Contact:
Matthew T. Heartney
matthew.heartney@
arnoldporter.com

New York
250 West 55th Street
New York, NY
10019-9710
(212) 836-8000
Contact:
Nelson Johnson
nelson.johnson@
arnoldporter.com

San Francisco
Three Embarcadero Center,
10th Floor
San Francisco, CA
94111-4024
(415) 471-3100
Contact:
Zachary Fayne
zachary.fayne@
arnoldporter.com

Editor: Michael B. Gerrard, michael.gerrard@arnoldporter.com
Managing Editor: L. Margaret Barry, margaret.barry@arnoldporter.com
This monthly newsletter provides general information concerning recent decisions and other developments, and should not be used as a substitute for legal advice in specific situations. Send new, unreported decisions and other information for possible inclusion to the editor. Articles represent the views of their authors and not necessarily those of the publisher or Arnold & Porter Kaye Scholer LLP.

Exhibit B

Ecology, 81(8), 2000, pp. 2314–2330

LONG-TERM IMPACTS OF AGRICULTURE ON SOIL CARBON AND NITROGEN IN NEW ENGLAND FORESTS

Jana E. Compton[1] and Richard D. Boone[2]

Harvard University, Harvard Forest, Petersham, Massachusetts 01366 USA

Abstract. Abandonment and reforestation of agricultural lands has been a major influence on the landscape of eastern North America. Cultivation and soil amendments can dramatically alter soil nutrient pools and cycling, yet few studies have examined the long-term (>50 yr) influence of pasturing and cultivation on soil processes in the forests that develop after abandonment. Twelve forested sites at Harvard Forest in central New England were compared 90–120 yr after abandonment from agricultural use. We measured soil carbon (C), nitrogen (N), and phosphorus (P); light fraction C, N, and $\delta^{15}N$; microbial chloroform-N; net N mineralization and nitrification; nitrification potential; and culturable nitrifiers on sites with differing land-use history and vegetation. The sites had similar soil series and topography but were arrayed along a soil disturbance gradient from permanent woodlots (selective logging but no mineral soil disturbance) to formerly pastured sites (clearcut and grazed but no deep [>10 cm] soil disturbance) to formerly cultivated sites (cleared-with-plow horizon 15–20 cm thick). Mineral soil C (0–15 cm soil depth) was very similar among all sites, but the forest floor C was lower in the cultivated sites than in the woodlots of both stand types. Mineral soil in cultivated sites contained 800 kg N/ha and 300 kg P/ha more than woodlots, a relative increase of 39% for N and 52% for P. The cultivated soils had lower C:N and C:P ratios, largely driven by higher soil N and P. The light fraction appeared to be more sensitive to prior land use than the bulk soil organic matter. The C content and C:N ratio of light fraction were lower in cultivated soils, which suggests that input and/or turnover of organic matter pools of relatively recent origin remain altered for a century after abandonment. Similar $\delta^{15}N$ for the light and heavy fraction organic matter pools in cultivated soils suggests that cultivation accelerates the mineralization of humus N, increasing the proportion of N available for plant uptake, resulting in a convergence of the light and heavy fractions. The N pool in the woodlot soils may have been subject to preferential losses of small amounts of ^{14}N over a longer time period, resulting in a more pronounced divergence between the light fraction (reflecting recent plant inputs) and the mineral-associated heavy fraction (more recalcitrant).

Nitrification was strongly influenced by land-use history, with highest rates in formerly cultivated sites. In contrast, soil net N mineralization and chloroform-N were more strongly influenced by present vegetation. Nitrifying bacteria were relatively abundant in all pastured and cultivated sites; however, only the cultivated hardwood sites, with lowest C:N ratios (16–18), had substantial net nitrification. Historical manure inputs may explain the more rapid soil net nitrification rates, by decreasing soil C:N ratios and thus reducing nitrate immobilization in the mineral soil of cultivated sites. Regionally, 65% of the land area was pastured, and a proportion of the nutrients obtained from grazing was transferred to the cultivated croplands, which comprise ≤15% of the land area. Pastures generally had intermediate nutrient ratios and N transformations but were often more similar to woodlots, which suggests that plowing and amendments, rather than forest clearance, have the greatest impact on soil organic matter and nutrients. The influence of land-use history on soil N and P and nitrification rates was more dramatic in hardwood sites, which indicates that characteristics of the recovering vegetation and/or changes in plant community composition associated with prior land use are important factors in the rate of recovery. Our findings lead to the surprising conclusion that 19th century agricultural practices decreased forest floor nutrient content and ratios, and increased nitrifier populations and net nitrate production for approximately a century after abandonment. Consideration of site history clearly deserves more attention in the design of field experiments, and in our understanding of patterns of element distributions and transformations in dynamic forested landscapes.

Key words: carbon; cultivation; $\delta^{15}N$ natural abundance; land-use history; light fraction organic matter; nitrification; nitrogen mineralization; pastures; phosphorus; reforestation; vegetation effects; woodlots.

Manuscript received 5 August 1998; revised 15 May 1999; accepted 15 July 1999.

[1] Present address: U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Western Ecology Division, Corvallis, Oregon 97333 USA. E-mail: jcompton@mail.cor.epa.gov

[2] Present address: Institute of Arctic Biology, University of Alaska, Fairbanks, Alaska 99775 USA.

Introduction

During the last 150 yr, most of the landscape of eastern North America has been transformed from predominantly agricultural lands to forest (Williams 1990). While 45–65% of New England's landscape was cleared for pasture or cropland by the mid-1800s, much of this area was abandoned after 1850, and today 70–90% of these lands support mature forests (Hendricksen 1933, Foster et al. 1998). Abandonment of agricultural lands during the last century has also occurred in the southeastern United States (Delcourt and Harris 1980, Kalisz 1986), and Puerto Rico (García-Montiel and Scatena 1994), and is predicted to occur in portions of New Zealand and Europe over the next few decades (Houghton 1996, Maclaren 1996).

Changes in soil organic matter and nutrient pools following the conversion of native systems to agriculture are well documented. Cultivation of temperate forest soils reduces soil C by an average of 30% (see reviews by Johnson 1992, Davidson and Ackerman 1993), through accelerated decomposition in cultivated horizons, reduced plant inputs, and erosion of surface horizons. Active soil C and N fractions are even more sensitive to the effects of conversion and continuous cultivation than total soil C and N pools. Upon tillage, relative losses of 50–75% for soil microbial biomass and light fraction (or relatively undecomposed particulate) organic matter are common (Cambardella and Elliot 1994, Tiessen and Stewart 1983, Collins et al. 1992). Potentially mineralizable N is often reduced by tillage (Campbell and Souster 1982), while net nitrification is typically elevated in agricultural soils (Schimel 1986).

In the northeastern United States, trees quickly invade abandoned farmland, yet recovery of soil organic matter and nutrient dynamics may not proceed rapidly. There often is a time lag between plant production and soil C storage; for example, nearly all the increase in ecosystem C went into standing biomass, not soil organic matter, during 30 yr of old-field succession in North Carolina (Richter et al. 1995). Total soil organic matter and N content increased in an old-field chronosequence spanning 75 yr in New Hampshire, but a minimum of 200 yr was predicted to be necessary to obtain soil organic matter found in nearby primary forests (Hamburg 1984). Although the estimate of Houghton et al. (1983) that temperate soil C recovers to 90% of predisturbance levels within 50 yr was supported in the southeastern United States (Schiffman and Johnson 1989), soil organic matter had not recovered to native levels 50 yr after abandonment in Colorado semi-arid grasslands (Burke et al. 1995). This 50-yr time frame may also be too short for northern temperate forests.

Many studies examining the recovery of nitrogen dynamics after abandonment have focused on the short-term recovery from 0–60 yr after abandonment (Haines 1977, Christensen and MacAller 1985, Kalisz 1986, Zak et al. 1990, Richter et al. 1994, Ihori et al. 1995) or have compared old fields with late successional forests which have very different plant communities and site history (Lamb 1980, Robertson and Vitousek 1981, Pastor et al. 1987). Examining changes on sites >60 yr after their abandonment is difficult in part because few aerial photos are widely available prior to the 1930s. Several recent studies, however, have used older maps or archaeological information to move back farther in time and have revealed older land-use legacies. Koerner et al. (1997), for example, compared 100-yr-old forests classified as either forest, pasture, cropland, or garden during the early 1800s, and found that soils in all former agricultural lands contained more P and lower C:N ratios than continuously forested areas. Other evidence of prior agriculture, based on soil chemistry, has been reported for Scottish highland sites last farmed in the 1700s (Entwistle et al. 1998) and for Andean grassland sites farmed 1500 yr ago (Sandor and Eash 1995).

Collectively, these findings suggest that land-use history could have an important long-term (>100 yr) legacy on nutrient pools throughout the reforested landscape of many temperate regions. Recent papers have suggested that land-use history is an important factor influencing the capacity of forested watersheds to retain increased atmospheric N inputs (Magill et al. 1997, Aber et al. 1998, Fenn et al. 1998). While there is some evidence for this (Silsbee and Larson 1982, Feger 1992, Magill et al. 1997), few replicated studies have examined the long-term (>100 yr) impacts of common land-use practices (e.g., logging, cultivation, and pasturing) on present-day nutrient transformations. Ignoring the potential importance of site history may be a significant oversight in studies of forest biogeochemistry.

Our study examines the residual impact of farming on soil C, N, and P pools; light fraction organic matter; microbial populations; and N transformations in an area of New England used for agriculture in the mid-1800s but which has been forested for >90 yr. Previous related work in a low-fertility sand plain in central Massachusetts suggested that soil C content and N transformations were influenced by prior cultivation (Motzkin et al. 1996, Compton et al. 1998). Those findings led us to design a broader study in a more fertile glacial till-derived soil, representative of a large proportion of southern New England. This study is unique in comparing three different land uses (woodlot, pasturing, and cultivation), and the recovery of two different vegetation types (hardwood vs. conifer). We also measured nitrifier populations and activity; the C, N, and $\delta^{15}N$ of the light fraction and heavy fraction organic matter pools; microbial N (chloroform fumigation-extraction); and field net N transformations during the 1994–1995 growing season. Our major questions were: (1) Are soil C, N, and P lower in areas previously used for agriculture? (2) Are N transformations and nitrifiers influenced by land-use history? (3) Do nutrient ratios in recently deposited organic matter (forest floor and light fraction) vary by prior land use? (4) Does the type of agricultural use or the composition of the new forest influence these pools and transformations?

Methods

Study site

Our study was conducted in the Prospect Hill tract of Harvard Forest (42°30′ N, 72°10′ W), in the central Massachusetts town of Petersham. Elevation ranges from 270 to 420 m above sea level. Soils are of the Canton and Scituate series (Typic Dystrochrepts), which are deep, well-drained loam soils derived from glacial till, and bedrock of mica-rich schist, granodiorite, and gneiss. Mean weekly air temperature varied from 20°C in July to −6°C in January, and precipitation averaged 126 cm (1990–1994 data).[3] A pronounced drought occurred throughout New England in early to mid-1995: January–August 1995 rainfall was ~40% less than the previous 5-yr average rainfall during the same months (Goulden et al. 1996). The vegetation is representative of the transition between the northern hardwoods region and the southern oak–hickory forests. In recently disturbed sites, red oak (*Quercus rubra*), white pine (*Pinus strobus*), and red maple (*Acer rubrum*) dominate, and older forests are dominated by eastern hemlock (*Tsuga canadensis*), white pine, beech (*Fagus grandifolia*), and yellow birch (*Betula alleghaniensis*).

History of the Prospect Hill tract

Historical information was assembled and summarized by Raup and Carlson (1941), Raup (1966), and Foster (1992). Petersham was settled by Europeans in 1733, and little information is available regarding pre-European land use. Forest clearance in Petersham proceeded at ~1–4% per year until the late 1700s, then accelerated to meet increased demands for cattle and sheep pasture. A one-hundred-acre farm might have contained 4–6 acres of crops, 8–10 acres of upland mowing, a similar amount in meadow, and the rest in pasture and woodland. In the early to mid-1700s, crops included vegetables and rotations of cereals and grasses with 7–15 yr of fallow after a decade of use. After the introduction of organic fertilizers, the fallow period was reduced to 1–2 yr. Cleared land increased from 50% of the area in 1800 to nearly 85% during the peak of agriculture in 1840. At that time, ~15% of the landscape was cultivated, while 65% of the landscape was occupied by grasses for pasture and mowing. The remaining forests occupied rocky slopes or swamps where timber was removed and grazing animals roamed during much of the summer. Beginning in 1850, residents left for jobs in developing urban centers or for agricultural opportunities in the Midwestern United States, and large areas of farmland were abandoned and allowed to revert to forest. In 1907 Harvard University purchased the Prospect Hill tract for use as a research forest.

Site selection and experimental design

We selected 12 plots from a subset of the study plots used for a broader vegetation survey (Motzkin et al. 1999). Three major historical land uses were identified: cultivation (frequent removal of plant biomass, removal of stumps and rocks, mixing surface organic matter and mineral soil to ≤20 cm, possible addition of animal manure), pasturing (removal of vegetation and forest floor, seeding of pasture grasses, removal of stumps and rocks, no mineral soil disturbance >5 cm), and woodlots (frequent removal of trees, no mineral soil disturbance, little forest floor disturbance). Ideally, a forest undisturbed by logging would be used for comparison of nutrient levels and transformations; however, at present estimate, all but one of the <25 old-growth stands in Massachusetts are found on steep slopes in western Massachusetts (Dunwiddie and Leverett 1996). Therefore it was not possible to find old-growth forest sites on similar soils for comparison in this study, and we use the woodlot sites as our ''least disturbed'' metric.

Former land use was identified based on field and historical evidence (Raup and Carlson 1941, Spurr 1950, Foster 1992; Motzkin et al. 1999), and more recent field examination. Field indicators of cultivation include the presence of an Ap horizon >10 cm deep, absence of surface stones, smooth microtopography, and bordering stone walls composed of small cobbles. Formerly cultivated sites had a 16–20 cm thick Ap horizon (plow layer) with moist soil color generally one Munsell hue darker than the B horizon below it and an abrupt lower boundary. Pastured soils refer to ''unimproved pasture'' of Motzkin et al. (1999), with no evidence of soil mixing to >5 cm. Woodlots were differentiated from pastures based on historical records, microtopography, and presence of old stumps or tip-up mounds, since stumps were often removed in the conversion to pasture.

Farmers tended to avoid the poorly drained areas of Prospect Hill (Raup and Carlson 1941, Foster 1992), thus land-use history and inherent site factors may be confounded. In order to minimize inherent site differences among land uses, we examined areas with slopes <10% and well-drained soils of the Canton and Scituate series. After field examination, we established four 20 m diameter plots each within former woodlots, pastures and cultivated areas (Table 1). Cultivation and logging ended on all plots ~90–120 yr prior to 1995. Since plant communities could influence soil processes, the plots were stratified into two broad vegetation classes: conifer and hardwood.

We acknowledge that increased replication is desirable in this type of study dealing with complex site

[3] URL: ⟨www.lternet.edu/hfr⟩.

TABLE 1. Soils, land-use history, and vegetation information for all plots.

Plot vegetation class	Prior land use	Plot no.	Soil series	Year last used	Basal area (m^2/ha)	Species (count > 1)
Conifer	Woodlot	76	Canton	1890	44	*Tsuga canadensis, Pinus strobus*
	Woodlot	132	Canton	1890	44	*T. canadensis*
	Pastured	218	Canton	1880	57	*Pinus resinosa, P. strobus*
	Pastured	227	Scituate	1880	60	*P. resinosa*
	Cultivated	155	Scituate	1908	44	*P. strobus, Acer rubrum*
	Cultivated	215	Canton	1908	46	*T. canadensis, Picea rubens, P. strobus, P. resinosa*
Hardwood	Woodlot	135	Scituate	1890	37	*A. rubrum, Quercus rubra, T. canadensis*
	Woodlot	46	Canton	1890	37	*Q. rubra, A. rubrum, Fagus grandifolia*
	Pastured	43	Canton	1908	32	*A. rubrum, Betula alleghaniensis*
	Pastured	91	Canton	1908	30	*A. rubrum, Q. rubra*
	Cultivated	235	Canton	1908	28	*Q. rubra, Acer saccharum, P. strobus*
	Cultivated	134	Canton	1908	46	*Fraxinus americana, Prunus serotina, A. rubrum, Acer pensylvanicum*

Note: ''Last used'' refers to when the area was abandoned from agriculture or last extensively logged.

histories or ''treatments.'' Sites were selected from a larger set of ~200 plots (Motzkin et al. 1999), and no bias was used in site selection other than holding soils and topography constant. Strong consideration was given to the possibility that inherent site differences were responsible for the patterns observed.

Mineral soil nitrogen transformations

Field net nitrogen mineralization was measured in late summer 1994 and from May–October 1995 using the in situ buried bag method (Eno 1960) as modified by Boone (1992) to use intact soil cores. We only measured mineralization in the mineral soil, because the strongest impact of agriculture was expected to be observed in the former plow layer. The forest floor was removed, and a pair of soil cores collected from the mineral soil 0–15 cm depth using a cylindrical metal corer. Five pairs of soil cores were collected from random locations within each plot in August 1994, and three pairs of cores were collected during the 1995 sampling periods. Time-zero cores were stored on ice and returned to the lab for processing. The second core was kept intact within a perforated plastic tube, which was then capped and placed in a gas-permeable polyethylene bag (0.025 mm thickness) within a nylon mesh bag to prevent disturbance of the core by soil fauna. The core was placed back in the original hole, covered with forest floor and incubated in the field for six weeks per measurement period.

Soil cores were kept cool (<5°C) until returned to the lab, sieved to <2 mm, and extracted within 24 hrs of collection. Sieved fresh soil (10 g) was shaken for 1 min with 100 mL 2 mol/L KCl, allowed to stand for 24 hrs, then suction filtered through Whatman GF A/E filters (Whatman, Clifton, New Jersey, USA). Soil KCl extracts were frozen until colorimetric analysis for ammonium and nitrite plus nitrate by flow injection ion analyzer (LACHAT Instruments, Milwaukee, Wisconsin, USA). Net nitrification was calculated as the net change in nitrate between the time-zero and six-wk cores. Net N mineralization was calculated as the change in ammonium plus nitrate. Moisture content (105°C for 24 hr) and loss-on-ignition (550°C for 4 hr) were determined, and oven-dried mass of the sieved soil in each core was used to determine bulk density of the <2 mm soil.

Forest floor and mineral soil carbon, nitrogen, and phosphorus

Forest floor (Oi, Oe, and Oa) was collected from a 15 × 15 cm area in early June 1995 from five random locations within each site, and the mass (<5.6 mm) corrected for moisture and ash content. Mineral soil was collected from the 0–15 cm soil depth at five random locations within each site in August 1995. Soil and forest floor materials were finely ground using a roller mill followed by mortar and pestle. Total C and N in the forest floor (June 1995) and 0–15 cm soil (August 1994) were determined by carbon–nitrogen analyzer (Fisons Instruments, Beverly, Massachusetts, USA) using 30 mg soil and 7 mg forest floor. Acidification of a subset of samples with 4 mol/L HCl indicated that no carbonates were present. Total P was determined by the modified Kjeldahl digest of Parkinson and Allen (1975) using 0.3 g of mineral soil and 0.2 g of forest floor. Phosphate in the digests was determined by ion analyzer using the molybdophosphate ascorbic acid technique (LACHAT Instruments, Milwaukee, Wisconsin, USA).

Light fraction carbon and nitrogen

Light fraction mass was determined on <2-mm sieved, air-dried mineral soil collected at time zero for the May 1995 sampling using a modification of Strickland and Sollins (1987). Ten grams of air-dried soil was placed in a centrifuge tube with 20 mL sodium metatungstate solution (density 1.75 g/cm^3). The tubes were shaken by hand for 30 s, then centrifuged at 1000 rpm for 15 min. The floating light fraction was siphoned off with a syringe fitted with 2 cm of Tygon tubing. The centrifugation and siphon process was repeated ≥four times until no floating material remained. The light and heavy fraction were washed over Whatman GF A/E filter paper with ≥100 mL deionized water and dried for two hr at 65°C. Both fractions were finely ground and analyzed for C and N. Percent light fraction of the total soil was multiplied by the <2 mm soil mass per hectare to obtain light fraction C and N contents. Light fraction and heavy fraction $\delta^{15}N$ was determined for one equal-weight composite sample per site using a Europa Hydra 20/20 continuous flow isotope ratio mass spectrometer (PDZ Europa, Cheshire, UK) dedicated for natural abundance samples at the University of California–Davis.

Microbial assays

Chloroform-extractable N was determined on time-zero soils collected for in situ N mineralization in June, July, and September 1995 by the fumigation-extraction method (Brookes et al. 1985) within two days of collection. Total N in the fumigated and non-fumigated 0.5 mol/L K_2SO_4 extracts was measured as nitrate following alkaline persulfate digestion (Cabrera and Beare 1993). No correction factor (K_{EN}) was used; therefore the data are presented as chloroform-extractable N to provide an index of microbial N.

Potential nitrification was determined for the July 1995 mineral soil samples using an aerobic shaken slurry method (Schmidt and Belser 1994). This short-term assay provides an indication of the activity and size of autotrophic nitrifier populations. Soil (20 g moist) was shaken for 24 hr with 90 mL phosphate buffer plus 0.2 mL 0.25 mol/L ammonium sulfate; aliquots of the solution collected at 2, 4, 18, and 24 hours were filtered through Whatman GF/A filters and frozen until analysis of nitrite plus nitrate (<48 hr). Chlorate was not added since it did not appear to stop conversion of nitrite to nitrate. Potential nitrification was calculated by determining the linear rate of nitrate increase for the 2–24 hr time period.

Counts of culturable nitrifying bacteria were conducted using a most probable numbers technique (Schmidt and Belser 1994) for one of the two replicate plots per land use by vegetation combination, chosen at random. Nitrite oxidizers (*Nitrobacter*) were enumerated in fresh soils collected 31 August 1995 (during drought) and 12 September 1995 (after rainfall), using five replicate ten-fold dilution series initiated for each sample within 24 hr of collection. We used both the recommended media nitrite concentrations and one-tenth nitrite concentrations, since forest soil nitrifiers are inhibited by high substrate N concentrations (Donaldson and Henderson 1989). Culture tubes were checked for the presence of nitrite or nitrate every week for several months until no further changes were observed.

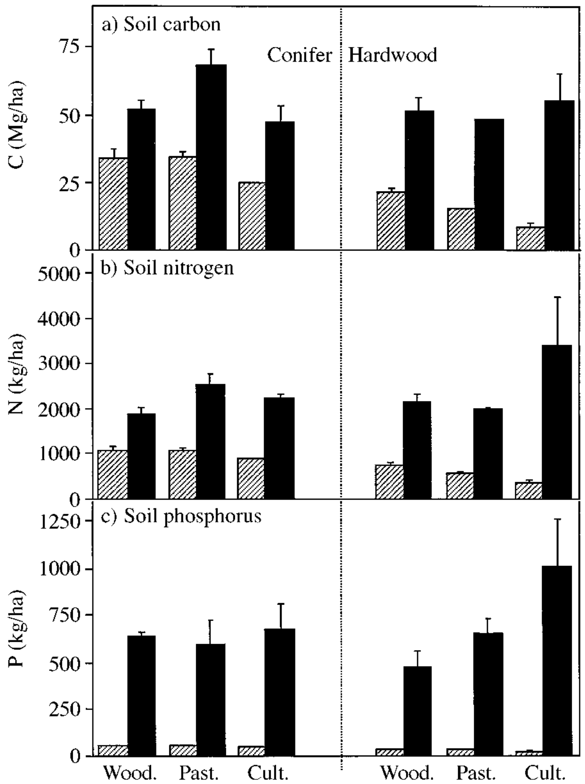


FIG. 1. Total C, N, and P in forest floor and mineral soil by prior land use (Wood. = woodlot; Past. = pastured; Cult. = cultivated) and present vegetation. Hatched bars depict data for the forest floor, and solid bars depict data for mineral soil; error bars indicate +1 SE (n = 2 sites).

Statistical analyses

The data were analyzed by two-way factorial analysis of variance using present vegetation (conifer or hardwood) and land-use history (woodlot, pastured, or cultivated) as main effects and site as a covariate. All ANOVAs were conducted using the general linear model in SYSTAT (Wilkinson 1992). Analyses for N mineralization and nitrification were conducted within each time period since it was expected that the rates would vary across time. Nitrification data were log-transformed because of non-normal distribution. Potential nitrification was ranked to avoid both non-normal distribution and nonhomogeneous variances, and the ANOVA performed using ranked data in SYSTAT (nonparametric ANOVA). Post hoc pairwise multiple comparisons were conducted using Tukey's honestly significant difference procedure.

RESULTS

Forest floor and mineral soil carbon, nitrogen, and phosphorus

Forest floor carbon was lower in previously cultivated sites than woodlots (Fig. 1, Tables 2 and 3), while mineral soil C did not vary significantly by prior land

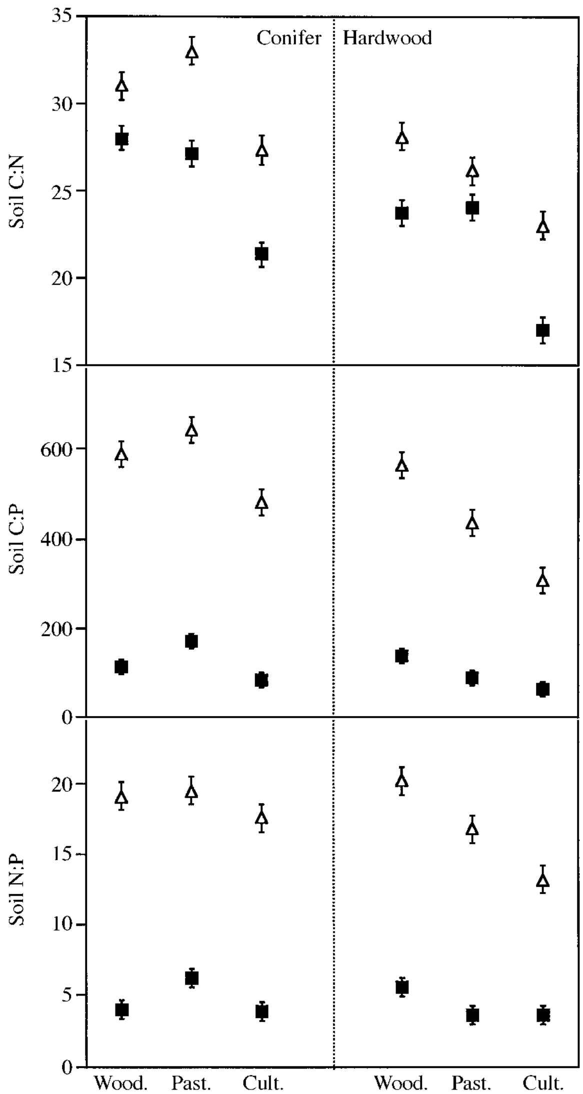


FIG. 2. Ratios of total C, N, and P in forest floor (open triangles) and 0–15 cm soil (solid squares) by prior land use (abbreviations as in Fig. 1) and present vegetation. Error bars indicate ± 1 SE ($n = 2$).

use ($P = 0.072$). Forest floor plus mineral soil C to 15 cm depth was influenced by land-use history, and was 13–16% greater in the woodlots than cultivated sites. Forest floor mass in cultivated sites averaged 20 Mg/ha under hardwoods and 59 Mg C/ha under conifers, while woodlots had 29 and 10 Mg C/ha greater forest floor mass under hardwood and conifer, respectively. Hardwood sites had less soil C to 15 cm than conifer sites, because they contained less forest floor mass. Loss-on-ignition was significantly lower in the hardwood forest floor (Table 2).

Mineral soil N and P contents were influenced by land-use history (Table 3), and were slightly greater in formerly pastured and cultivated sites relative to woodlots (Fig. 1). The cultivated hardwood sites had the highest soil N and P concentrations, and highest P concentrations in the forest floor. Forest floor N or P concentrations did not vary consistently between conifer and hardwoods (Table 2), but the conifer sites had greater forest floor mass, leading to higher forest floor N and P contents. Because cultivated sites had slightly less forest floor mass, forest floor N was lower than in pastures or woodlots. Forest floor P content did not vary by land use, and a much smaller proportion of soil P was contained in this pool as compared to the forest floor C or N. The pastured conifer stands, dominated by red pine (90% of basal area), had the highest overall forest floor mass and mineral soil C and N content. There was a strong interaction between land-use history and present vegetation for soil N and P contents (Table 3): the hardwood sites varied more strongly by land use than did the conifer sites.

Present vegetation and prior land use influenced C:N, C:P, and N:P ratios, especially in the forest floor (Fig. 2, Table 3). Mineral soil and forest floor C:N ratios were consistently 5 units lower in cultivated sites than pastured or woodlot sites, regardless of vegetation. Forest floor C:P and N:P ratios were lower in cultivated soils, especially under hardwoods. Mineral soil C:P and N:P ratios varied strongly by land-use history only under hardwoods, since cultivated hardwood sites had higher soil P. Pastured site C:N ratios were more similar to woodlots than cultivated sites. Conifer pastures (dominated by red pine) had the highest forest floor C:N ratios, they also had had higher forest floor and soil C:P ratios and higher soil N:P ratios. In contrast, hardwood pasture forest floor and soil C:P and N:P ratios were intermediate or more similar to cultivated sites.

Soil organic matter density fractions: carbon, nitrogen, and natural abundance $\delta^{15}N$

Although mineral soil carbon did not differ by prior land use, light fraction carbon was 5–11 Mg C/ha lower (36–61% less) in cultivated soils than woodlot soils (Fig. 3). Prior land use influenced light fraction mass and C content ($P = 0.019$ and $P = 0.001$, respectively), but did not influence light fraction N content. Light fraction organic matter comprised 4–10% of soil mass, 21–39% of soil C, and 16–36% of soil N. Heavy fraction C was not influenced by prior land use, while heavy fraction N was slightly greater in the cultivated sites ($P = 0.084$). Carbon to nitrogen ratios were generally lower in the heavy fraction than the light fraction. Cultivated sites had lower C:N ratios in both the light fraction and heavy fraction (Fig. 3). The heavy fraction C:N ratio was lower for hardwoods than for conifers.

Strong trends were observed for natural abundance $\delta^{15}N$ in the light and heavy fraction organic matter pools (Fig. 4). The heavy fraction was enriched by 1–3% compared to the light fraction, with little overlap of values. No statistical comparisons were made due to low sample sizes (one composite sample per site).

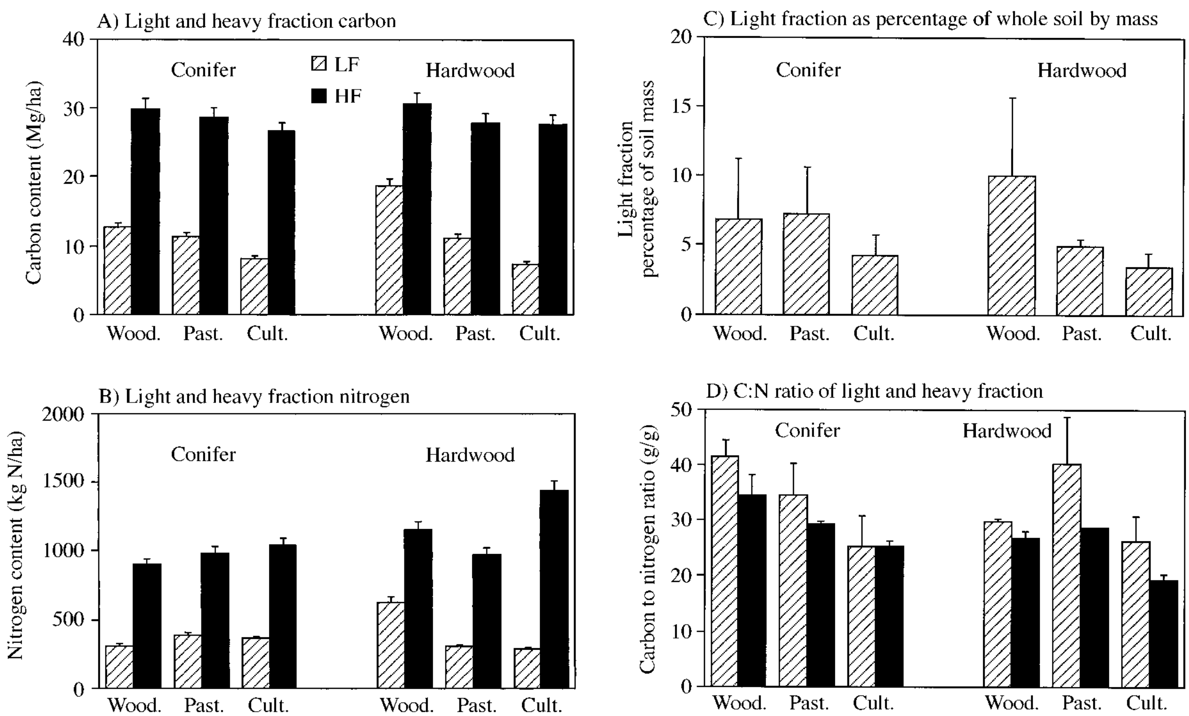


FIG. 3. Light-fraction (LF) and heavy-fraction (HF) mass, C:N ratio, and C and N content in mineral soil collected in May 1995 (0–15 cm depth). Error bars indicate + 1 SD. Prior land use abbreviations are as in Fig. 1.

The variability between replicate plots was generally low (CV = 10%), but the conifer sites showed greater variability than hardwoods. Pastured and cultivated heavy fraction pools were depleted in ^{15}N compared to the woodlot soils. In contrast, the light fraction $\delta^{15}N$ was higher in the pastured and cultivated sites. For both vegetation types, the light fraction and heavy fraction of the cultivated sites were most similar, while the greatest divergence between light fraction and heavy fraction $\delta^{15}N$ values occurred in the woodlots.

Nitrogen mineralization and nitrification

Net N mineralization was quite variable within plots and over time, and was greater under hardwoods than

TABLE 2. Soil properties for the forest floor (<5.6 mm) and mineral soil (0–15 cm depth).

Site	Mass (Mg/ha)	BD (g/cm^3)	LOI (g/g)	pH [2:1, water]	C (g/kg)	N (g/kg)	P (mg/kg)
Forest floor							
Conifer							
Woodlot	69^a	0.13^a	0.92	3.36^c	487^a	15.9	842^{bc}
Pastured	74^a	0.12^{ab}	0.86	3.48^{bc}	460^{ab}	14.0	720^c
Cultivated	59^{ab}	0.10^{abc}	0.83	3.71^b	418^{ab}	15.3	880^{bc}
Hardwood							
Woodlot	49^b	0.13^a	0.79	3.61^{abc}	428^{ab}	15.3	768^c
Pastured	36^{bc}	0.09^{bc}	0.78	3.70^{abc}	419^{ab}	16.1	958^b
Cultivated	20^c	0.08^c	0.77	4.27^a	393^b	17.0	1 321^a
Mineral soil							
Conifer							
Woodlot		0.86	0.11	4.29^{ab}	58.2^b	2.10^b	636^{ab}
Pastured		0.80	0.09	4.08^b	83.6^a	3.09^{ab}	598^b
Cultivated		0.84	0.11	4.41^{ab}	54.6^b	2.53^b	677^{ab}
Hardwood							
Woodlot		0.86	0.10	4.29^{ab}	58.6^b	2.46^b	476^b
Pastured		0.86	0.14	4.46^a	51.8^b	2.17^b	652^{ab}
Cultivated		0.86	0.09	4.44^a	58.8^b	3.66^a	1 009^a

Notes: Within each column and material type, values with the same superscript letter are not significantly different ($P > 0.05$). Abbreviations: BD = bulk density; LOI = loss-on-ignition.

TABLE 3. Effects of present vegetation and land-use history on soil properties and processes as indicated by *P* values for a two-way ANOVA, using site as a covariate.

Property or process	Variable	Source of variation: Vegetation	Land use	Interaction
Forest floor + mineral soil (kg/ha)	C	0.000	0.001	0.001
	N	NS	0.010	0.000
	P	NS	0.002	0.010
Mineral soil (0–15 cm; kg/ha)	C	NS	NS	0.000
	N	0.013	0.000	0.000
	P	NS	0.001	0.008
Forest floor (kg/ha)	C	0.000	0.002	NS
	N	0.000	0.007	NS
	P	0.000	NS	NS
Mineral soil (g/g)	C:N	0.000	0.000	NS
	C:P	NS	0.002	0.010
	N:P	NS	NS	0.007
Forest floor (g/g)	C:N	0.000	0.000	NS
	C:P	0.000	0.000	0.007
	N:P	0.017	0.000	0.013
Light fraction (kg/ha)	C	NS	0.001	NS
	N	NS	NS	0.024
	C:N	NS	0.007	0.029
Heavy fraction (kg/ha)	C	NS	NS	NS
	N	0.041	NS	NS
	C:N	0.000	0.000	0.022
N mineralization ($kg \cdot ha^{-1} \cdot 28\ d^{-1}$)				
Aug 1994		NS	0.017	0.000
May 1995		0.007	NS	NS
June 1995		0.003	NS	NS
July 1995		NS	NS	NS
Sept. 1995		0.031	NS	NS
Nitrification ($kg \cdot ha^{-1} \cdot 28\ d^{-1}$)				
Aug 1994		NS	0.009	0.011
May 1995		NS	0.010	0.001
June 1995		0.001	0.002	0.016
July 1995		0.018	0.001	0.042
Sept. 1995		0.017	0.000	0.009
$CHCl_3$-extractable N (mg/kg)				
June 1995		0.019	NS	NS
July 1995		0.039	NS	NS
Sept. 1995		NS	NS	0.005
Potential nitrification (mg/kg)				
Per unit soil		0.006	0.020	NS
Per unit organic matter		0.010	0.015	NS
1995 net mineralization (kg N/ha)		0.002	NS	NS
1995 net nitrification (kg N/ha)		0.010	0.010	NS

Note: NS, not significant ($P > 0.05$).

conifers (Fig. 5, Table 3). During two time periods (May and June 1995) the hardwood cultivated sites had much higher net N mineralization rates than all other sites. In August 1994, net N mineralization was much higher in conifer pastured and woodlot sites than other sites. Present vegetation was a more important factor influencing net N mineralization than land-use history (Table 3). Vegetation was a significant factor in May, June, and September 1995, while land-use history was significant only in August 1994, when there was also a significant interaction between land-use history and present vegetation. Comparing growing season N mineralization rates (Table 4), present vegetation appears to have a greater influence, with rates being more than two times higher in hardwoods than conifers.

Nitrification varied by land-use history and vegetation (Fig. 6; Table 3). Land-use history was a significant factor during all five time periods, while vegetation was a significant influence during three time periods. Net nitrification was 2–24% of total N mineralization (Table 4), and was less variable over time than N mineralization. Although nitrification was detected for at least one time period in all sites, substantial nitrification occurred only in the hardwood cultivated sites. There was a significant interaction between land-use history and vegetation during all time periods, largely because

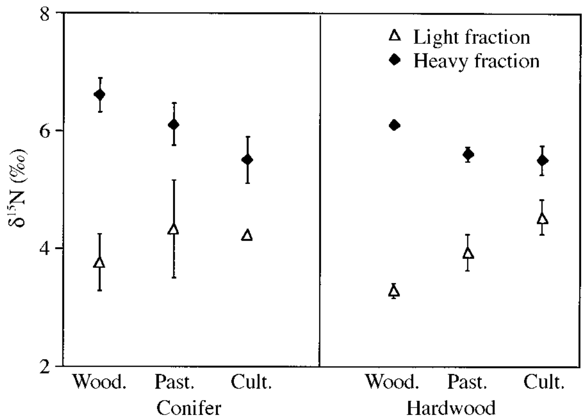


FIG. 4. Natural abundance of $\delta^{15}N$ in light and heavy fraction soil. Error bars represent ±1 SD (n = 2 sites). Prior land-use abbreviations are as in Fig. 1.

cultivation had a more pronounced effect on net nitrification under hardwoods than conifers.

Net N mineralization was lowest in July 1995, the driest sampling period; gravimetric soil moistures (0–15 cm depth) dropped from 25% at the previous sampling date to 15%. Net nitrification did not decrease as markedly as N mineralization during the drought, implying that nitrification is not as sensitive to moisture changes or could occur within moist microsites in drying soils. Net nitrification was highest in June.

Chloroform-nitrogen and nitrifying bacteria

Although cultivated hardwood sites consistently had the highest chloroform fumigation-extraction N, land-use history was not a significant factor during any time period (Fig. 7, Table 3). As observed for N mineralization, present vegetation appeared to have more influence on chloroform-N than did land-use history. Chloroform-N was similar across plots and less temporally variable than N mineralization. The interaction between vegetation and land-use history was a significant term in June and September, since formerly cultivated hardwood sites had much greater chloroform-N.

Vegetation and prior land use influenced potential nitrification. While only the cultivated hardwood sites had appreciable net nitrification (Fig. 6: July 1995), both former pastures and cultivated sites had high potential nitrification relative to the woodlots (Fig. 8a). Hardwood sites had higher potential nitrification than conifer sites. Potential nitrification and in situ net nitrification in July 1995 were not well correlated (R^2 = 0.025).

Culturable autotrophic nitrifying bacteria were more abundant in pastured and cultivated sites than in woodlot sites (Fig. 8b). These data were quite variable, but autotrophic nitrifiers were observed in all stands. The more dilute media (10% the concentrations of Schmidt and Belser 1994) generally yielded higher nitrifier counts, as observed by Donaldson and Henderson (1989). Counts were much higher in early September, when the drought ended, than in late August.

DISCUSSION

Persistent effects of land-use history on total soil carbon, nitrogen, and phosphorus

The cultivation of forest soils reduces soil carbon by an average of 30% (Johnson 1992, Davidson and Ackerman 1993). Soil carbon (forest floor + mineral soil 0–15 cm depth) for both vegetation types was 13–16% lower in sites last cultivated 90–120 yr prior to sampling than in permanent woodlots, mainly because of lower forest floor C content in the cultivated sites. Although we expected the Ap horizon to more strongly reflect prior land use, mineral soil C content (0–15 cm depth) did not vary by prior land use. There are several possible explanations for this result: (1) soil C was not reduced by 19th century agricultural practices, (2) soil C has recovered within a century, (3) logging also reduces soil C, and (4) the 0–15 cm depth mineral soil samples do not accurately reflect the full mineral soil profile. The first two possibilities may not be the case in glacial till soils of New England: Hamburg (1984) indicates that soil organic matter was lower in sites abandoned <70 yr prior to sampling than in an uncultivated stand, and forest floor mass was still accu-

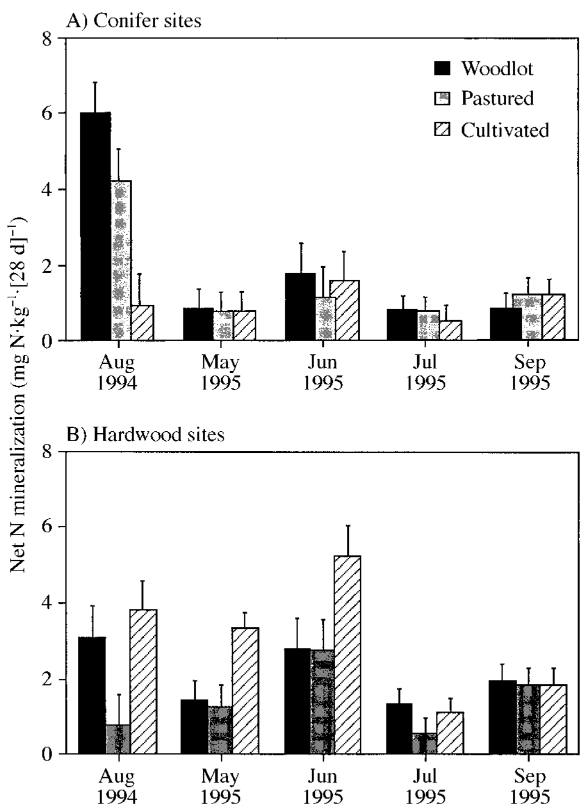


FIG. 5. In situ net nitrogen mineralization rates per gram soil in the 0–15 cm mineral soil. Error bars indicate +1 SE (n = 2 plots).

TABLE 4. Growing season net nitrogen mineralization (net ammonium plus nitrate accumulation in buried bags), nitrification, and nitrification as a percentage of N mineralization.

Plot type	N mineralization ($kg \cdot ha^{-1} \cdot yr^{-1}$)	Nitrification ($kg\ N \cdot ha^{-1} \cdot yr^{-1}$)	Nitrification as a percentage of N mineralization
Conifer			
Cultivated	7.9[b]	0.8[ab]	9.3
	(3.5)	(0.5)	(2.4)
Pastured	8.2[b]	0.5[b]	6.0
	(0.6)	(0.2)	(2.1)
Woodlot	7.7[b]	0.1[b]	1.6
	(0.1)	(0.0)	(0.1)
Hardwood			
Cultivated	18.1[a]	4.3[a]	23.7
	(0.7)	(0.3)	(2.5)
Pastured	11.8[ab]	0.6[ab]	4.6
	(1.6)	(0.2)	(1.3)
Woodlot	14.7[ab]	0.6[ab]	4.2
	(0.0)	(0.1)	(0.5)

Notes: The time period is 11 May through 31 October 1995. One standard error of the mean of two plots is shown in parentheses. Within the same column, values with the same superscript letter are not significantly different ($P > 0.05$).

mulating at a linear rate. Logging can reduce soil C to some extent, although the reduction is less dramatic than for agriculture (Johnson 1992). Downward translocation of organic matter (Motzkin et al. 1996) and absence of woody root inputs (Richter et al. 1990) during the agricultural period are important factors that may influence C content of soil below the Ap horizon. While carbon in the 0–15 cm mineral soil depth did not vary by land-use history, deeper mineral soil must be considered in any assessment of land-use effects on soil C.

The forest floor was influenced by land-use history and accumulated more rapidly under conifers than hardwoods: accumulation in the cultivated sites after abandonment was ~0.23 $Mg \cdot ha^{-1} \cdot yr^{-1}$ under hardwoods and 0.68 $Mg \cdot ha^{-1} \cdot yr^{-1}$ under conifers. Forest floor mass was not strongly related to basal area ($R^2 = 0.34$). Accumulation of the forest floor is regulated by the balance between litter inputs and outputs, including decomposition and organic matter transfer to the mineral soil through mixing and to a lesser extent leaching. The lower forest floor masses in the cultivated soils, as compared to woodlot soils, may result from increased turnover or less litter production in the cultivated sites.

Our findings suggest that cultivation increased soil N and P levels, persisting long after the agricultural period ended. In contrast, modern-day temperate-zone agriculture, despite addition of inorganic N and P fertilizers, generally reduces soil N and P (Tiessen et al. 1982, Post and Mann 1990). Organic amendments may yield a different result. In the Hoosfield continuous barley experiment at Rothamsted, an agricultural plot manured from only 1852–1871 had 26% more N and 56% more C to 23 cm soil depth in 1975 than an adjacent unamended plot (Jenkinson and Johnston 1976). In contrast, soil C and N were lower where inorganic NPK fertilizers were added for over a century. In our study, mineral soil in cultivated sites contained 2800 kg N/ha and 843 kg P/ha, compared with 2010 kg N/ha and 556 kg P/ha in woodlots, a relative increase of 39% for N and 52% for P.

There is evidence that animal manures were added to cultivated soils at Prospect Hill. In the mid-1800s ~65% of the landscape was used for pasturing of cattle and sheep in Petersham (Raup and Carlson 1941), and spring plowing of manure into the soil was practiced at Prospect Hill during the 1800s (Raup and Carlson 1941; F. M. Wheeler, *unpublished manuscript*) (see *Discussion: Long-term effects on organic matter density fractions* [last paragraph]). Animal densities in 1831 were 0.66 animal units/ha of pasture and meadow (includes horses, oxen, steers, cows, heifers, sheep, and swine; Petersham tax records in Harvard Forest Archives). Hamburg (1984) did not observe higher N in cultivated soils in central New Hampshire, where domestic animal densities were lower (peak of 0.3 animal units/ha in 1845; 0.15 units/ha from 1825–1925). The addition of ~800 kg N/ha and 300 kg P/ha to the Ap horizon during the >100-yr agricultural period is quite possible, considering that ~65% of the landscape was used for pasture. Animal manure derived from some fraction of the pastures would have been added to cultivated lands, which comprised ≤10–15% of the area. Manures also have low N:P ratios, and the cultivated forest floor and the hardwood cultivated soil reflected this. Rough calculations for the Sanderson farm indicate that of the ~100 kg N in manures produced from two oxen and one cow in 1771, ~10 kg N/ha might have made its way to the 0.25 ha cultivated area (Raup and Carlson 1941; Harvard Forest Research File 1974–04); continuation of this practice for 80 yr could roughly explain the accumulation of N in the cultivated sites.

Soil nutrient levels appear to have been enhanced by 19th century farming practices, reflected as increased N and P levels and lower C:N and C:P ratios. Altered nutrient ratios may be the result of manure additions, or may reflect the influence of an altered decomposition environment. The ratios were largely influenced by higher N and P levels, rather than lower C; hence, addition of animal manure as a farming practice may be an important factor in the postagricultural recovery of soil nutrient dynamics. While inorganic fertilizers may result in net N losses, organic amendments may accumulate in the soil (Drinkwater et al. 1998). In the Andes, agricultural practices 1500 yr ago included adding large amounts of guano to terraced Mollisols, and these sites still have higher C, N, and P than native soils (Sandor and Eash 1995). Our findings also suggest that the effects of animal manure amendments can per-

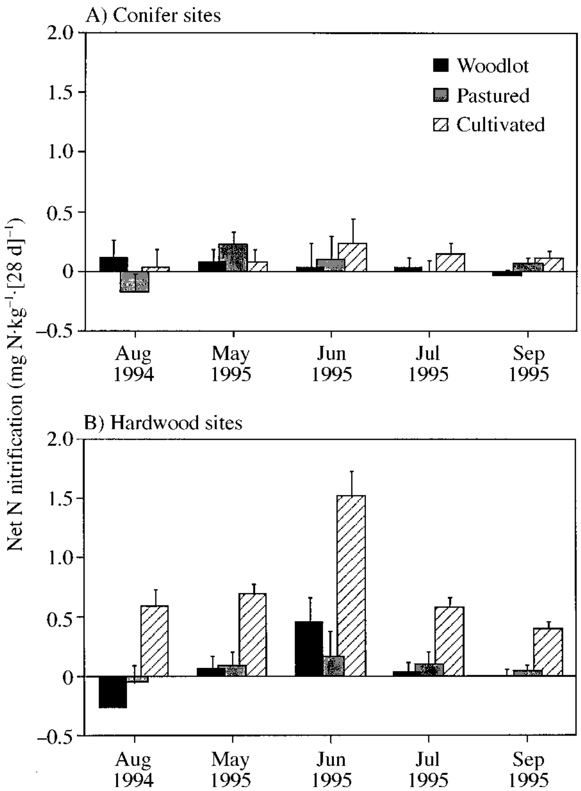


FIG. 6. In situ net nitrification rates per gram soil in the 0–15 cm mineral soil. Error bars indicate +1 SE ($n = 2$ plots).

sist for at least a century in the form of elevated N and P and lower C:N and C:P ratios.

Long-term effects on organic matter density fractions

While bulk mineral soil C did not vary by prior land use, cultivated mineral soil had substantially less light fraction C (36–61% less) than did woodlot soils. The light fraction is largely derived from below- and above-ground litter, and is likely to be of more recent origin, as indicated by lower $\delta^{15}N$ and higher C:N ratios than the mineral-associated heavy fraction. The higher C:N ratios of the light fraction suggest that in the short term it could be a site for N immobilization; however, over the long term, there is a net transfer of organic matter and N from the light fraction to the heavy fraction. Light fraction dynamics might then parallel inputs and decomposition of litterfall, and would exhibit a pattern similar to forest floor mass, which is also lower in the cultivated sites.

The decline in total soil organic matter in agricultural soils has been attributed to losses of the light fraction (Cambardella and Elliott 1994). Agricultural soils tend to have lower light fraction masses, usually <2% of the soil (Janzen 1987, Janzen et al. 1992, Boone 1994). Our light fraction masses of 4–10% of the soil are within the range of those observed in forest soils (Spycher et al. 1983, Sollins et al. 1984, Strickland and Sollins 1987, Boone 1994), although the cultivated soils had much less light fraction mass and C than did the woodlot soils. While the light fraction N in cultivated soils was similar to or lower than that in pastured or woodlot soils, heavy fraction N was greater in cultivated soils, indicating an accumulation of N in this pool.

The natural variation in the ratio of $^{15}N/^{14}N$ in soils can reflect both differences in sources of N and fractionation of N during decomposition (Delwiche and Steyn 1980, Shearer et al. 1978, Nadelhoffer and Fry 1988, 1994). The lighter isotope is preferentially released during decomposition, and losses of inorganic N though leaching, denitrification, and ammonia volatilization result in preferential losses of ^{14}N from soils, resulting in an increase in soil $\delta^{15}N$ over time (Nadelhoffer and Fry 1988, 1994, Handley and Raven 1992). Plants are depleted in ^{15}N relative to soil nitrogen (Nadelhoffer and Fry 1994). The light fraction is mostly sand-sized leaf and root fragments of recent plant origin, and therefore has a lower $\delta^{15}N$ than bulk soil or heavy fraction N.

By more rapidly incorporating isotopically light plant inputs into the mineral soil, cultivation and pasturing can decrease soil $\delta^{15}N$. Soil $\delta^{15}N$ has been found to be lower in cultivated soils than in native grassland (Tiessen et al. 1984) or zero-till agriculture (Selles et al. 1984). Pasturing also decreased the surface soil $\delta^{15}N$ in several tropical grasslands (Piccolo et al. 1994). The heavy fraction (mineral-associated) $\delta^{15}N$ was lower in the cultivated and pastured soils than in woodlot soils. These observed patterns could be driven by similar mechanisms.

Cultivation may increase the mixing of light fraction and heavy fraction, resulting in a convergence of $\delta^{15}N$ values between the two fractions. Tiessen et al. (1984) reported that long-term cultivation of a grassland soil resulted in lower $\delta^{15}N$ in the bulk soil and coarse clay and silt fractions, but higher $\delta^{15}N$ in the sand fraction. These observations were interpreted as a greater accumulation of more labile N compounds depleted in ^{15}N (from byproducts of microbial decomposition) in the mineral-associated fractions under cultivation. Although we measured soil density fractions rather than size fractions, our findings are similar. The $\delta^{15}N$ values for the light fraction and heavy fraction were most similar in cultivated sites, which suggests that turnover of light fraction and incorporation of N into the heavy fraction is more rapid in these soils. Compared to woodlot soils, the heavy fraction N (mineral-associated) of the cultivated soils was depleted in ^{15}N, while the light fraction was enriched in ^{15}N. The closer $\delta^{15}N$ values in the previously cultivated soils suggest a tighter coupling between the two fractions in the cultivated soils than in woodlots.

The $\delta^{15}N$ of soil amendments in cultivated sites is not known, therefore we cannot directly implicate animal wastes as a source of N in the cultivated soils

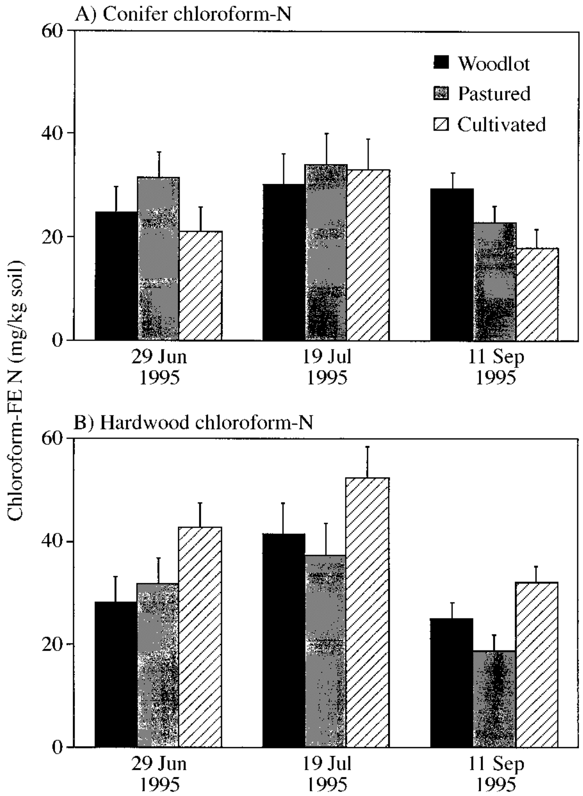


FIG. 7. Chloroform fumigation-extraction (FE) nitrogen as a relative estimate of microbial N by vegetation type and land use. Error bars represent +1 SD (n = 2 plots).

using these values. The $\delta^{15}N$ of animal wastes varies widely depending upon the diet, type of waste, and time of day (Steele and Daniel 1978, Kerley and Jarvis 1996, Kielland and Bryant 1998). Consumption of N_2-fixing plants such as red and white clover (*Trifolium repens* and *Trifolium pratense*), as well as application of urine (Lincoln 1851) which has a lower signature (Steele and Daniel 1978), might have caused the $\delta^{15}N$ values of amendments to be relatively low. The $\delta^{15}N$ values of fresh animal wastes may not be outside the range of soil values (Macko and Ostrom 1994), therefore we cannot use the actual $\delta^{15}N$ values to directly implicate manure N as a source.

Early land-use practices would have increased losses of nitrogen from the soil via harvest, leaching, and erosion, and therefore depleted native soil nitrogen pools. There is widespread discussion of exhaustion of the native soil nutrient capacity after a few years of growing crops on a recently cleared site (see references in Whitney 1994). However, manures were used widely as a soil amendment in the 1800s (Bidwell and Falconer 1941, Russell 1982), including in Petersham (Raup and Carlson 1941; F. M. Wheeler, Diary from 1881–1882, File No. HF 1882–1 in Harvard Forest Archives); thus the re-accumulating active N pool at this time would be largely composed of manure N. The cumulative effect of these practices would be to deplete the native N pool and cause manure N inputs to dominate the actively cycling N pools. In the woodlot soils, little N would have been lost, and the slow process of losing small amounts of ^{14}N over the long term would result in a divergence between the light fraction (recent plant inputs) and the more recalcitrant and older heavy fraction. In the cultivated areas, frequent incorporation of amendments into the soil may have also strongly influenced the $\delta^{15}N$ signal. By increasing the mineralization of humus N and the addition of easily soluble animal wastes, annual cultivation would have made much of the soil N available to plants, resulting in a convergence of the light and heavy fractions. In the woodlot soils, plant uptake and ecosystem losses of N may have preferentially removed ^{14}N from the soil humus for a longer time period, allowing the heavy fraction to become more enriched in ^{15}N over time. We suggest that more than a century after abandonment is required to establish this divergence of $\delta^{15}N$ values for the light and heavy fraction.

Recovery of processes after agricultural abandonment

Nitrification rates and nitrifiers remain elevated 90–120 yr after abandonment of cultivated sites. Nitrification as a percentage of N mineralization was higher in previously cultivated sites (9% and 24% in conifer and hardwoods) than in pastured and woodlot sites (1–6%, Table 4). The 1995 growing season N mineralization rates ranged from 7 to 18 $kg \cdot ha^{-1} \cdot yr^{-1}$ (Table 4). These values are low for this forest type (Magill et al. 1997), presumably because of the summer 1995 drought.

Several mechanisms explain the higher net nitrification found in cultivated sites relative to uncultivated sites 90–120 yr after their abandonment. Cultivation generally increases net nitrification through liming, aeration, enhanced ammonium levels, and lower immobilization (Schimel 1986). Soil pH was slightly higher in formerly cultivated sites (Table 2). Lime was not locally available at Prospect Hill during the 1800s, but burning was the predominant method of land clearing in New England in the 17th and 18th centuries (Bidwell and Falconer 1941), which can increase soil pH and extractable cations (Woodmansee and Wallach 1981). Autotrophic nitrifiers were present at low levels in all stands, and relatively abundant and active in all cultivated and pastured sites. However, net nitrification was substantial only in the cultivated hardwood sites. Potential nitrification and net nitrification rates were not well correlated (R^2 = 0.025), which suggests that factors other than the presence and activity of nitrifiers determines whether a soil will exhibit net nitrification. Immobilization of nitrate is expected to be lower in the cultivated sites, as seen by Schimel (1986) for cultivated grassland soils as compared to native grassland soils. Soil C:N ratios strongly reflect the agricultural legacies, and relatively low ratios (16–18) in the hardwood cultivated sites may shift the nitrate immobilization–mineralization balance, resulting in net nitrate release during decomposition.

Land-use history may influence forest response to increasing N supply. Specific farming practices, in this case the addition of organic amendments during the 1800s, appear to be important in the rate and direction of the long-term postabandonment N transformations, as was suggested by Vitousek et al. (1989). If formerly cultivated soils have higher net nitrate release, then nitrate leaching and ecosystem N retention might be lower in formerly cultivated areas. In a study designed to mimic dramatic increases in atmospheric N inputs, Magill et al. (1997) cite land-use history as a possible explanation for more rapid initiation of N saturation of a formerly cultivated red pine stand than a formerly pastured hardwood stand at Harvard Forest. Consideration of site history may be critical in understanding N retention and response to changing atmospheric loading across the diverse landscape mosaic of the eastern United States.

Vegetation effects—also a land-use legacy?

The interaction between land-use history and vegetation was almost always a significant factor influencing soil properties and transformations (Table 3). Soil nutrients and nitrification appeared to vary more by land use in hardwood stands than in conifer stands. Several explanations are possible: (1) sites presently occupied by hardwoods had a more intensive cultivation history (longer cultivation, higher rates of fertilization), (2) the variation in stand composition within the categories ''conifer'' and ''hardwood'' has an important effect on soil nutrient processes, and (3) nutrient use and allocation vary between hardwoods and conifers. To address the first point, the hardwood cultivated plot 134 (see Table 1) was classified as farmland from 1805 until 1908, and did have the highest mineral soil N and P content of all sites (1266 kg N/ha and 752 kg P/ha), while rates of net mineralization and nitrification were very similar to the other hardwood cultivated plot. The other cultivated sites were also farmed for $\geq$100 yr by the Sandersons and subsequent landowners (Raup and Carlson 1941, Raup 1966).

Although land-use history was a statistically significant factor in many of our measures, we acknowledge that greater replication would have increased the strength of our findings, since the history or ''treatment'' effects are possibly confounded with inherent site properties. We cannot reconstruct specific site histories because this level of detail is not available (e.g., crops grown, duration of different uses, or amount of manures added). However, we were still able to detect strong differences among our general land-use categories. Error values were generally low between replicates (with exceptions), indicating that the sites within a ''treatment'' were relatively similar.

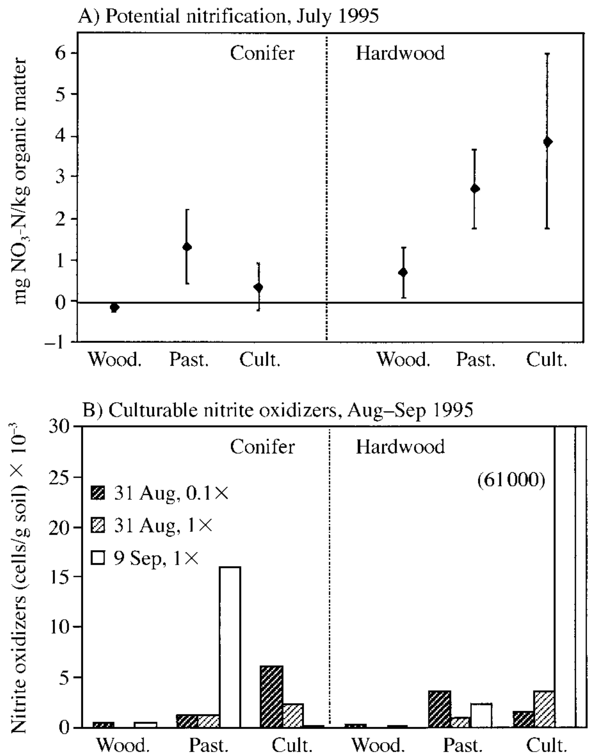


FIG. 8. Potential nitrification and most-probable-numbers (MPN) estimates of nitrite oxidizers. Error bars represent $\pm$1 SE for potential nitrification. For MPN estimates, media nitrite concentrations were full strength (1×) or diluted to one-tenth of those recommended (0.1×) in Schmidt and Belser (1994). The August MPN data were during the height of the drought, while the September sampling occurred after a rain.

Land-use history has influenced the distribution of several plant species across Prospect Hill (Motzkin et al. 1999), although not as dramatically as on a nearby low-fertility sand plain (Motzkin et al. 1996). It is possible that the presence of certain species is facilitated by agricultural practices at Prospect Hill. Out of a wide range of edaphic and disturbance factors at this site, the presence of *Prunus* spp. and *Acer saccharum* was best predicted by soil N and the C:N ratio (Motzkin et al. 1999). The variation in species composition within and between the ''hardwood'' and ''conifer'' categories could subsequently influence soil processes. Organic matter produced by the fertile-site species listed above can support higher N mineralization and nitrification rates than oaks and conifers (Zak et al. 1986, Boerner and Koslowsky 1989). While conifer woodlots were dominated by hemlocks and/or white pine, the conifer pastured plots were red pine (*Pinus resinosa*) plantations. Red pine litter has high lignin:N ratios and slow decomposition rates (Bockheim and Leide 1986), and the low soil pH, high forest floor and soil C, and high C:N ratios found here support this. Both conifer woodlots were dominated by hemlock (Table 1); the presence of hemlock has been shown to be associated with low nitrification rates (Mladenoff 1987).

Present vegetation (hardwood or conifer) was an important factor controlling many soil properties. Net N mineralization and chloroform-extractable N were greater in hardwood stands, and not consistently influenced by land-use history (Table 3). Soil microbial biomass may reflect present-day organic matter supply and quality, rather than total soil nutrient content.

Forest floor mass recovered more quickly after abandonment under conifers than under hardwoods. Litterfall may be slightly higher in conifer and hardwood stands (3.2 $Mg \cdot ha^{-1} \cdot yr^{-1}$ in red pine vs. 2.9 in oak–maple [Magill et al. 1997]); this combined with slower decomposition under conifers than hardwoods (Nadelhoffer et al. 1982, Berg and McClaugherty 1987) would promote more rapid accumulation of the forest floor under conifers. There also appeared to be slightly more mixing of the forest floor with underlying mineral soil in hardwoods, as evidenced by the higher ash content of the hardwood forest floor (Table 2). Recovery of soil C pools may be more rapid under conifers.

Importance of specific management practices

By altering site nutrients and increasing nitrification rates, we speculate that each of the three land uses we examined could have important long-term effects on carbon storage, nitrogen retention, and nutrient cycling. Values for forest floor C, N, and P in the pastured sites were generally intermediate between the cultivated and woodlot sites; while pasture soil C:N ratios and N transformations rates were more similar to woodlots. Nitrifier levels and activity were similar in cultivated and pastured sites (Fig. 8), but net nitrification was consistently higher in the cultivated sites, perhaps because of lower C:N ratios, as discussed above. Our results indicate that cultivation has the most persistent influence on soil nutrients and nitrification, perhaps driven by the addition of amendments combined with the depletion and subsequent slow accumulation of the forest floor and mineral soil light fraction organic matter.

Our findings are somewhat in contrast to the view that early New England agriculture decreased soil fertility (Cronon 1983, Merchant 1989). Agricultural practices in the early 1800s included manure amendments (Bidwell and Falconer 1941), which appear to have enriched soil N and P levels, and decreased C:N and C:P ratios. These amendments and lower C:N ratios persist over a century after abandonment, and may stimulate soil nitrification. However, agriculture also depleted forest floor and light fraction organic matter, and complete recovery of these levels has not occurred. We also have no information on soil erosion rates, which could have influenced soil fertility and affected aquatic ecosystems.

Implications for ecosystem recovery from disturbance

Few long-term studies of human disturbances on ecosystem processes exist, except in unusual cases, such as the long-term agricultural record at Rothamsted, UK (Leigh and Johnston 1994). Therefore we must rely on historical reconstruction, archaeology, dendrochronology, and paleoecology to see the long-term effects of disturbance (Foster et al. 1996, 1998, Entwistle et al. 1998, Fuller et al. 1998). Response to disturbance has been a major focus in ecology, but it is important to develop an understanding of not only immediate but long-term effects and recovery. A literature survey in 1984 found that site history was rarely mentioned in ecological studies (Hamburg and Sanford 1986); more recent studies document the persistent importance of agricultural impacts on vegetation dynamics, soil nutrient pools, and microbial activity (García-Montiel and Scatena 1994, Burke et al. 1995, Fernandes and Sanford 1995, Motzkin et al. 1996, Garcia et al. 1997, Koerner et al. 1997).

Consideration of site history is important in ecosystem process studies. Our study concludes that nutrient levels, microbial processes, and actively cycling organic matter fractions of cultivated sites may be distinctly different from less-disturbed sites even after a century of recovery via reforestation. The nature and specific impacts of a disturbance (i.e., cultivation vs. pasturing) are important in determining subsequent soil processes, as is the litter quality and productivity of the recovering plant community. Alteration of nutrient content, ratios, and form may continue to have long-term feedbacks (>100 yr) on soil organic matter dynamics and microbial populations. The underlying importance of site history deserves more thought and attention in the design of field studies and in our understanding of biogeochemical processes in dynamic forested landscapes.

Acknowledgments

The authors thank David Foster, Glenn Motzkin, and Arthur Allen for discussions of the study design and evaluation of land-use history. We gratefully acknowledge Adrien Elseroad and Laurel Schaider for their hard work in the field and lab. Robyn Long and Bethany Foster helped with the light fraction analyses, and Erica Cline initiated the nitrifier cultures. We also thank John Burk for assistance with the Harvard Forest archives. Steven Perakis, David Foster, Glenn Motzkin, Toby Hooker, Richard Bowden, Joseph Yavitt, and two anonymous reviewers provided critical feedback on the manuscript. This work was supported by the Andrew Mellon Foundation, the National Science Foundation LTER #BSR-8811764 and REU #BIR-9300219, the Rhode Island Agricultural Experiment Station, and is a contribution of the Harvard Forest Long-Term Ecosystem Research program.

Literature Cited

Aber, J., W. McDowell, K. Nadelhoffer, A. Magill, G. Berntsen, M. Kamakea, S. McNulty, W. Currie, L. Rustad, and I. Fernandez. 1998. Nitrogen saturation in temperate forest ecosystems: hypotheses revisited. BioScience **48**:921–34.

Berg, B., and C. McClaugherty. 1987. Nitrogen release in relation to the disappearance of lignin. Biogeochemistry **4**: 219–224.

Bidwell, P. W., and J. I. Falconer. 1941. History of agriculture in the northern United States 1620–1860. Peter Smith, New York, New York, USA.

Bockheim, J. G., and J. E. Leide. 1986. Litter and forest-floor dynamics in a *Pinus resinosa* plantation in Wisconsin. Plant and Soil **96**:393–406.

Boerner, R. E. J., and S. D. Koslowsky. 1989. Microsite variations in soil chemistry and nitrogen mineralization in a beech–maple forest. Soil Biology and Biochemistry **21**: 795–801.

Boone, R. D. 1992. Influence of sampling date and substrate on nitrogen mineralization: comparison of laboratory-incubation and buried-bag methods for two Massachusetts forest soils. Canadian Journal of Forest Research **22**:1895–1900.

Boone, R. D. 1994. Light-fraction soil organic matter: origin and contribution to net nitrogen mineralization. Soil Biology and Biochemistry **26**:1459–1468.

Brookes, P. C., A. Landman, G. Pruden, and D. S. Jenkinson. 1985. Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method to measure microbial biomass nitrogen in soil. Soil Biology and Biochemistry **17**:837–842.

Burke, I. C., W. K. Lauenroth, and D. P. Coffin. 1995. Soil organic matter recovery in semiarid grasslands: implications for the conservation reserve program. Ecological Applications **5**:93–801.

Cabrera, M. L., and M. H. Beare. 1993. Alkaline persulfate oxidation for determining total nitrogen in microbial biomass extracts. Soil Science Society of America Journal **57**: 1007–1012.

Cambardella, C., and E. Elliott. 1994. Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils. Soil Science Society of America Journal **58**:122–130.

Campbell, C. A., and W. Souster. 1982. Loss of organic matter and potentially mineralizable nitrogen from Saskatchewan soils due to cropping. Canadian Journal of Soil Science **62**:651–656.

Christensen, N., and T. MacAller. 1985. Soil mineral nitrogen transformations during succession in the piedmont of North Carolina. Soil Biology and Biochemistry **17**:675–681.

Collins, H. P., P. E. Rasmussen, and C. L. J. Douglas. 1992. Crop rotation and residue management effects on soil carbon and microbial dynamics. Soil Science Society of America Journal **56**:783–788.

Compton, J. E., R. D. Boone, G. Motzkin, and D. R. Foster. 1998. Soil carbon and nitrogen in a pine–oak sand plain in central Massachusetts: role of vegetation and land-use history. Oecologia **116**:536–542.

Cronon, W. S. 1983. Changes in the land. Hill and Wang, New York, New York, USA.

Davidson, E., and I. Ackerman. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry **20**:161–193.

Delcourt, H., and W. Harris. 1980. Carbon budget of the southeastern U. S. biota: analysis of historical change in trend from source to sink. Science **210**:321–323.

Delwiche, C. C., and P. L. Steyn. 1980. Nitrogen isotope fractionation in soils and microbial reactions. Environmental Science and Technology **4**:929–935.

Donaldson, J. M., and G. S. Henderson. 1989. A dilute medium to determine populations size of ammonium oxidizers in forest soils. Soil Science Society of America Journal **53**: 1608–1611.

Drinkwater, L. E., T. Wagoner, and M. Sarrantonio. 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature **396**:262–265.

Dunwiddie, P. W., and R. T. Leverett. 1996. Survey of old-growth forest in Massachusetts. Rhodora **98**:419–444.

Eno, C. F. 1960. Nitrate production in the field by incubating the soil in polyethlyene bags. Soil Science Society of America Proceedings **24**:277–279.

Entwistle, J. A., P. W. Abrahams, and R. A. Dodgshon. 1998. Multi-element analysis of soils from Scottish historical sites: interpreting land-use history through the physical and geochemical analysis of soil. Journal of Archaeological Science **25**:53–68.

Feger, K. H. 1992. Nitrogen cycling in two Norway spruce (*Picea abies*) ecosystems and effects of a $(NH_4)_2SO_4$ addition. Water, Air and Soil Pollution **61**:295–307.

Fenn, M. E., M. A. Poth, J. D. Aber, J. S. Baron, B. T. Bormann, D. W. Johnson, A. D. Lemly, S. G. McNulty, D. F. Ryan, and R. Stottlemyer. 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses and management strategies. Ecological Applications **8**:706–733.

Fernandes, D., and R. L. Sanford. 1995. Effects of recent land-use practices on soil nutrients and succession under tropical wet forest in Costa Rica. Conservation Biology **9**: 915–922.

Foster, D. R. 1992. Land-use history (1730–1990) and vegetation dynamics in central New England, USA. Journal of Ecology **80**:753–771.

Foster, D., G. Motzkin, and B. Slater. 1998. Land-use history as long-term broad-scale disturbance: regional forest dyanmics in central New England. Ecosystems **1**:96–119.

Foster, D. R., D. A. Orwig, and J. McLachlan. 1996. Ecological and conservation insights from retrospective studies of old-growth forests. Trends in Ecology and Evolution **11**: 419–424.

Fuller, J. L., D. R. Foster, J. S. McLachlan, and N. Drake. 1998. Impact of human activity on regional forest composition and dynamics in central New England. Ecosystems **1**:76–95.

Garcia, C., A. Roldan, and T. Hernandez. 1997. Changes in microbial activity after abandonment of cultivation in a semiarid mediterranean environment. Journal of Environmental Quality **26**:285–291.

Garća-Montiel, D. C., and F. N. Scatena. 1994. The effect of human activities on forest structure and composition in Puerto Rico. Forest Ecology and Management **63**:57–78.

Goulden, M. L., J. W. Munger, S.-M. Fan, S. C. Wofsy, and B. C. Daube. 1996. Exchange of carbon dioxide by a deciduous forest: response to interannual climate variability. Science **271**:1576–1578.

Haines, B. 1977. Nitrogen uptake: apparent pattern during old-field succession in southeastern U.S. Oecologia **26**: 295–303.

Hamburg, S. P. 1984. Effects of forest growth on soil nitrogen and organic matter pools following release from subsistence agriculture. Pages 145–158 *in* E. L. Stone, editor. Forest soils and treatment impacts. Proceedings of the Sixth North American Forest Soils Conference (June 1983). University of Tennessee, Knoxville, Tennessee, USA.

Hamburg, S. P., and Sanford, R. L. 1986. Disturbance, *Homo sapiens* and ecology. Bulletin of the Ecological Society of America **67**:169–171.

Handley, L. L., and J. A. Raven. 1992. The use of natural abundance of nitrogen isotopes in plant physiology and ecology. Plant, Cell and Environment **15**:965–985.

Hendricksen, C. I. 1933. The agricultural land available for forestry. Pages 151–169 *in* A national plan for American forestry (Copeland Report). Document No. 12. Vol. 1. U.S. Senate. U.S. Government Printing Office, Washington, D.C., USA.

Houghton, R. A. 1996. Land-use change and terrestrial carbon: the temporal record. Pages 117–134. *in* M. J. Apps and D. T. Price, editors. Forest ecosystems, forest management and the global carbon cycle. North Atlantic Treaty Organization Advanced Science Institute Series. Springer-Verlag, Berlin.

Houghton, R. A., J. E. Hobbie, J. M. Melillo, B. Moore, B.

J. Peterson, G. R. Shaver, and G. M. Woodwell. 1983. Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: a net release of CO_2 to the atmosphere. Ecological Monographs **53**:235–262.

Ihori, T., I. C. Burke, and P. B. Hook. 1995. Nitrogen mineralization in native cultivated and abandoned fields in shortgrass steppe. Plant and Soil **171**:203–208.

Janzen, H. H. 1987. Soil organic matter characteristics after long-term cropping to various spring wheat rotations. Canadian Journal of Soil Science **67**:845–856.

Janzen, H. H., C. A. Campbell, S. A. Brandt, G. P. Lafond, and L. Townley-Smith. 1992. Light-fraction organic matter in soils from long-term crop rotations. Soil Science Society of America Journal **56**:1799–1806.

Jenkinson, D. S., and A. E. Johnston. 1976. Soil organic matter in the Hoosfield continuous barley experiment. Pages 87–101 *in* Annual report for 1976, Part II. Rothamsted Experiment Station, Harpenden, UK.

Johnson, D. W. 1992. Effects of forest management on soil carbon storage. Water, Air and Soil Pollution **64**:83–120.

Kalisz, P. 1986. Soil properties of steep Appalachian old fields. Ecology **67**:1011–1023.

Kerley, S. J., and S. C. Jarvis. 1996. Preliminary studies of the impact of excreted N on cycling and uptake of N in pasture systems using natural abundance stable isotopic discrimination. Plant and Soil **178**:287–294.

Kielland, K., and J. P. Bryant. 1998. Moose herbivory in taiga: effects on biogeochemistry and vegetation dynamics in primary succession. Oikos **82**:377–383.

Koerner, W., J. L. Dupouey, E. Dambrine, and M. Benoît. 1997. Influence of past land use on the vegetation and soils of present day forest in the Vosges mountains, France. Journal of Ecology **85**:351–358.

Lamb, D. 1980. Soil nitrogen mineralization in a secondary rainforest succession. Oecologia **47**:257–263.

Leigh, R. A., and A. E. Johnston. 1994. Long-term experiments in agricultural and ecological sciences. CAB International, Wallingford, UK.

Lincoln, J. W. 1851. Farming in Worcester, Massachusetts. Pages 268–277*in* Report of the commissioner of patents for the year 1850. Part II. Agriculture. Printer to the House of Representatives, Washington, D. C., USA.

Macko, S. A., and N. E. Ostrom. 1994. Pollution studies using stable isotopes. Pages 45–62 *in* K. Lajtha and R. H. Michener, editors. Stable isotopes in ecology and environmental science. Blackwell Scientific, Oxford, UK.

Maclaren, J. P. 1996. Plantation forestry: its role as a carbon sink. Pages 117–134 *in* M. J. Apps and D. T. Price, editors. Forest ecosystems, forest management and the global carbon cycle. North Atlantic Treaty Organization Advanced Science Institute Series. Springer-Verlag, Berlin.

Magill, A. H., J. D. Aber, J. J. Hendricks, R. D. Bowden, J. M. Melillo, and P. A. Steudler. 1997. Biogeochemical response of forest ecosystems to simulated chronic nitrogen deposition. Ecological Applications **7**:402–415.

Merchant, C. 1989. Ecological revolutions: nature, gender, and science in New England. University of North Carolina Press, Chapel Hill, North Carolina, USA.

Mladenoff, D. J. 1987. Dynamics of nitrogen mineralization and nitrification in hemlock and hardwood treefall gaps. Ecology **68**:1171–1180.

Motzkin, G., D. Foster, A. Allen, J. Harrod, and R. Boone. 1996. Controlling site to evaluate history: vegetation patterns of a New England sand plain. Ecological Monographs **66**:345–365.

Motzkin, G., P. Wilson, D. R. Foster, and A. Allen. 1999.Vegetation patterns in heterogeneous landscapes: the important of history and environment. Journal of Vegetation Science **10**:903–920.

Nadelhoffer, K. J., J. D. Aber, and J. M. Melillo. 1982. Leaf-litter production and soil organic matter dynamics along a nitrogen-availability gradient in Southern Wisconsin (USA). Canadian Journal of Forest Research **13**:12–21.

Nadelhoffer, K. J., and B. Fry. 1988. Controls on natural nitrogen-15 and carbon-13 abundance in forest soil organic matter. Soil Science Society of America Journal **52**:1633–1640.

Nadelhoffer, K. J., and B. Fry. 1994. Nitrogen isotope studies in forest ecosystems. Pages 22–44 *in* K. Lajtha and R. Michener, editors. Stable isotopes in ecology and environmental science, Blackwell, Oxford, UK.

Parkinson, J. A., and S. E. Allen. 1975. A wet oxidation procedure suitable for the determination of nitrogen and mineral nutrients in biological material. Communications in Soil Science and Plant Analysis **6**:1–11.

Pastor, J., M. A. Stillwell, and D. Tilman. 1987. Nitrogen mineralization and nitrification in four Minnesota old fields. Oecologia **71**:481–485.

Piccolo, M. C., C. Neill, and C. C. Cerri. 1994. Natural abundance of ^{15}N in soils along forest-to-pasture chronosequences in the western Brazilian Amazon Basin. Oecologia **99**:112–117.

Post, W. M., and L. K. Mann. 1990. Changes in soil organic carbon and nitrogen as a result of cultivation. Pages 401–407 *in* A. F. Bowman, editor. Soil and the greenhouse effect, Wiley, New York, New York, USA.

Raup, H. 1966. The view from John Sanderson's farm. Forest History **10**:2–11.

Raup, H. M., and R. E. Carlson. 1941. The history of land use at the Harvard Forest. Bulletin No. 20. Harvard Forest, Petersham, Massachusetts, USA.

Richter, D. D., L. I. Barbar, M. A. Huston, and M. Jaeger. 1990. Effects of annual tillage on organic carbon in a fine-textured Udalf: the importance of root dynamics to soil carbon storage. Soil Science **149**:78–83.

Richter, D. D., D. Markewitz, J. K. Dunscomb, P. R. Heine, C. G. Wells, A. Stuanes, H. L. Allen, B. Urrego, K. Harrison, and G. Bonani. 1995. Carbon cycling in a loblolly pine forest: implications for the missing carbon sink and the concept of soil. Pages 233–251 *in* W. W. McFee and J. M. Kelly, editors. Carbon forms and functions in forest soils. Soil Science Society of America, Madison,Wisconsin, USA.

Richter, D. D., D. Markewitz, C. G. Wells, H. L. Allen, R. April, P. R. Heine, and B. Urrego. 1994. Soil chemical change during three decades in an old-field loblolly pine (*Pinus taeda* L.) ecosystem. Ecology **75**:1463–1473.

Robertson, G. P., and P. M. Vitousek. 1981. Nitrification potentials in primary and secondary succession. Ecology **62**: 376–386.

Russell, H. S. 1982. A long deep furrow: 400 years of farming in New England. University Press of New England, Hanover, New Hampshire, USA.

Sandor, J. A., and N. S. Eash. 1995. Ancient agricultural soils in the Andes of southern Peru. Soil Science Society of America Journal **59**:170–179.

Schiffman, P., and W. C. Johnson. 1989. Phytomass and detrital carbon storage during forest regrowth in the southeastern United States Piedmont. Canadian Journal of Forest Research **19**:69–78.

Schimel, D. S. 1986. Carbon and nitrogen turnover in adjacent grassland and cropland ecosystems. Biogeochemistry **2**:345–357.

Schmidt, E. L., and L. W. Belser. 1994. Autotrophic nitrifying bacteria. Pages 159–177 *in* R. W. Weaver, S. Angle, P. Bottomley, D. Bezdicek, S. Smith, A. Tabatabai, and A. Wollum, editors. Microbiological and biochemical properties. Part two. Methods of soil analysis. Volume five. Soil Science Society of America, Madison,Wisconsin, USA.

Selles, F., R. E. Karamanos, and K. E. Bowren. 1984. Changes in natural ^{15}N abundance of soils associated with tillage practices. Canadian Journal of Soil Science **64**:345–354.

Shearer, G., D. H. Kohl, and S.-H. Chien. 1978. The nitrogen-15 abundance in a wide variety of soils. Soil Science Society of America Journal **42**:89–902.

Silsbee, D. G., and G. L. Larson. 1982. Water quality of streams in the Great Smoky Mountains National Park. Hydrobiologia **89**:97–115.

Sollins, P., G. Spycher, and C. A. Glassman. 1984. Net nitrogen mineralization from light- and heavy-fraction forest soil organic matter. Soil Biology and Biochemistry **16**:31–37.

Spurr, S. H. 1950. Stand composition in the Harvard Forest as influenced by site and forest management. Dissertation. Yale University, New Haven, Connecticut, USA.

Spycher, G., P. Sollins, and S. Rose. 1983. Carbon and nitrogen in the light fraction of a forest soil: vertical distribution and seasonal patterns. Soil Science **135**:79–87.

Steele, K. W., and R. M. Daniel. 1978. Fractionation of nitrogen isotopes by animals: a further complication to the use of variations in the natural abundance of ^{15}N for tracer studies. Journal of Agricultural Science Cambridge **90**:7–9.

Strickland, T. C., and P. Sollins. 1987. Improved method for separating light- and heavy-fraction material from soil. Soil Science Society of America Journal **51**:1390–1393.

Tiessen, H., R. E. Karamanos, and J. W. B. Stewart. 1984. Natural nitrogen-15 abundance as an indicator of soil organic matter transformations in native and cultivated soils. Soil Science Society of America Journal **48**:312–315.

Tiessen, H., and J. W. B. Stewart. 1983. Cultivation effects on organic matter composition and size fractions. Soil Science Society of America Journal **47**:509–514.

Tiessen, H., J. W. B. Stewart, and J. R. Bettany. 1982. Cultivation effects on the amounts and concentrations of carbon, nitrogen, and phosphorus in grassland soils. Agronomy Journal **74**:831–835.

Vitousek, P. M., P. A. Matson, and K. Van Cleve. 1989. Nitrogen availability and nitrification during succession: primary, secondary and old-field seres. Plant and Soil **115**: 229–239.

Whitney, G. 1994. From coastal wilderness to fruited plain: a history of environmental changes in temperate North America, 1500 to the present. Cambridge University Press, Cambridge, UK.

Wilkinson, L. 1992. SYSTAT: the system for statistics. SYSTAT, Evanston, Illinois, USA.

Williams, M. 1990. Deforestation. Pages 179–201 *in* B. Turner, W. Clark, R. Kates, J. Richards, J. Mathews, and W. Meyer, editors. The earth as transformed by human action. Cambridge University Press, Cambridge, UK..

Woodmansee, R. G., and L. S. Wallach. 1981. Effects of fire regimes on biogeochemical cycles. Pages 649–669 *in* F. E. Clark and T. Rosswall, editors. Terrestrial nitrogen cycles. Volume 33. Swedish Natural Resource Science Council, Stockholm, Sweden.

Zak, D. R., D. F. Grigal, S. Gleeson, and D. Tilman. 1990. Carbon and nitrogen cycling during old-field succession: constraints on plant and microbial biomass. Biogeochemistry **11**:111–129.

Zak, D. R., K. S. Pregitzer, and G. E. Host. 1986. Landscape variation in nitrogen mineralization and nitrification. Canadian Journal of Forest Research **16**:1258–1263.

Exhibit C

Journal of Biogeography, **29**, 1279–1304

The forests of presettlement New England, USA: spatial and compositional patterns based on town proprietor surveys

Charles V. Cogbill[1]*, John Burk[2] and G. Motzkin[2] [1]*82 Walker Lane, Plainfield, VT, USA and* [2]*Harvard Forest, Harvard University, Petersham, MA, USA*

Abstract

Aim This study uses the combination of presettlement tree surveys and spatial analysis to produce an empirical reconstruction of tree species abundance and vegetation units at different scales in the original landscape.

Location The New England study area extends across eight physiographic sections, from the Appalachian Mountains to the Atlantic Coastal Plain. The data are drawn from 389 original towns in what are now seven states in the north-eastern United States. These towns have early land division records which document the witness trees growing in the town before European settlement (*c.* seventeenth to eighteenth century AD).

Methods Records of witness trees from presettlement surveys were collated from towns throughout the study area (1.3×10^5 km^2). Tree abundance was averaged over town-wide samples of multiple forest types, integrating proportions of taxa at a local scale (10^2 km^2). These data were summarized into genus groups over the sample towns, which were then mapped [geographical information system (GIS)], classified (Cluster Analysis) and ordinated [detrended correspondence analysis (DCA)]. Modern climatic and topographic variables were also derived from GIS analyses for each town and all town attributes were quantitatively compared. Distributions of both individual species and vegetation units were analysed and displayed for spatial analysis of vegetation structure.

Results The tally of 153,932 individual tree citations show a dominant latitudinal trend in the vegetation. Spatial patterns are concisely displayed as pie charts of genus composition arrayed on sampled towns. Detailed interpolated frequency surfaces show spatial patterns of range and abundance of the dominant taxa. Oak, spruce, hickory and chestnut reach distinctive range limits within the study area. Eight vegetation clusters are distinguished. The northern vegetation is a continuous geographical sequence typified by beech while the southern vegetation is an amorphous group typified by oak.

Main conclusions The wealth of information recorded in the New England town presettlement surveys is an ideal data base to elucidate the natural patterns of vegetation over an extensive spatial area. The timing, town-wide scale, expansive coverage, quantitative enumeration and unbiased estimates are critical advantages of proprietor lotting surveys in determining original tree distributions. This historical–geographical approach produces a vivid reconstruction of the natural vegetation and species distributions as portrayed on maps. The spatial, vegetational and environmental patterns all demonstrate a distinct 'tension zone' separating 'northern hardwood' and 'central hardwood' towns. The presettlement northern hardwood forests, absolutely dominated by beech, forms a continuum responding to a complex climatic gradient of altitude and latitude. The oak forests to the south are distinguished by non-zonal units, probably affected by fire. Although at the continental scale, the forests seem to be a broad transition, at a finer scale they respond to topography such as the major valleys or the northern mountains. This study resets some preconceptions about the original forest, such as the overestimation of the role of pine, hemlock and chestnut and the

*Correspondence: 82 Walker Lane, Plainfield, VT 05667 USA. E-mail: cogbill@sover.net

underestimation of the distinctiveness of the tension zone. Most importantly, the forests of the past and their empirical description provide a basis for many ecological, educational and management applications today.

Keywords
Historical ecology, New England, northern hardwood forest, plant biogeography, presettlement vegetation, proprietory town, surveyor's records, witness tree, vegetation classification, tension zone.

INTRODUCTION

Starting in 1620, settlers from Europe profoundly changed the supposedly 'infinite' woods of New England (Cronon, 1983). By 1850, the land in the current states of Connecticut, Rhode Island, Massachusetts, Vermont and New Hampshire had been completely settled and more than 60% of the land over the entire region cleared for agriculture (Harper, 1918). Areas remaining in forest, mostly on shallow soils, steeper slopes or low-productivity land, were being relentlessly harvested for timber by settlers and businessmen (Williams, 1989). Thus by the middle of the twentieth century, virtually the entire forest had been altered by human activities (Whitney, 1994).

Today less than a small fraction of 1% of the forest in the north-eastern United States remains as a few fragmented scraps of 'old-growth' landscape (Davis, 1996). Although reconstruction of the nature of the presettlement landscape is severely hampered by the lack of modern analogues, several approaches can be used to infer the nature of historical forests by extrapolating from modern data (Whitney, 1994; Russell, 1997; Egan *et al*., 2001). For example, much of New England is (re)forested today and it has been posited that the 'recovered' vegetation appears similar to the forests before 1775 (MacCleery, 1992). Moreover, several proxies of the ostensibly original forest have been drawn from different sources: 'virgin' remnants (e.g. Nichols, 1913; Cline & Spurr, 1942; Braun, 1950); successional tendencies and silvicultural experience in managed stands (e.g. Hawley & Hawes, 1912; Westveld *et al*., 1956); and theoretical models of forest development and response to the environment, particularly climate (e.g. Weaver & Clements, 1938; Bormann & Likens, 1979; Pacala *et al*., 1993). Yet each of these approaches has its limitations. 'Remnant' stands, by definition, have escaped expected natural disturbances and are atypical of the 'common' landscape at any time. The few uncut stands are a limited and selective spatial sample and survived exactly because they have unusual histories or extreme settings (Cogbill, 1996). Predictable physiological and silvicultural responses are dependent on temporal continuity and stable environmental conditions, which are constrained by species migrations, climate changes and soil development in glaciated regions (Russell, 1997). Models tend to be simplistic, deterministic and linear expressions of a few common stereotypes. Thus the current vegetation in New England is potentially a biased evidence of the past and the common surrogates of past forests have questionable applicability in quantifying forest vegetation before settlement.

Altogether inferential methods, such as modern vegetation, ecological models, and/or theoretical relationships to environment are problematic in determining a spatially comprehensive and temporally accurate view of the historic forest of New England. There are, however, empirical observations which describe the forests of the time. These historical data can also be used to test the accuracy of surrogate inferences. Contemporary observations of explorers, naturalists, diarists, authors and publicists abound, although they are subjective, limited in coverage and typically qualitative (Whitney, 1994; Russell, 1997; Bonnicksen, 2000; Edmonds, 2001). Significantly, the classical syntheses of the native vegetation across New England (e.g. Nichols, 1913; Bromley, 1935; Cline & Spurr, 1942; Braun, 1950; Westveld *et al*., 1956; Bormann & Buell, 1964) are based on a combination of anecdotal accounts, together with sampling in the last putative remnants and intensive field knowledge of the existing vegetation. Interestingly, despite explicitly linking these proverbial inferences (variously called 'natural', 'virgin', 'climax', 'original', 'primeval' or 'old-growth' forests) to the past, none of these studies compare their theorized species composition or pattern of forest types with actual historical data.

Fortunately, a spatially comprehensive and temporally relevant representation of past vegetation is contained in the land division surveys carried out in anticipation of European settlement. In the Midwest, the United States General Land Office (GLO) surveys have long been the primary resource for hundreds of studies of the historical landscape, and several states have recently digitized their entire survey database (Whitney, 1994; Whitney & DeCant, 2001). These federal surveys typically include descriptions of general species composition, changes in community units and reference to blazed trees marking predetermined points at intervals along the survey lines (e.g. Bourdo, 1956; White, 1984; Manies & Mladenoff, 2000). In contrast, in colonial New England and later in the original states, town-mediated surveys regularly cited only 'witness' trees. These unregulated and unstandardized eastern surveys have received relatively little interest, perhaps because they are obscure and are found in widely scattered repositories (e.g. McIntosh, 1962; Russell, 1981; Loeb, 1987; Seischab, 1990; Marks & Gardescu, 1992; Abrams & Ruffner, 1995; Cowell, 1995; Abrams & McCay, 1996; Cogbill, 2000; Black & Abrams, 2001). In New

England, case studies have analysed the presettlement composition for selected sections of Vermont (Siccama, 1963, 1971), Maine (Lorimer, 1977) and Massachusetts (Foster *et al.*, 1998; Burgi & Russell, 2000). In addition, Whitney (1994) has collated data from 145 presettlement surveys documenting witness trees across the northeast quarter of the United States and produced isopleth contour maps of equal presettlement abundance ('isowits') for fourteen taxa. These include data from seventeen studies representing twenty-five separate sites from New England and they display coarse zonal distributions on a continental scale.

Town proprietor surveys

In the eighteenth century, a distinctive land tenure system, the proprietory town, arose in the northern English colonies of North America (Clark, 1983; Price, 1995). The colonies, or later states, granted unsettled land in the form of regularly shaped 'towns', typically 6-mile square, to absentee groups of individuals. The work of this business 'proprietorship' was threefold: to divide the commonly held land into individual lots; to locate and mark those lots by a survey; and then to get settlers to move onto the lots and 'improve' them (Woodard, 1936). This sequence resulted in an unintended objective sample of the landscape before European settlement. Archived land division records, primarily lotting surveys citing 'witness' trees as permanent markers of the corners of small lots (1–160 acre; 0.5–65 ha), are available in many of the proprietory towns across the northeast (Whitney, 1994; Whitney & DeCant, 2001). Surveys of individual lots in the earlier granted towns of southern New England and the grants/patents/tracts/manors of New York share many of the same basic characteristics: placement of samples regularly across the town; marked trees to 'witness' lot corners or town divisions and a regular record in archival documents (Foster *et al.*, 1998; Cogbill, 2000).

The available New England lotting surveys date from just after the first established English settlement (1620) to after the Erie Canal (1825) enabled midwestern expansion. The first towns on the coast and in the Connecticut Valley typically tended not to cite witness trees, and some were surveyed before 1700 when citation became a consistent practice (Price, 1995). The frontier moved into the interior during the eighteenth century, but settlements were limited to the southern regions (roughly to the northern boundary of Massachusetts) until the end of the French and Indian Wars in 1763 (Clark, 1983). Northern New England towns were surveyed from *c.* 1770–1810. Some tracts in the northern mountains were never settled, but were granted as late as 1850. Although some surveys, especially in the southern coastal towns, were done after settlement began, the surveys overwhelmingly represent the undisturbed vegetation as European settlement proceeded through towns in the eighteenth century.

Lotting surveys of individual towns were usually completed quickly (i.e. 1–10 years), but the overall vegetation was sampled over a shifting period (1620–1850) spanning more than 200 years. Furthermore, the surveys were spatially transgressive, regularly progressing from the southern coast to the northern uplands. Therefore, spatial patterns could be confounded as a result of parallel, but variable temporal trends in forest conditions. The climate and natural disturbance regimes, such as hurricane occurrence, however, do not show obvious temporal trends across either the two century sampling period or the previous century of tree growth leading into it (Jones & Bradley, 1992; Boose *et al.*, 2001). Culturally during this period aboriginal populations were drastically reduced and indirect European activities, such as the fur trade and its effect on animal populations, were far-reaching. Nevertheless, the inherent longevity of trees and the relative remoteness of the woods insulated the forest itself from most anthropogenic influences. Significantly the processes that most affect the forest (i.e. clearing, logging, grazing, setting of fires) were very localized and closely tied to either indigenous or European settlement activities (Cronon, 1983; Whitney, 1994; Russell, 1997). Therefore, during the majority of the time over most of the study area there was minimal human disturbance before European settlement (Day, 1953). Nevertheless, coastal locations with the largest initial aboriginal populations were generally surveyed close to the time of maximum indigenous influence (Cronon, 1983). In addition, in this study, any effects of the native inhabitants are explicitly considered to be part of the 'original' pattern. Thus the timing of the survey sample is linked to the consistent conditions just before European settlement and represents a narrow spatial-temporal window, herein simply called 'presettlement'.

Vegetation structure

Vegetation patterns are a kaleidoscope of units which become more generalized as the resolution scales up from the tree species and their composition within a single community ($c.10^{-2}$ km^2 extent), through the regional assemblages of those communities in a landscape ($c.10^2$ km^2 extent), to groupings of those landscapes by similar physiognomy or constituent flora ($c.10^5$ km^2 extent). This nested hierarchy of plant community units can be loosely termed the 'community type', the 'association', and the 'formation', respectively (Whittaker, 1975; Delcourt *et al.*, 1983; Poiani *et al.*, 2000). The change in scale typically balances increasing variability and extent with decreasing detail and taxonomic specificity as the scale of resolution decreases (Turner *et al.*, 2001). Significantly, most geographical studies focus on broad-scale patterns among vegetation formations or species ranges, while ecological studies deal with the fine-scale patterns among types. The typical town-size sample already averages tree abundance over multiple forest types, and is thus an ideal scale to reflect the local proportion of trees, as well as species variation at the association scale (Delcourt & Delcourt, 1996).

The distinctiveness of the vegetation in the New England forest likewise depends on the scale of resolution. At a continental scale, plant geographers have traditionally viewed the region as reflecting a gradual transition between climatic 'biomes' or floristic 'provinces', blending the northern coniferous forest with the southern deciduous forest (Merriam, 1898; Gleason & Cronquist, 1964; Bailey, 1996). At an increased resolution, the vegetation of the northeastern United States has been viewed either as a distinct transitional 'formation' supporting a suite of endemic species (e.g. white pine, yellow birch, red spruce and hemlock) or as a pair of distinct 'associations' (i.e. Hemlock–White Pine–Northern Hardwoods and Oak–Chestnut or variously Oak–Hickory) within the deciduous forest formation (Nichols, 1935; Weaver & Clements, 1938; Braun, 1950; Jorgensen, 1971; Vankat, 1979; Delcourt & Delcourt, 2000). At the finest scale, plant ecologists generally find the land covered with a mosaic of individual community types responding to site-specific history or discrete site factors, such as topography, soil or geology (Siccama, 1971; Poiani *et al*., 2000; Thompson & Sorenson, 2000).

Although the vegetation of New England is composed of both discrete forest types and a blending of zones, there has been no agreement on the position or width of any vegetational boundaries in the region (e.g. Hawley & Hawes, 1912; Bromley, 1935; Braun, 1950; Westveld *et al*., 1956; Kuchler, 1964; Keys *et al*. 1995). Some of the uncertainty in the vegetation structure is because of its obfuscation by pervasive land use, but also important is the nature of transitions or 'ecotones' in a region of continuous forest (Weaver & Clements, 1938; Gosz, 1991). Although there are seldom discontinuities between higher order vegetation units (e.g. associations), in some cases, such as the transition between the prairie oak woodlands and northern hardwoods forest in Midwest, there is a narrow zone where distinct floristic provinces overlap and multiple species reach their range limits (Clements, 1905; Curtis, 1959; Grimm, 1984; Neilson, 1991). Curtis (1959) named this ecotone the 'tension zone' in Wisconsin. Because of the gentle and continuous geographical gradients, he associated the relatively sharp landscape boundary with climatic variables. Given the similar meeting of floristic provinces in New England, including the southern boundary of the same northern hardwood forest, the presence and location of such an ecotone, albeit abutting another forest type, is of particular interest.

Objectives

Despite a 200-year hiatus since the surveyors' initial collections, the contemporary quantitative sample of the forests of New England before European settlement has not been comprehensively analysed. Therefore, this study compiles and analyses witness tree tallies from available town-wide surveys in order to produce a spatial representation of species distributions and abundances across the region. The primary objective is to use the combination of town proprietor surveys and spatial analysis to produce an empirical reconstruction of the vegetation structure within the presettlement forests. This application of the historical–geographical approach is an ideal opportunity to document the vegetation at both a time (before confounding land use) and spatial scale (landscape) heretofore unavailable. Secondarily, by classifying the assemblages of species it identifies landscape patterns at various scales, including the potential presence of ecotones between adjacent vegetation associations. Finally, by correlating the composition of the vegetation with factors of the physical environment, this study investigates the influence of physical factors upon species and vegetation distribution at a regional scale before any recent changes in climate.

MATERIALS AND METHODS

Study area

A dense network of similarly sized towns from a contiguous area between the Androscoggin River in Maine and the Hudson River in New York have archived presettlement surveys (Fig. 1). This study area is herein broadly termed 'New England', although this term traditionally encompasses all of Maine, but excludes New York. The study area explicitly incorporates a tier of New York grants in adjacent areas in the Champlain and Hudson valleys, the Taconic Mountains, and on Long Island, and a group of erstwhile Massachusetts towns in what is now western Maine. The roughly rectangular block covers 1.3×10^5 km^2 and ranges from latitude 40°35′ N to 45°40′ N and longitude 69°55′ W to 73°55′ W.

The sample area (Fig. 1) lies across eight physiographic sections and incorporates varied geomorphology from the coastal plain along the Atlantic Ocean to the Appalachian Mountains (regularly to 1200 m a.s.l.) in the north and west (Fenneman, 1938). The topography of New England is mainly a rolling upland surmounted with residual ancient (Precambrian Period) mountains lowering gently towards the sea. The mountains consist of three discrete ranges, each a separate physiographic section. The eroded Taconic Mountain Section is along the New York border, the long-folded Green Mountain Section is in Vermont, and the younger, rugged (maximum 1916 m a.s.l.), intrusive White Mountain Section is in northern New Hampshire extending into adjacent Vermont and Maine. The bedrock geology of New England is commonly a metamorphic complex last uplifted in the Devonian Period and decidedly acidic in reaction. The major exceptions are the carbonate rocks of the Champlain Lowland Section, the Valley of Vermont (part of the Taconics) and, to a lesser extent, the eastern Vermont uplands. The Hudson Valley Section forms a distinct trough connecting with the Champlain Valley through Lake George. The flat Seaboard Lowland Section is a strip bordering the uplands in southern Maine, eastern Massachusetts, Rhode Island and southern Connecticut. Several major south-flowing river valleys (e.g. Hudson, Connecticut, Merrimack, Androscoggin) form low-altitude corridors through the dominant central New England Upland Section. The glaciers left a generally thin layer of stony glacial till over most of the area, except for post-glacial seabed in the Champlain Valley and central Maine lowlands and alluvium in major river valleys. Prominent moraines and outwash are

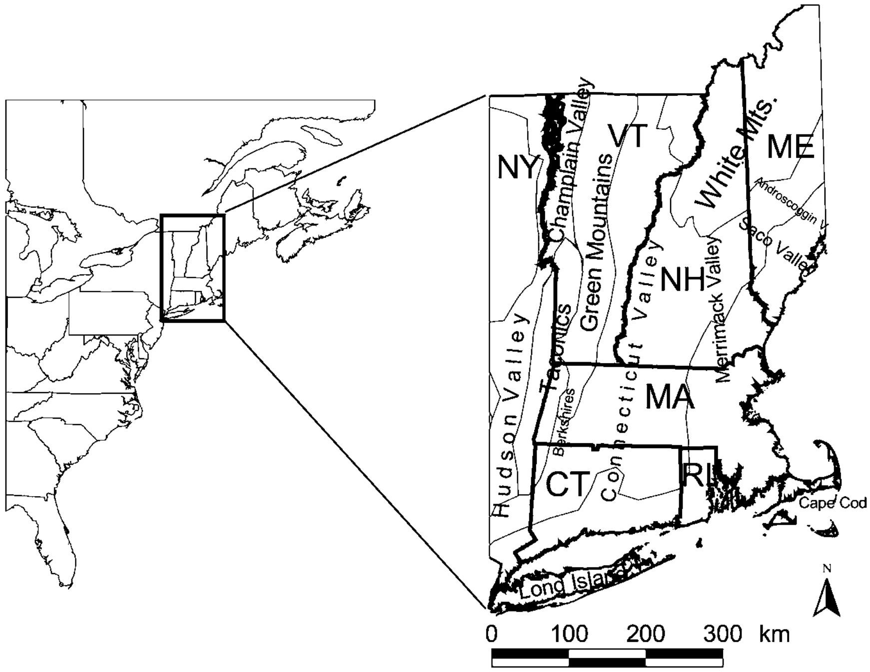


Figure 1 Study area on the east coast of North America and indicating the location of place names cited. State outlines in heavy lines New York (NY), Vermont (VT), New Hampshire (NH), Maine (ME), Massachusetts (MA), Connecticut (CT) and Rhode Island (RI). Outlines of the Physiographic Sections of Fenneman (1938) indicated in light lines.

found on Cape Cod and the Long Island Section. Glacial melt also left sandy outwash plains in some of the lowlands, particularly the Merrimack Valley. The climate varies latitudinally and altitudinally across the region, but all sites have a humid temperate continental climate with an even distribution of roughly 1100 mm annual precipitation with one-fourth as snow in the north (Trewartha, 1968). Mean annual temperatures vary from 5.5 °C in the north to 9.5 °C in the south. Most sites have cool summers, except at the southern edge of the region where summers warm to over 22 °C in July.

Town sample

We searched repositories of town records throughout the study area to locate proprietors' records, field books, manuscripts, maps and published records of town land surveys before settlement. Recorded surveys were regularly found in the Proprietors' Books typically housed with the land records in their respective town clerk's, county registry of deeds, or state/colonial public records offices. Lotting surveys are the primary authority for each town, but when they are lacking or cite a minimal number of trees, they were supplemented with other sources containing equivalent contemporary tree data. In general, all surveys from granting of the town until completion of the lotting of unsettled land, and if necessary, up to 80 years of deeds, were included in the data base. Each 'town' with fifty or more trees was considered to have an adequate sample to estimate town-wide composition (Bourdo, 1956; Whitney & DeCant, 2001). Moreover, in order to obtain a sufficient number of stems, surveys from adjacent towns were occasionally combined to produce a sample for a 'composite town'. All data were taken directly from primary documents or verbatim copies (i.e. transcriptions, microfilms, photocopies, printed records), except for ten town surveys which were already compiled in secondary sources (i.e. Torbert, 1935; Winer, 1955; Hamburg, 1984; Whitney & Davis, 1986; Loeb, 1987; Glitzenstein *et al.*, 1990).

The outlines of the original grants represent the sample units, nominally 'towns'. The boundaries of these 'parent' towns/tracts were derived from historical records, foremost being the initial surveys themselves. Many have changed names or been subsequently subdivided yielding multiple towns as they exist now (Melnyk, 1999). The surveys cover different parts of the original town and in many instances the exact positions of the sample trees were located. Despite this potential spatially explicit control, tallies were simply taken from the town as a whole at a nominal grain size of 100 km^2.

Tree sample

Objective conclusions depend on the tree records being an accurate representation of town vegetation. Land division documents were carefully read for any 'mentions' of trees within each town. All specifically identified witness or boundary trees were tallied by the name given by the surveyor. General listing of the composition of the vegetation or the numerous citations of survey 'posts' or 'stakes', even if identified by species, were excluded from the tally. Because of intermittent citation and difference in structure from trees, all identifiable shrub species were similarity not included in any totals. Thus the sample consisted only of trees actually growing, or dead standing, at predetermined survey points. Throughout proprietary lotting surveys, virtually all tree citations were of a single stem at each corner or end of an outline segment, on a variable (primarily a 20–40 ha) grid. Whenever possible, the sample points were located on an original lotting plat, and a special effort was made to avoid duplication of lots or trees shared between adjacent parcels. Some tallies did contain inadvertent 'multiple counting' of trees, but other than increasing the number this is inconsequential to the relative proportions of taxa from the town. Lotting surveys were not a random sample of the trees and they occurred in widely different patterns and intensities. The uncontrolled survey design produced samples at quasi-regular locations determined *a priori* and drawn from the whole town. These samples are assumed to be proportionally representative of the town-wide composition. As with much historical data, the sampling methods were variable and poorly documented, but there was apparently no consistent bias in the choice of trees, in tree identification, or in their spatial placement (Bourdo, 1956; Siccama, 1963; Grimm, 1984; Whitney, 1994; Russell, 1997; Cogbill, 2000; Whitney & DeCant, 2001). At face value, the lotting tree tallies were a reliable statistical sample and their frequencies were an unbiased estimate of overall forest composition of the towns before settlement.

Common names are used exclusively in this work. The land surveys invariably cited English colloquial names, and many of the surveys were carried out before the introduction of the Linnaean system in 1753. Surveyors were discerning naturalists and remarkably consistent; thus their vernacular usage is accepted (Cogbill, 2000). The surveyors, however, often did not distinguish species within genera and there is additional ambiguity with some unusual or questionably applied names. Thus for reduction in taxonomic uncertainty and consistency across all towns, the named trees are classified into widely represented genera. The categories are strict divisions by genus, except for groups defined by ambiguous surveyor's names. Ironwoods include both *Ostrya* and *Carpinus*. Cedar includes two genera, and as treated here, the present range disjunction separates *Thuja* (northern) from *Chamaecyparis* (Atlantic). 'Whitewood' is not classified as it certainly includes the sympatric *Liriodendron,* sporadically *Tilia* ('basswood'), some *Populus* (in part, now cottonwood), and possibly some *Acer* ('white maple'). Additionally, following the vernacular usage of the time, 'walnut' alone is considered under *Carya* and 'witch hazel' or 'hazel' is treated as *Ostrya* (Cogbill, 2000). All common names are corrected for equivalence in spelling and form and associated with the most exact scientific taxa as compiled in Appendix 1 where nomenclature follows Gleason & Cronquist (1991).

The relative frequency of each taxon across each town is an estimate of the presettlement abundance at that location. Because of the sample size of trees within towns, the precision of the frequency estimates is roughly one tree in 200, i.e. 0.5%. Restricted types or infrequent species were incompletely sampled, but the analyses explicitly focus on an accurate spatial estimate for the common species responsible for gross vegetational patterns. Moreover, scattered towns have samples of thousands of trees and this accounts for some estimates of the range (detection limit of about one tree in 1000, i.e. 0.1%) and the occurrence of uncommon taxa. The lumping of species within some genera necessarily loses resolution in taxa with several common species, such as birches (*Betula*), pines (*Pinus*), ashes (*Fraxinus*), or oaks (*Quercus*). To preserve some of the distinction between species, all separate vernacular names are maintained and their frequency by constancy (number of towns) and occurrence (number of trees) calculated (Appendix 1). Moreover, when congeneric species were specifically distinguished in the same survey, ratios between the numbers of trees in these taxa are used to estimate the local ratios of particular species.

Vegetation analysis

A matrix of proportions of genera within all the sample towns described the basic tree distribution and composition across the study area. Analysis of the vegetation structure and relationship among taxa was further elucidated by a reciprocal ordination of both taxa and towns with detrended correspondence analysis (DCA) using PC-ORD software (MjM Software Design, Gleneden Beach, OR) (McCune & Mefford, 1997). The towns were also classified using a Cluster Analysis (Euclidian distance measure with Ward's Method in PC-ORD). This clustering agglomerated a series of units by minimizing an 'objective' function of the variance among the average composition (centroids) of the groups (Legendre & Legendre, 1998). Both the patterns in the indirect ordination and the hierarchy within the classification allow for identification of the natural groups in the vegetation. The heterogeneity of the unit was calculated as the average Euclidian distance within the group. The vegetational similarity between groups was scaled by the objective distance at which they were joined (i.e. 'fusion distance') within the hierarchy (McCune & Mefford, 1997; Legendre & Legendre, 1998).

Additionally, basic geographical, topographic, and climatic parameters were analysed to determine their association with the vegetation and its divisions. Both Pearson's product–moment (r) and Kendall's rank (τ) correlations and scatter plots of taxa frequencies along the ordination axes and directly against environmental variables (with proportions arcsin transformed for normality) indicated the association of specific taxa with the environment (Sokal & Rohlf, 1981; McCune & Mefford, 1997). Similarly, correlations of environmental variables with the ordination axes indicated the importance of factors underlying the overall vegetation variation. Furthermore, multiple single classification ANOVAs were used to test for the significance and ranking of individual environmental variables in discriminating differences between the classification units (Sokal & Rohlf, 1981). Finally the vegetation composition was summarized by mean proportion of each taxon over all towns falling within defined classes such as state, type, cluster or environmental parameter.

Spatial analysis

In order to elucidate spatial patterns in the vegetation, several data layers in a geographical information system (GIS) were developed for the sample area. ARCVIEW® GIS 3.2a (Environmental Systems Research Institute, Inc., Redlands, CA) enabled analyses of the geographical distribution and relationships among the taxa groups, vegetation clusters and environmental factors across the sample area. The proportions of taxa were simply displayed as individual points positioned at the central geographical (latitude, longitude) coordinates of the sampled town. The spatial pattern of the abundance of individual taxa across the whole study area was also expressed as an integrated surface derived from the network of town samples. The SPATIAL ANALYST extension in ARCVIEW interpolated an abundance surface (0.02 degree grid using inverse squared-distance weighting of the five nearest neighbours) for each taxon. This surface smoothed relative frequency across the entire study area and displayed a continuous (roughly 6 km^2 scale of resolution) distribution preserving the linear combination of proportions in species composition. Several taxa reached their effective geographical limit within the study area. The current composite species ranges of these taxa were mapped for comparison with the presettlement sample. Tree atlases (Little, 1971, 1977) were the basic source for current distributional limits, and the distributions were supplemented using specific local studies (i.e. J. Goodlett & G. Zimmerman, unpubl. obs.; Manning, 1973; White & Cogbill, 1992). The distribution of presettlement vegetation at a regional scale was also compared with various modern classifications, vegetation maps and descriptions of forest types (e.g. Braun, 1950; Westveld *et al*., 1956; Kuchler, 1964; Bailey, 1976; Keys *et al*. 1995).

Environmental analysis

As environmental variables are more properly associated with the entire area of the sampled town, a series of environmental attributes of the town polygons were derived from spatial surfaces created by SPATIAL ANALYST. GIS resampling (mean 168 grid cells) of a USGS (0.01 degree) Digital Elevation Model (DEM) grid produced estimates of town-averaged topographic variables [mean altitude (m a.s.l.), maximum and minimum altitude, and standard deviation (SD) of altitude or 'ruggedness']. As variable local topography averaged out at the town scale, the integrated slope or aspect is a trivial horizontal surface. The potential direct solar radiation flux summed over a midsummer day (27 July) was indicative of the topographic moisture regime and was directly calculated from latitude (Frank & Lee, 1966). Furthermore, the distance in metres from the generalized coniferous/deciduous ecotone in the north-east determined an elevation index combining altitude and latitude [FCE (from conifer ecotone) elevation = Altitude (m a.s.l.) + 100 × Latitude (°) – 5129] (Cogbill & White, 1991).

Several GIS climatic surfaces were also created from a network of 255 climatological stations across the region (data from Climate Atlas of the Contiguous United States, National Climate Data Center, Meteorological Service of Canada). For each climate station on the prevailing land surface (excluding high-altitude stations), a Fourier analysis of the monthly mean temperatures, normalized to a 30-year (primarily 1960–90) period, determined a sinusoidal fit to the annual temperature curve expressed in three coefficients: the mean annual temperature (°C), the amplitude of the annual temperature curve (°C) and the seasonal lag of the curve (*d*) (Cogbill & White, 1991). SPATIAL ANALYST then created an interpolated surface (0.0262 degree grid using inverse squared-distance weighting of the five nearest neighbours) for each Fourier coefficient, as well as surfaces for the daily range of temperature and annual precipitation. All surfaces were resampled for all (mean 24) grid cells within each sample town to derive the mean and extreme values over the entire town. These averaged interpolations intentionally smoothed the variability across the area, but they were directly tied to the local empirical climate regime. They also represented complex climatic patterns, especially in variable terrain, much better than linear regressions (cf. Ollinger *et al*., 1995). Climate also varied with altitude, but the prevailing land surface best represented the average conditions experienced by the vegetation across a town. In addition, the inclusion of independent topographic parameters potentially accounted for altitude variation. The interpolated Fourier coefficients of mean (T_{bar}) and amplitude (T_{amp}) determined the estimated annual temperature curve for each cell or town which was evaluated for mean January temperature ($T_{bar} - T_{amp}$), mean July temperature ($T_{bar} + T_{amp}$), solved for the length of the frost-free growing season [G_{Season} = curve days (d) > 10 °C], or integrated for Growing Degree Days [GDD (°days) = area > 10 °C].

RESULTS

Sample towns

Overall 389 'towns' in the study area had surveys citing at least fifty witness trees (Table 1). Lotting surveys (1623–1850) contained in Proprietors' Records or Maps were the sole source for 79% of the sample (306 towns), while a minority of town compositions were derived from grant outlines or large subdivisions of the town (twenty towns), records in early deeds for the original sale of lots (seventeen towns), surveys of the course of the original roads in town (eight towns), or mixtures of more than one source (thirty-eight towns). The median town size of 119 km^2 is close to the classic 100 km^2 proprietory New England town. The eighteen 'composite towns' are larger than single towns and together with the relatively large older parent towns in southern New England produce a mean 'town' (sample grain) size of 150 km^2. Sample towns are spread across the entire region and are drawn from 45% of the actual land area. Coverage varies from *c*. 75% of the land sampled in southern New England to *c*. 25% of the land covered in the northern regions. The densest sample of towns is from eastern

Table 1 Characteristics of sampled towns and trees included in the New England study area

	State							
	ME	NH	VT	MA	CT and RI	NY	Overall	Range
Dates (AD)	1662–1835	1673–1850	1763–1820	1623–1835	1642–1818	1760–1811	1623–1850	
Towns (number)	21	47	97	118	67	39	389	
Town size mean (km^2)	214	144	117	134	175	207	150	23–1066
Trees tallied (stems)	4473	17,735	23,496	52,403	45,560	10,265	153,932	
Trees per town (median)	112	235	165	200	444	187	200	50–4404
Sample density (trees km^{-2})	1.4	2.9	2.4	3.7	3.8	2.1	2.6	0.09–49.5
Taxa cited per town (mean)	17.5	17.3	16.0	18.5	21.7	18.9	18.2	5–45
Area in sample (%)	26	28	46	76	77	31	45	

Connecticut, north-central Massachusetts and western Vermont, while the sparsest representation is from south-eastern Vermont, eastern New Hampshire and western Maine.

Sample trees

Overall, 153,932 trees were tallied in the study area with a mean of 396 (median 200) trees per town (Table 1). The sample density over the study area averaged 2.6 trees km^{-2}. This represented one tree every 38 ha which, not coincidentally, was nearly the size of a traditional 100 acre (40 ha) farm lot. Also, the sample intensity was only slightly less than the range (2.9–6.7) of sample trees km^{-2} found in public land surveys in other regions of the United States (Schwartz, 1994; Delcourt & Delcourt, 1996).

The early New England lotting surveys recorded 136 separable colloquial names for trees (Appendix 1). An additional sixteen shrub names (most abundant: sassafras, alder, willow and moosewood) were not included, but comprise only 0.3% of all stems. Some thirty-six of the tree names, representing only 214 (0.1%) stems, were odd descriptive combinations, enigmatic vernacular usage, or otherwise unrecognizable even as to genus (Cogbill, 2000). The 100 recognizable names could be combined by synonymy to document at least fifty-one distinct tree species found in these surveys. All these identified species were prominent members of the *c.* sixty-four species in the region's modern tree flora (Little, 1971, 1977). The most commonly used names were 'white oak' with 21% of all citations, followed by 'beech' (12%), 'black oak' (11%), 'hemlock' (7%), 'maple' (6%) and 'pine' (6%). Five genera contain multiple common species which were not distinguished; but, based on the ratio when specifically cited, the most named species in the generic groups were white oak (55% of named oaks), pitch pine (51% of named pines), rock or hard maple (66% of named maples), black birch (49% of named birches), and white ash (48% of named ashes) (Appendix 1).

Despite the lack of species distinctions, generic groups alone clearly express the regional composition. Altogether forty-five of the identifiable species and 99.7% of all the stems can be unequivocally placed into one of twenty-two categories. The uncommon, but still recognizable genera (e.g. apple, mountain ash, red cedar, mulberry) total only 193 stems or *c.* 0.1% of the sample. The number of taxa named in each town varied with the locality and the number of trees cited, but the towns have a mean of 18.2 separately named taxa incorporated into a mean of 10.8 generic groups found in each sample (Table 2). The richness of the tree names is greatest (21.7) in Connecticut towns (where up to 4400 trees were cited in one town) and least (16.0) in Vermont towns (Table 1).

The mean generic composition over the 389 towns is a grand-scale view of the vegetation in New England over 200 years ago (Table 2). Only fourteen genera, which were found in more than 30% of the towns, were regionally prominent. Oaks (*Quercus*), with a mixture of hickories (*Carya*) and chestnut (*Castanea*), were distinctive in the southern states, while beech (*Fagus*) with a mixture of hemlock (*Tsuga*), birch (*Betula*), spruces (*Picea*) and maples (*Acer*), typified the northern states. Among all towns, oaks (30%) and beech (17%) were absolute dominants. Eight other major taxa had much lower mean town proportions from 9 to 2.5% in descending order: pines, maples, hemlock, spruces, birches, hickories, chestnut and ashes. Together the ten most common genera also comprise 95% of the tree citations and all, except chestnut, were widespread, occurring in more than 60% of the towns. Interestingly, 50.1% of the tree abundance in the region, including the top two and the ninth ranked genera, are from a single plant family (Fagaceae). All genera beyond the twenty-two categories, except the composite 'whitewood', were found in less than twenty towns (5%). Despite being distinctive elements in the flora, other genera and their constituent species, were evidently inconsequential to the prevailing composition of the forest.

Oaks, beech, pines and spruces clearly dominated the vegetation in some towns, with > 50% maximum abundance. Although of moderate abundance, maples and birches were nearly ubiquitous, found in more than 90% of the towns. Additionally each of the common taxa (except for ash), plus fir (*Abies*), could be locally important, with > 20% maximum abundance in individual towns. A series of secondary genera [i.e. elms (*Ulmus*), ironwoods (*Ostrya* and *Carpinus*), basswood (*Tilia*), poplars (*Populus*), butternut (*Juglans*), cherries (*Prunus*)] were regularly (23–60% constancy) found in scattered towns, and despite having low (< 1%) overall representation, reached moderate (3–10% maximum) local abundance. Three conifer genera [fir,

Table 2 Average town-wide presettlement generic composition (relative frequency*) within New England states and among all sample towns

	ME	NH	VT	MA	CT and RI	NY		All Towns		
Towns	21	47	97	118	67	39	All trees	389	389	389
Sample stems	4473	17,735	23,496	52,403	45,560	10,265	153,932	Mean	Maximum	Constancy†
Taxa	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)
Oaks	16.1	11.8	2.1	45.4	59.5	33.7	39.5	30.2	82.1	80.2
Beech	10.6	25.5	36.0	6.1	3.0	15.7	12.1	16.6	68.2	75.6
Pines	14.6	12.3	2.2	15.9	3.1	8.5	9.6	9.0	69.8	75.3
Maples	10.7	9.4	15.3	6.4	4.1	7.8	7.7	9.0	30.8	94.9
Hemlock	8.7	14.7	11.1	6.3	2.6	9.8	6.7	8.4	39.6	74.3
Spruces	13.7	10.0	12.3	1.2	0.4	3.0	3.1	5.7	52.6	62.5
Birches	12.3	8.2	8.6	3.2	2.5	3.7	4.2	5.6	37.8	91.8
Hickories	0.2	0.3	0.2	4.5	9.8	4.3	5.2	3.6	23.7	59.9
Chestnut	0.0	0.8	0.1	4.5	8.6	3.7	4.4	3.3	31.7	46.5
Ashes	2.7	2.2	2.6	2.4	2.8	2.5	2.6	2.5	11.7	84.1
Fir	5.2	1.5	2.5	0.0	0.0	0.1	0.4	1.1	24.2	22.9
Elms	0.5	0.7	1.4	0.8	0.6	1.3	0.9	1.0	8.2	59.6
Basswood	0.4	0.9	1.9	0.4	0.3	1.2	0.7	0.9	7.6	49.9
Ironwoods	0.4	0.3	2.0	0.4	0.4	1.3	0.8	0.9	9.5	50.1
Poplars	2.0	0.9	0.2	1.2	0.9	0.8	0.9	0.9	10.0	55.5
Cedar (northern)	1.5	0.1	0.7			0.4	0.1	0.3	7.1	13.4
Butternut	0.0	0.1	0.3	0.1	0.3	0.4	0.2	0.2	8.6	22.6
Cherries	0.1	0.1	0.2	0.2	0.2	0.3	0.1	0.2	3.8	27.5
Cedar (Atlantic)	0.0	0.0		0.4	0.1	0.2	0.2	0.2	6.6	13.4
Pepperidge	0.0			0.0	0.2	0.3	0.1	0.1	5.5	8.7
Tamarack	0.2	0.1	0.1	0.0	0.0	0.2	0.0	0.1	5.7	5.1
Buttonwood	0.0	0.0	0.0	0.1	0.1	0.0	0.1	0.1	2.2	11.1
Other	0.4	0.1	0.1	0.5	0.6	0.7	0.3	0.4	–	–

*A 0.0 indicates trace (< 0.05%), while no entry indicates not recorded.
†Frequency of towns in which recorded.

northern cedar (*Thuja*), tamarack (*Larix*)] had low overall abundance, but could be important (> 5%) in the north. In the south, Atlantic cedar (*Chamaecyparis*) and pepperidge (*Nyssa*) had scattered low abundances.

Presettlement composition

The amalgamation of genera forming the presettlement vegetation of New England is dramatically displayed as pie charts of taxa proportions arrayed within the sampled towns (Fig. 2). Despite expected local variation, there was a strong spatial correlation in the compositions among towns. Pronounced geographical patterns were evident as the northern conifer species (greens: spruces, fir, cedar, tamarack) blended through 'northern hardwoods' (reds: beech, birches, maples) to 'central hardwoods' (yellows: oaks, chestnut, hickories) in the south. The temperate conifers (blues: pines, hemlock) fell roughly between the two hardwood sectors, with pines mixing to the south and hemlocks to the north. Although all towns had a decidedly mixed assemblage, four genera consistently characterized distinct geographical regions. The transition from spruce through beech to pine and then to oak prominence roughly paralleled the physiographic change from northern mountains through the uplands to the southern lowlands. Within this gross gradient, locally distinctive patterns, generally a more equitable mix of genera, appeared in the transitions, especially between the oak and beech sectors of the Champlain Valley, Taconics, southern Berkshires, north-central Massachusetts and west-central Maine. Major river valleys (i.e. Hudson, Connecticut, Merrimack) displayed a northward extension of oak and pine, while the uplands (western Massachusetts, south-western New Hampshire) displayed complementary tongues of beech and hemlock to the south.

Species distributions

The vegetation pattern was a composite of the individual genera, but each genus had a distinctive distribution and contributed differently to the overall pattern. Maps of interpolated grids of each of the twenty-two taxa displayed continuous surfaces of town-wide frequency across the study area. In addition to range limits, these distribution maps indicated the actual spatial pattern of the taxon's abundance across the range. The dense network of towns and their local consistency in composition resulted in a detailed, and remarkably simple, 'topographic' pattern of isowits. The explicit minimum resolution of these maps was the town scale ($c.100\ km^2$) and areas with fewer towns had more generalized patterns. For the eight common genera that had clear geographical patterns, these abundance surfaces were shown divided into classes of relative frequency beginning at

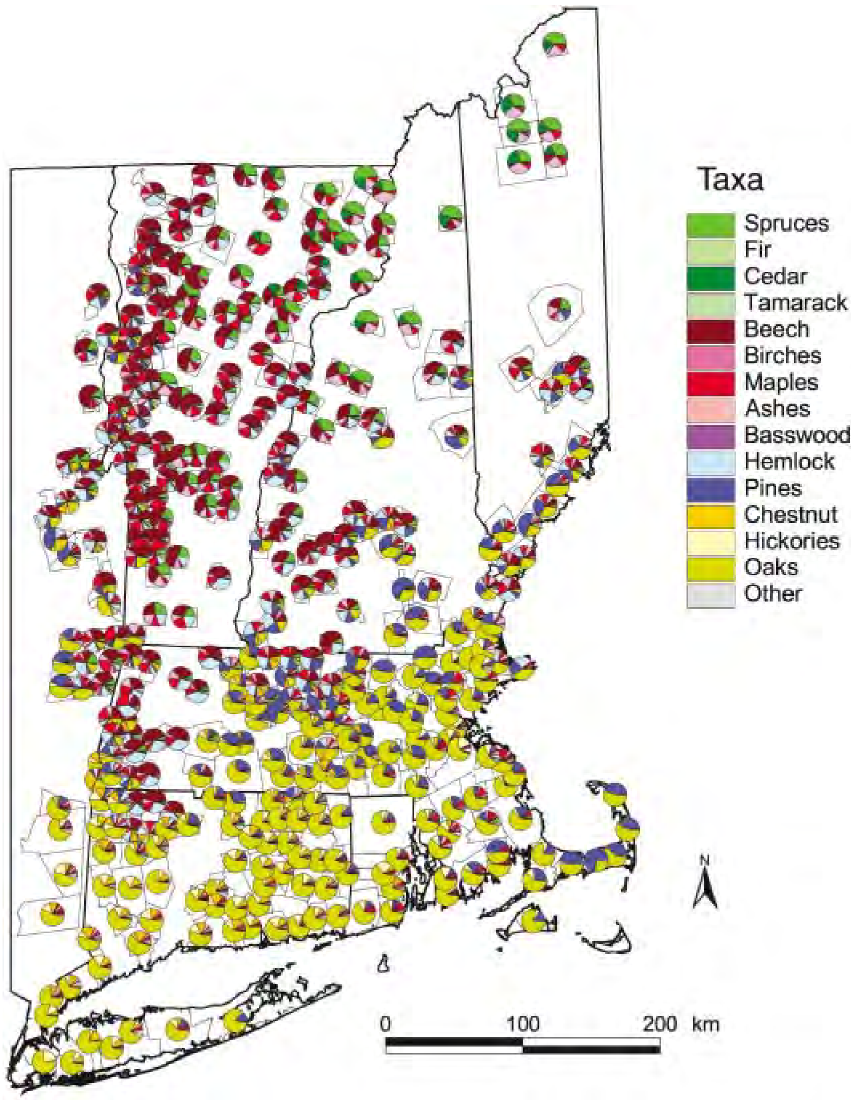


Figure 2 Spatial distribution of the proportions of genera in presettlement surveys arrayed on sample towns. Sample towns outlines indicated in light lines.

the general detection limit of 0.1% (Fig. 3). Individual towns with local extreme values appear as small 'dots' while more regional variability results in 'dappling' on the maps.

Four of the taxa, including the classic 'northern hardwoods' and their common associate hemlock had a distribution centred on the uplands. Beech (Fig. 3a) was the most abundant species throughout most of the uplands and reached its general maximum abundance in Vermont (> 50%). Beech and hemlock shared the dramatic boundary near the southern edge of the uplands and both, despite being within their ranges, fell below the detection limit in southern New England. Beech reappeared as a minor component near the coast. Hemlock (Fig. 3f) had a moderate, but variable abundance on the uplands, with minima (< 10%) in the Green Mountains and Taconics and a maximum presence in western Massachusetts (> 30%). Maple (Fig. 3b), the vast majority 'hard' or 'rock' maple, had the most widespread and equitable distribution of the common genera with a broad maximum (> 20%) centred on north-eastern Vermont and falling to < 5% in the southern lowlands. Birch (not shown) was the least abundant of the northern hardwoods and had a long ridge of maximum abundance in the mountains, varying from > 25% in north-western Maine and gradually decreasing through Vermont to a consistent 10% in the Taconics. Birch consisted of three species which presumably replaced each other as predominant across the genera's wide range. Black birch was clearly responsible for the greater 3% abundance in lowland New England, yellow birch accounted for most presence (> 10%) in the uplands, while white birch mixed with yellow birch combined for the maxima (> 25%) at higher elevations of the northern mountains.

Four conifers had restricted ranges in the northern edge of the study area. Spruce (Fig. 3e) dominated (> 40%) the northern White Mountain region. It was consistently mixed in tongues down the Green Mountains of Vermont (> 20%) and to a lesser extent into south-western New Hampshire (> 10%), and fell to low levels (< 5%) before reaching the

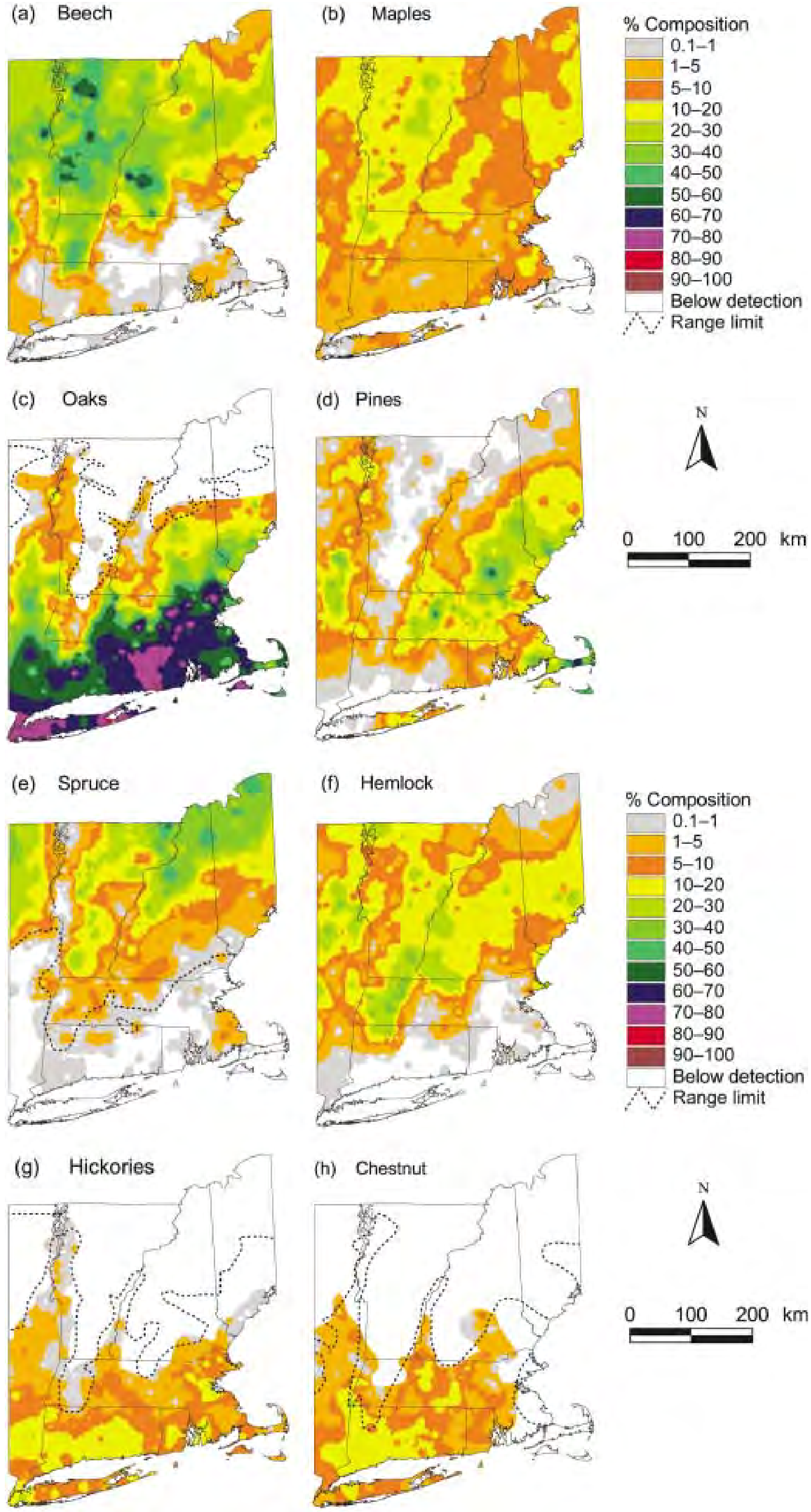


Figure 3 Maps of abundance surfaces of taxa: (a) beech, (b) maples, (c) oaks, and (d) pines, (e) spruces, (f) hemlock, (g) hickories and (h) chestnut in presettlement New England. The interpolated surfaces are shown divided into classes of relative frequency, from the general detection limit of 0.1%. The explicit minimum resolution is the town-scale (c.100 km^2). Modern composite species range distribution of (c) oaks from J. Goodlett & G. Zimmerman, unpubl. obs. and Little (1971), (e) red spruce from White & Cogbill (1992), (g) hickories from Manning (1973) and Little (1971), and (h) chestnut from Little (1977) are shown by dashed lines.

edge of the upland. A distinct corridor of low spruce abundance followed the Champlain Valley. The range limit of red spruce closely followed the 1% isowit, but there were significant occurrences of spruce, presumably black, in southern Connecticut and south-eastern Massachusetts. Fir, northern cedar and tamarack were much more restricted than spruce and were locally common only in the northern mountains with rather modest maxima (8–20%) in conjunction with spruce. Interestingly northern cedar also had a secondary maximum (3%) in the southern Champlain Valley, exactly where spruce was below detection.

Three 'central hardwood' (Leopold *et al*., 1998) genera had ranges restricted to southern New England. Oak (Fig. 3c) had tongues up the Hudson–Champlain, the Connecticut, and a broad extent in the Merrimack Valleys. Oak reached a consistent and remarkably high (> 70%) abundance from Long Island through eastern Connecticut into eastern Massachusetts. Many oak species contributed to this pattern, but the absolute dominant was clearly white oak with more than 67% of the citations in the heart of the oak forest in eastern Connecticut. Notable amounts of white oak also extended to the slopes of the White Mountains and into the Champlain Valley. The commonly combined pair, black and red oaks, were regular associates with white oak in many southern areas and red oak formed the range limit on the northern uplands. Despite being below the detection limit in the small range extensions into northern valleys and central Maine, oak's presettlement range closely traces its current range. Within its range, oak had low abundance (< 5%) in the uplands in western Massachusetts and southwestern New Hampshire, but is more important (> 20%) just to the south in the Taconics. Hickory (Fig. 3g) has a modern range which is similar to, but more restricted than oak and is virtually missing from New Hampshire. Its presettlement range actually followed the current range (a combination of shagbark and bitternut) fairly well, except for falling below detection in central Maine and the upper Connecticut Valley. Interestingly, hickories reached or exceeded the putative modern range in the Champlain Valley. Despite paralleling the oak range, hickories had much lower abundances and reached maxima (15%) both well to the west of oak in Connecticut and in spots in the eastern edge of oak's dominance in eastern Massachusetts. Hickories also barely penetrated the uplands and were in very low abundance in western Massachusetts. Chestnut (Fig. 3h) followed the hickories range very closely, but was missing from the immediate coast to the east. It was also below the limit of detection within its modern range in the Champlain Valley and in Maine. Chestnut has been much altered in abundance because of blight (Paillet, 2002), but had its prominent maxima (> 10%) in western Connecticut as did hickories, but differed in having a marked second maxima in central Massachusetts. Although possibly poorly documented today, chestnut was found beyond the mapped range in much of western Massachusetts and to a lesser extent in the Merrimack Valley. Curiously, chestnut displayed its maximum abundance in a town in the southwest corner of Massachusetts which is on the border of its ostensible range.

Pines (Fig. 3d) had a dramatic distribution with large extended patches pinching either side of central New England. The 10% isowit bounded roughly three polygons: the Hudson–Champlain corridor; an extensive band from central Massachusetts through southern Maine and Cape Cod. The nested maxima of high pine abundance were scattered in pockets in the Champlain (to 20% abundance with 90% of the named trees white pine), Hudson (to 30% with > 75% pitch pine), Connecticut (to 40% with 60% pitch pine), Merrimack (maxima > 50% with 67% pitch pine) and Saco (to 20% with 55% white pine) Valleys. In areas of maximum pine representation in the large southern valleys and on Cape Cod (> 50% pine, almost 100% pitch), pitch pine was clearly predominant. The proportion of white pine increased towards the northern valleys, in concert with a dramatic decrease in overall pine expression. Even on the uplands, white pine increased from *c*. 55% of the named pines in central Massachusetts to nearly all of the citations on the northern uplands, except the Taconics. Although within its overall range, white pine was remarkably uncommon (< 1%) on the New England uplands (Abrams, 2001). Furthermore, any pine species was uncommon or undetectable in a band slicing through Vermont, western Massachusetts, and spreading along the Connecticut coast.

Several other commonly cited genera had distributions independent of the pattern of major species. A series of common hardwood genera reached their modest maxima (< 5%) scattered throughout central New England. Ash, basswood, elm, butternut and ironwood taxa all had patchy but still widespread distributions (Table 2). All also had a joint prominent low peak in the Champlain Valley. Interestingly, each also had various secondary areas of relatively high abundance: both basswood and ironwoods in the upper Connecticut Valley, the Taconics and adjacent New York; ash in north-central Massachusetts and north-western Connecticut; elms in coastal New Hampshire and Massachusetts; and butternut in the Green Mountains and western Connecticut. Poplars had a southern distribution similar to pine, but were below detection in the mountains. Poplars maxima (> 2%) were scattered in central regions from north-central Connecticut to south-western Maine. Cherries had low importance (< 3%) and are scattered throughout the region. Buttonwood reached a maximum in Rhode Island. Two restricted (< 13% constancy) southern species just reached New England and were more common in 'swamps' reaching moderate maxima (7%) in scattered lowland towns. Atlantic cedar was tightly restricted to the southeast coastal area, reaching maxima (> 2%) in southeastern Massachusetts and Long Island. Pepperidge was rare in Massachusetts, but was relatively more abundant (> 0.5%) in eastern Connecticut and especially (> 1%) on Long Island.

Town classification

The visual overview of vegetation (Fig. 2), together with the patterns of individual genera (Fig. 3), were reiterated by a formal classification of the overall vegetation composition of the 389 towns. The final steps in the Cluster Analysis defined eight separate clusters that correspond well, in number and spatial consistency, to perceived vegetation defined units (Table 3). Three levels of the hierarchy produced objectively derived units at various orders (notably two, four and eight groupings). The dendrogram (Fig. 4) shows the nested rela-

Table 3 Average town-wide presettlement generic composition (relative frequency*) of classification clusters in New England

	Cluster							
	1	2	3	4	5	6	7	8
Towns	14	49	55	62	48	25	91	45
Trees	1441	11,230	17,004	17,603	21,149	11,753	50,022	23,730
Taxa	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)
Cedar (northern)	2.6	0.6	0.3	0.4	0.1			
Fir	15.6	3.2	0.6	0.3	0.1	0.0		
Tamarack	0.5	0.1	0.0	0.1	0.1		0.0	
Spruces	39.4	22.1	3.6	3.8	1.7	1.1	0.5	0.1
Birches	21.9	10.6	6.4	6.6	5.9	1.6	2.2	1.2
Beech	8.7	33.9	45.5	26.5	7.9	2.4	0.8	0.3
Hemlock	2.6	11.8	12.0	21.5	10.4	1.9	0.9	0.4
Basswood	0.1	0.7	2.4	1.7	0.6	0.3	0.3	0.1
Ironwoods	0.0	0.7	1.9	2.0	0.6	0.4	0.3	0.2
Maples	7.0	12.6	16.0	14.1	8.6	4.9	4.1	2.4
Ashes	0.7	1.4	2.8	4.0	3.5	1.8	2.4	1.5
Poplars	0.1	0.2	0.4	0.8	1.7	1.1	1.1	0.9
Pines	0.4	0.9	2.2	7.5	19.1	37.9	8.1	6.2
Elms		0.6	1.7	1.3	1.1	0.7	0.8	0.6
Butternut		0.2	0.4	0.2	0.4	0.1	0.2	0.0
Cherries		0.2	0.1	0.2	0.3	0.1	0.2	0.1
Buttonwood		0.0	0.1	0.0	0.1	0.0	0.1	0.1
Oaks		0.3	2.7	6.8	30.5	41.5	58.9	73.5
Chestnut			0.4	1.5	5.3	1.3	7.9	3.9
Hickories			0.3	0.5	1.8	2.1	9.8	7.1
Cedar (Atlantic)			0.0		0.1	0.3	0.4	0.3
Pepperidge					0.0		0.1	0.2

*A 0.0 indicates trace (< 0.05%), while no entry indicates not recorded.

tionships of the cluster groups to one another. For example, clusters 7 and 8 were combined at the lowest order [fusion distance (FD) of 10], while the most distinctive single cluster (cluster 1) remained uncombined the longest. The highest division (last combination FD = 63) separated the eight objective clusters into two primary four cluster groups. In turn, a prominent secondary division (FD = 24) split the southern group into two pairs: clusters 7, 8 (yellows) and clusters 5, 6 (blues). The four northern clusters were closely allied, but a secondary division (FD = 17) divided a mountain cluster (cluster 1, green) from an upland group (clusters 2–4, red). The three upland clusters formed a triplet series with a nearly equal tertiary separation (FD = 13–14). The two southern 'parallel' pairs of clusters were each distinguished at a slightly lower tertiary level (FD = 10–11), and consisted of both the most variable [cluster 5: mean Euclidean distance (ED) = 0.27] and the most homogenous (cluster 8: ED = 0.12) single clusters. By the tertiary level of division in the cluster classification (Fig. 4), the clusters became relatively homogenous, but membership of particular towns within the units became fuzzy (Brown, 1998). Moreover, vegetational unity and geographical coherence weaken with more than eight clusters.

The composite composition (centroid) of the fourteen to ninety-one towns in each cluster demonstrated the slowly shifting importance of specific genera in typifying these units (Table 3). The overriding division was between upland 'northern hardwood' beech, clusters 1–4 and southern lowland 'central hardwood' oak, clusters 5–8. Altogether secondary clustering produced three distinct forest types typified by dominant genus (> 26% mean): spruce (cluster 1), beech (clusters 2–4), or oak (clusters 5–8). At this same level of distinction, the oak cluster was split into a high oak pair (clusters 5, 6) and a transitional pair (clusters 7, 8) with high pine abundance.

The four northern clusters formed a loose concentric pattern in relation to the mountains and were further distinguished by the amount of beech (Fig. 4; Table 3). The spruce cluster 1, was composed of the high conifer towns with the least beech. This mountain cluster had > 50% combined spruce and fir composition with a mixture (*c*. 35%, primarily birch) of northern hardwoods. Cluster 3 formed the nucleus of the northern hardwood triplet with composition totally dominated by beech (46%). A tight 'chain' was formed, first linking with the moderate spruce (22%) cluster 2, then the higher hemlock (21%) cluster 4, and finally with the most distinctive cluster 1. Cluster 2 was found throughout the lower mountains and higher hill towns, cluster 3 was prominent on the lower hills around the perimeter of the mountains, while cluster 4 generally lay in valleys still within the uplands or beyond in the northern lowlands (Fig. 4). Thus the resultant classic northern hardwood unit (clusters 2–4, red) covered much of northern New England and had both a mixed spruce component from the higher elevations and an equitable mixture of hardwood species, including some oak and pine from the lowlands.

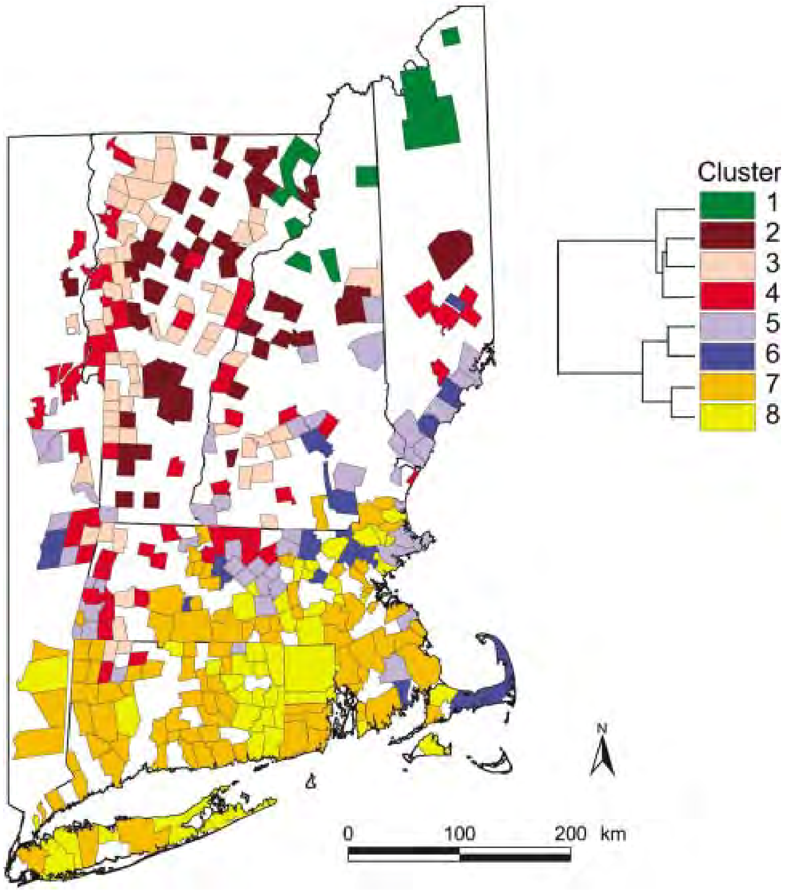


Figure 4 Geographical distribution of towns as classified by Cluster Analysis into eight clusters. The dendrogram next to the legend is scaled proportional to objective distance at which the cluster units are joined (i.e. 'fusion distance') in the hierarchy.

Except for the secondary separation of northern 'transition' oak-pine, the four oak vegetation clusters had weak spatial coherence (Fig. 4). The oak 'transition' (clusters 5, 6) of central regions had a higher component of pine (> 15%) than the typical oak forest (clusters 7, 8) of southern New England (Table 3). The two oak pairs were further separated by major compositional differences in oak. The southern 'typical' oak pair was first split into cluster 8 of very high (73%) oak composition and cluster 7 with less oak (59%) and more balanced composition, including more chestnut. The 'transitional' oak-pine was a combination of a small scattered cluster 6 of high pine (38%) and cluster 5 with low oak (31%). The latter cluster was still typified by oak-pine, but had a distinct mixture of both northern (maples, hemlock) and other southern elements (chestnut).

Ordination

The variation among town compositions was further illustrated by an indirect ordination (DCA) of the towns and genera. The compositional matrix was reduced in dimension so that the 389 towns and twenty-two genera were located along two axes in relation to their similarity (Fig. 5). The constellation of sites showed two clouds along the first axis; the northern sites form a continuous series (from clusters 1–4) to the right side, while the southern oaks (clusters 7, 8) formed a tight aggregation at the other pole. Significantly the variable transitional oak, cluster 5, lay in the thin area just left of centre and the two primary groups could be cleanly separated (clear gap between cluster 4 and cluster 5). The second axis separated the distinct pine dominance (cluster 6) at one end from the secondary southern hardwood genera (chestnut, pepperidge, hickories) drawing some of cluster 7 to the lower pole. In the centre of the space the second axis was muted, but the 'richer' genera (e.g. butternut, ironwood, ash, basswood, elm) drew elements of cluster 4 downward while hemlock was above the axis with other towns of cluster 4, which had a significant mix of hemlock within the hardwood forest. Thus, the ordination confirmed the overall integrity and distinctiveness of the individual clusters, as well as the primacy of the division between the northern and the central hardwoods. At the same time there was obvious variability within the clusters with significant compositional overlap between most adjacent clusters (e.g. clusters 7, 8 and clusters 2–4). The vegetation apparently formed a continuous gradient, particularly in the north (right on ordination), with a change from spruce and fir through beech, hemlock and maple composition. The oak clusters deviated from this gradient and had a more variable composition, especially near the transition.

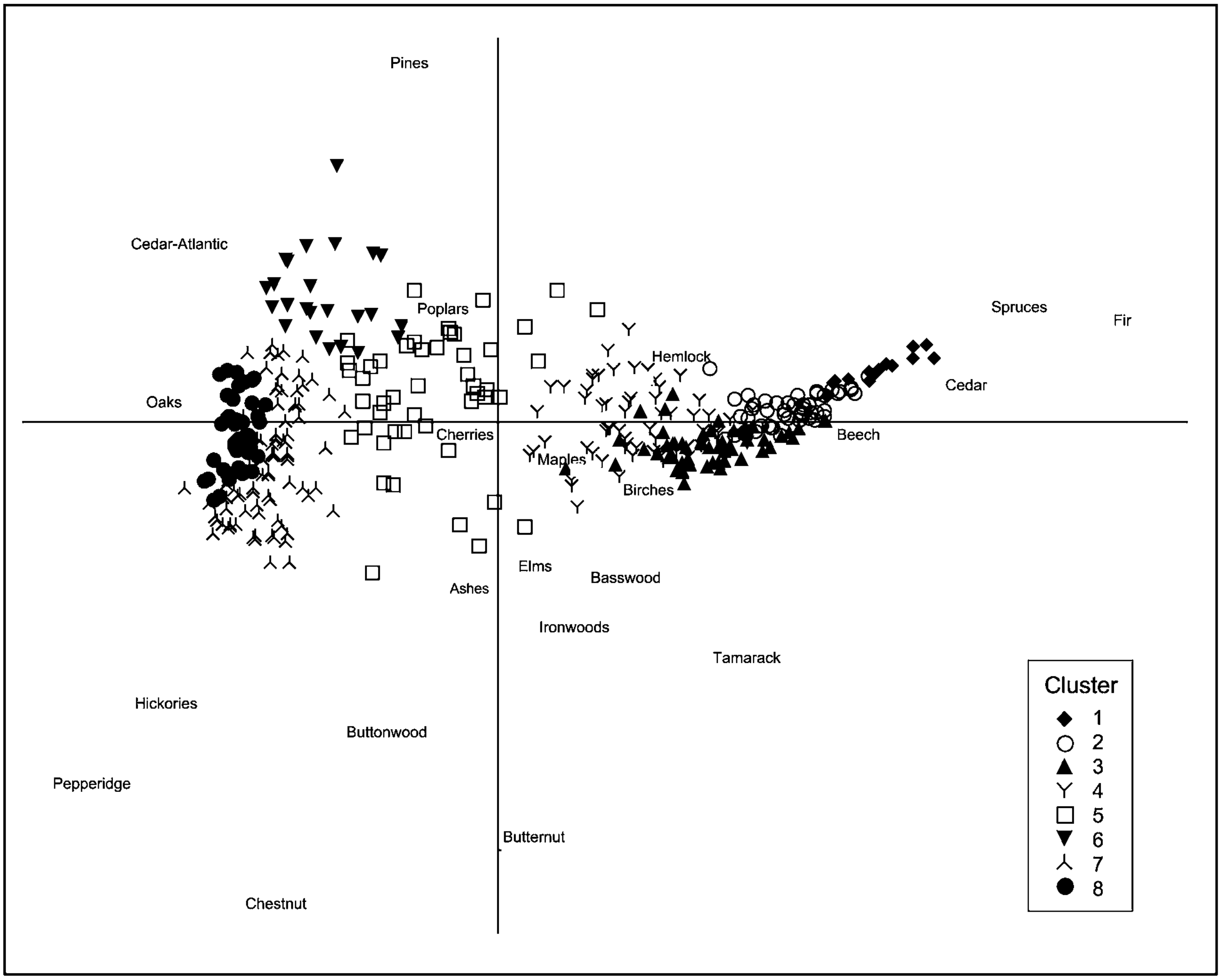


Figure 5 Detrended correspondence analysis ordination of twenty-two genera and 389 towns in the presettlement surveys of New England. Towns are classified by their cluster identity and the genus groupings are labelled at their centroid.

Environmental correlates

The classification and ordination of the presettlement vegetation were driven by compositional patterns, but the underlying relationship to environmental factors was undeniably strong. Each town was characterized by three geographical, five topographic and nine climatic parameters. Their overall statistical summary was a composite of the environment of the study area (Table 4). As ANOVAs for all variables indicated high statistical significance ($P < 0.001$, $F > 3.47$), the town clusters differ markedly from one another on the basis of any of the environmental parameters. There is a tight cocorrelation between most of these environmental variables (average $r^2 = 0.36$ and 0.71 among the climatic variables), so many of the associations are redundant (Table 4). The best discriminator by far, based on ranked ANOVA F-values ($F = 237$), was the FCE-elevation (a composite topographic-geographical variable closely correlated with temperature regime) followed by three other allied climate variables, headed by mean annual temperature ($F = 168$). The highest ranked variables of other types [i.e. solar (geographical) and maximum altitude (topographic)] were mixed in with a host of highly significant, but less discriminating ($F < 125$) parameters. The mean environmental parameters of the towns formed a smooth environmental series southward past the vegetation transition into oak. The difference between cluster means (Table 4) indicated the maximum statistical separation was between the spruce (cluster 1) and the northern hardwoods (cluster 2), and secondarily between the upland (cluster 4) and the lowland series (cluster 5). Although still significantly different [$P \leqslant 0.01$, determined by ANOVA least significant differences (LSD)], the oak transition environment (cluster 5) was marginally closer to the northern sites than the neighbouring southern oak cluster (cluster 7). Significantly, there were no statistical environmental differences ($P \geqslant 0.01$) between the two southern oak (clusters 7, 8) clusters. This mirrored the ordination (Fig. 5) diagram where there was a continuous

Table 4 Characteristics of environmental variables across 389 New England towns and individual clusters

	All towns				Cluster									
					1	2	3	4	5	6	7	8		
$n =$	389	389	389	389	14	49	55	62	48	25	91	45	LSD*	$F^{\dagger}$
Parameter	Mean	Min.	Max.	SD	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	$n_0 = 47.2$	d.f. = [7,381]
FCE-Elevation‡ (m)	−610	−1044	41	244	−98	−265	−442	−543	−676	−781	−823	−842	57	237
Tbar§ (°C)	7.8	3.3	12.2	1.6	4.3	5.9	6.8	7.3	8.0	8.6	9.4	9.4	0.4	168
GSeason** (days)	164	128	203	14	136	148	155	159	165	171	177	177	4	157
Jan. Temperature (°C)	−5.6	−11.4	−0.2	2.3	−10.3	−8.2	−7.2	−6.6	−5.3	−4.3	−3.4	−3.3	0.6	155
GDD†† (°days)	2230	1286	3294	334	1513	1821	2048	2156	2234	2366	2535	2527	97	130
Solar‡‡ (Ly)	966	959	970	2	961	963	964	965	966	967	968	968	1	123
Latitude (°N)	42.9	40.7	45.6	1.1	44.9	44.0	43.7	43.3	42.8	42.6	41.9	41.9	0.3	122
July Temperature (°C)	21.2	18.0	24.5	1.1	18.8	19.9	20.7	21.1	21.2	21.5	22.1	22.0	0.3	94
Altitude max. (m a.s.l)	415	29	1351	298	890	844	586	441	327	162	224	179	97	93
Altitude mean (m a.s.l)	229	8	709	170	542	460	317	255	172	91	120	98	56	89
Tamp§§ (°C)	13.4	10.8	15.0	0.8	14.6	14.0	14.0	13.8	13.2	12.9	12.7	12.6	0.3	75
Altitude min. (m a.s.l)	118	0	457	108	360	236	167	129	97	45	49	42	39	67
Ruggedness*** (m)	66	6	248	49	112	133	90	70	51	27	42	31	19	51
Precipitation (mm)	1091	843	1317	101	1016	1031	1015	1035	1115	1094	1169	1170	41	39
Seasonal lag (days)	30.6	28.3	38.5	1.5	30.3	29.8	29.7	29.9	30.6	31.9	31.3	31.6	0.7	22
Longitude (°W)	72.2	73.8	70.0	0.9	71.2	72.4	72.8	72.7	72.0	71.3	72.3	71.9	0.4	15
Daily Range (°C)	11.8	7.6	14.1	1.2	12.3	11.8	11.6	12.1	12.2	11.4	11.6	11.4	0.6	4

*Least significant difference between group means at $P = 0.01$, LSD $= 2.58 \times \sqrt{(2 \times n^{-1} \times MS_{within})}$.
†ANOVA variance ratio, for $P = 0.001$, $F_{[7,381]} = 3.47$.
‡Elevation from the coniferous ecotone.
§Mean annual temperature.
**Growing season length > 10 °C.
††Growing degree days > 10 °C.
‡‡Potential direct beam flux summed for 27 July (in Langleys day^{-1}; 1 Ly = 4.18 J cm^{-2}).
§§Amplitude of the annual temperature curve.
***SD of altitude.

gradient on the first axis until the three southern clusters were reached. The clusters then spread on the second axis.

The trends in environmental significance were formalized by the correlation analysis of the same variables in the ordination. All environmental vectors, except longitude, projected into the ordination space (not shown) were rotated only a few degrees counterclockwise (FCE-elevation was virtually coincident) from the first ordination axis. The highest correlation with the first ordination axis was with FCE-elevation ($r^2 = 0.82$, $\tau = -0.73$) followed by mean temperature ($r^2 = 0.76$, $\tau = -0.69$) and several similar climatic variables, with latitude the best geographical correlate ($r^2 = 0.71$, $\tau = 0.64$). Interestingly, the second ordination axis had low significance against most environmental variables ($r^2 < 0.04$), but a high correlation with longitude ($r^2 = 0.29$, $\tau = 0.37$), primarily reflecting the strong tendency of the high pine clusters to be on the east side of the area.

The responses of genera to the environment were quantified by direct gradient analysis of the composition (coenocline) along the environmental axes (Whittaker, 1975). All genera except ash, cherry, elm and butternut had statistically significant (generally $0.25 < r^2 < 0.60$) correlations with the suite of environmental variables. Depending on the taxa, the highest correlations were found in either FCE-elevation, mean annual temperature, January temperature, or latitude. The average town generic composition within FCE-elevation classes displayed the broadly overlapping ranges of the common species (Fig. 6). The dominance of oaks below 600 m FCE-elevation and the dominance of beech to just (150 m FCE-elevation) below the actual coniferous ecotone where it was exceeded by spruce is clearly demonstrated. Similar plots against latitude (Fig. 7) had exactly the same pattern with crossover at 43.0°N and 45.0°N. While three major 'zones' were evident both latitudinally and elevationally, variation was subsumed in the mean values, and the clusters of specific composition were obscured. The response of the three dominant species to mean annual temperature (Fig. 8) showed the same statistically significant responses (oak $r^2 = 0.69$, spruce $r^2 = 0.48$, and beech $r^2 = 0.37$, lower as a result of normal distribution). The average curves still crossed (at 7.3 and 4.5 °C), but the details, such as large variance in all species and patches of low abundance (e.g. 'warm' spruce) began to appear. Most dramatic was the consistency of oak in the warm sector and its decline below 8 °C.

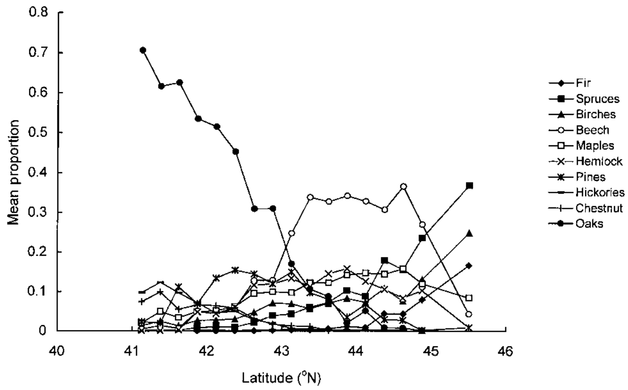


Figure 7 Coenocline plot of town average presettlement composition against classes of latitude. Mean of town-wide relative frequencies are plotted at the midpoint of 0.25 degree classes.

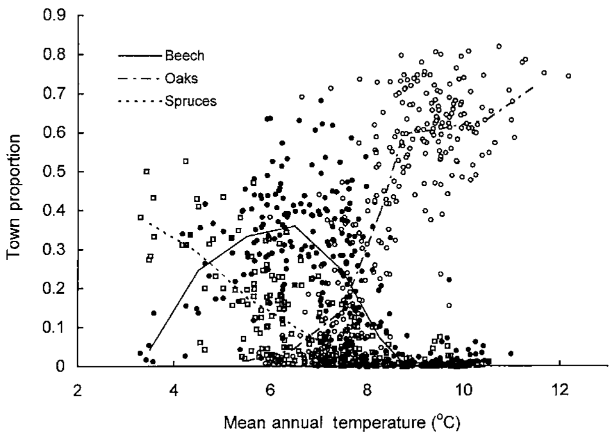


Figure 8 Plot of town-wide (open circle) oak, (solid circle) beech, and (open square) spruce presettlement abundances against mean annual temperature. Lines connect means in 0.5 °C classes plotted at the midpoint.

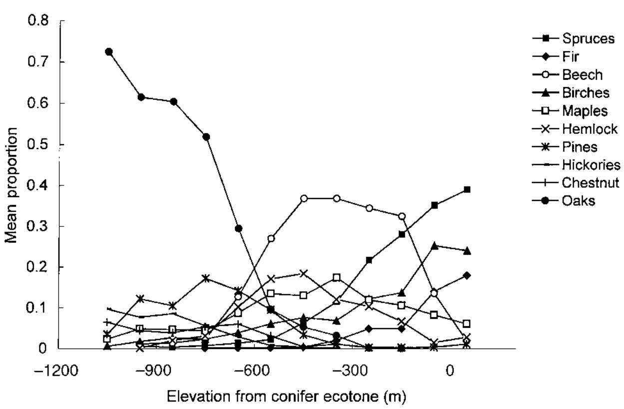


Figure 6 Coenocline plot of town average presettlement composition by classes of FCE-elevation (distance from the conifer ecotone). Mean of town-wide relative frequencies are plotted at the midpoint of 100-m classes.

DISCUSSION

New England tension zone

The geographical, vegetational and environmental patterns all demonstrate a broad regional gradient among towns across New England and reiterate the gross latitudinal trend in the vegetation. Embedded in this trend, however, is a distinct division separating 'northern' and 'southern' vegetation (Figs 2, 4 and 5). This discontinuity reflects the coincident boundary of a suite of taxa abundances (i.e. especially beech, hemlock, oaks, hickories, chestnut, and to a lesser degree spruce and maples). The boundaries are relatively abrupt, and interestingly in several cases, are not actual range limits (Fig. 3). This steepening of the vegetation gradient most simply marks the shift from oak to beech dominance and indicates the presence of a vegetation ecotone. An objective linear approximation of the position of this discontinuity was constructed by following the joint boundary (or splitting the gap between town outlines) of the towns constituting cluster 4 (rarely cluster 2 or 3) with those in cluster 5 or 7 (rarely cluster 6) (Figs 4 and 9). Only seven towns were dislocated by this line (on the wrong side of the ecotone) from their primary division determined by the cluster analysis. Both its location at the southern boundary of the northern hardwood forest and its rapid transition over a moderate environmental gradient are very reminiscent of the 'tension zone' in Wisconsin (Curtis, 1959). Thus we term this sharp boundary the 'New England tension zone'.

The coincidence of these changes, over roughly a town's width, ran across several physiographic sections, bedrock groups and apparently across temperature regime or altitude (Fig. 9; Table 4). For example, the tension zone included former sea coast in central Maine and the southern slopes of the White Mountains and cut diagonally across the Berkshires following neither the calcareous valley nor the Taconic ridges. Although the ecotone was generally at modest elevations (250–350 m a.s.l.) on the uplands, it did not coincide with any break in landforms or environmental variables (Grimm, 1984; Gosz, 1991). The environment of this boundary was calculated in ARCVIEW as the mean values of the seventy-nine sample towns that are within 1 km of the tension zone line (Figs 4 and 9). The tension zone obviously winds across a range of latitude (mean = 42.7°N, SD = 0.63°), but also has variable altitude (mean = 260 m, SD = 102 m), FCE-elevation (mean = −596 m; SD = 108 m), and climate (T_{bar} mean = 7.6 °C, SD = 0.7 °C). This geographically coherent boundary, despite inconsistent topography (60–350 m a.s.l.) and climate (T_{bar} range 5.7–8.6 °C), indicates a moderately low sensitivity to the regional environment. The cause of such a discontinuity in a continuous gradient may be, in part, because of a response to undetermined factors (e.g. soils, glacial substrates, bedrock), historic legacies (e.g. disturbance regimes, species migrations), or biotic interactions. Echoing the pattern in Wisconsin, a very likely cause may have been the influence of fire on the vegetation to the south of the boundary (Curtis, 1959; Grimm, 1984; Parshall & Foster, 2002).

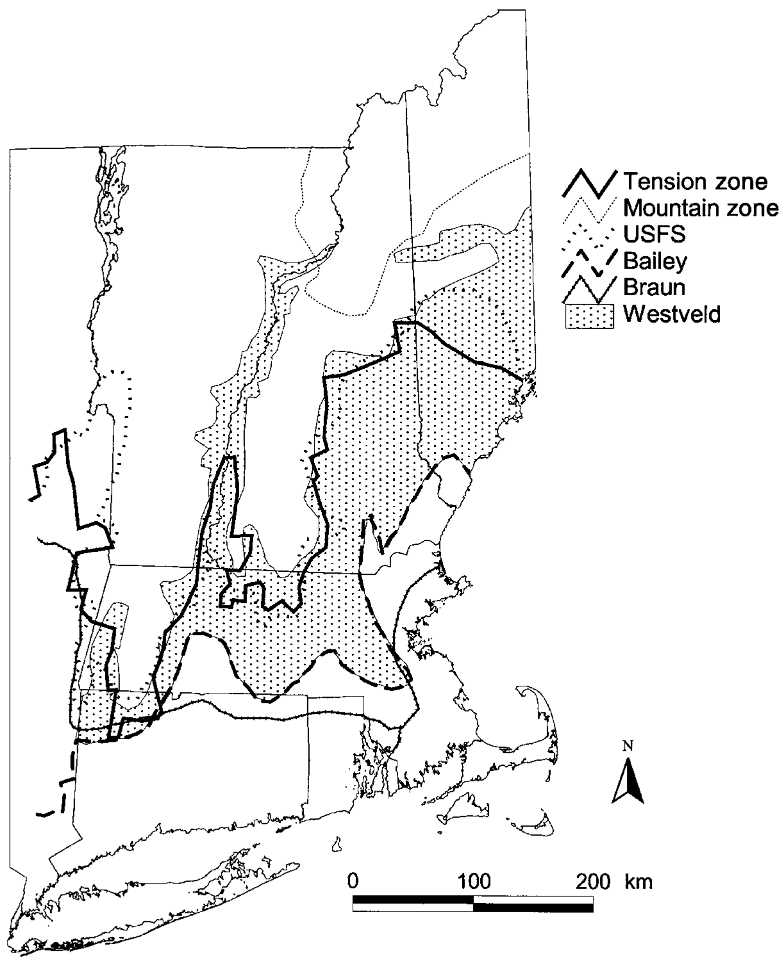


Figure 9 Location of New England Tension Zone, mapped as the town boundaries from presettlement surveys clusters. The shaded area is the extent of Westveld *et al.*'s (1956) Transition-White Pine-Hemlock Zone, which was also adopted by Kuchler (1964, 1978) as a transition zone between Northern Hardwoods and Appalachian Oak Potential Natural Vegetation and by Bailey (1976) as the boundary, in part, between the Laurentian Mixed Forest and Eastern Deciduous Forest Provinces. Also shown is Braun's (1950) boundary between the Hemlock-White Pine-Northern Hardwoods and Oak-Chestnut Regions, and the United States Forest Service's (Keys *et al.*, 1995) boundary between the Lower New England and Vermont/New Hampshire Upland Sections. The dotted line is the mountain zone boundary delimiting the extent of coniferous dominated Cluster 1.

Despite some discrepancy in the valleys, there is an obvious close correspondence between the tension zone and boundaries of previously mapped units of New England forest regions (Fig. 9). Some past studies seem to place a transition further to the south (Raup, 1940; Braun, 1950; Bailey, 1976), but the northern limits of the Westveld *et al.* (1956) transition zone aligns closely with the presettlement division. A recent compilation of ecological units in the United States (Keys *et al.*, 1995) places a modern section boundary in virtually the same location as the division in the presettlement composition, except in the southern Berkshires (Fig. 9). As these recent boundaries are all derived from modern vegetation and environmental surrogates, there is evidently a lasting and close connection between past vegetation patterns and modern, albeit altered vegetation.

Northern Hardwoods

The vegetation in the uplands of northern New England forms a single continuous sequence from mixed spruce to pure hardwoods (Fig. 5). All four northern clusters have a mixture of beech, maple and (undoubtedly yellow) birch. This is the classic 'northern hardwood' forest, with spruce gradually becoming important in the northernmost two clusters (clusters 1, 2) and hemlock increasing in the southernmost (cluster 4). Many factors of climate and topography vary in unison across the region (Table 4), such that it is difficult to separate the effects of any single factor. Significantly, the best environmental discriminator is the composite FCE-elevation index which integrates climatic factors into the complex gradient. There are also two congruent geographical gradients, altitude and latitude, but they apparently form a single vegetational sequence (Figs 6 and 7). Interestingly, the dual gradients combine to produce, on the average, clearly defined elevational 'zones' which dip to the north on the mountains (Cogbill & White, 1991). In addition, a series of distinctive hardwood species (i.e. ash, basswood, elm, butternut) were more abundant in the 'richer' environments on calcareous bedrock, in 'bottomlands' along major rivers, or in the Champlain Valley. These lowlands together with the southern edges of the region, such as north-central Massachusetts (cluster 4) also had an additional minor occurrence of southern elements (i.e. pine, oak).

The major traditional vegetational boundary in the Appalachian Mountains is the coniferous/deciduous ecotone marking the lower elevational boundary of 50% abundance of spruce-fir dominance (Siccama, 1974; Bormann & Likens, 1979; Cogbill & White, 1991). This classic montane 'coniferous' ecotone is mostly missed in the presettlement survey as high altitude sites were seldom settled and many were never surveyed. This shortfall is part of the reason that there are few sample towns in the White Mountains and it causes some 'invisibility' of montane coniferous forests on the vegetation maps (Figs 2–4).

The fourteen mixed spruce towns (cluster 1) at the northeast limits of the region certainly encompassed part of the montane coniferous zone (Fig. 4). The cluster averaged just 98-m elevation below the coniferous ecotone, but had an average maximum altitude 346 m actually above it (Table 4). A single line following the boundary of the towns in Cluster 1 delimits what we term the 'mountain zone', with only two cluster 2 towns dislocated within the zone (Figs 4 and 9). This vegetation boundary was the third division in the cluster analysis (Fig. 4), but based on the overlap of towns in the ordination (Fig. 5) and the similar composition (Table 3), it was apparently not a discontinuity on the continuum. The mountain zone lay within the Spruce-Fir-Northern Hardwood or alternatively the Northern Hardwoods Spruce zones in previous classifications (Westveld *et al.*, 1956; Kuchler, 1964), but it was more restricted, not extending down the Green Mountains or into south-western New Hampshire as in previous maps.

Within-town averaging over variable upland topography tended to dilute the proportion of any high altitude coniferous vegetation in Cluster 2, with only a shadow of the mountain zone. This originally mixed northern hardwoods cluster was warmer ($T_{\text{bar}} = 6.3$ °C) and had much lower altitude (FCE-elevation $= -362$ m) than the coniferous ecotone. Despite the decrease in spruce, the 20% isowit of spruce abundance still approximated the southern boundary of cluster 2 (Fig. 3e). This boundary was also aligned with the southern boundary of the Spruce-Fir-Northern Hardwood forest zone (Westveld *et al.*, 1956). Geographically, cluster 2 filled in the areas of the modern Spruce-Hardwood Zone not occupied by cluster 1, but this combination was a low order division in the clustering (Fig. 4). Thus there are qualitatively recognizable zones in the uplands, but the priority of the relationships and the distinctiveness of any boundaries are fuzzy.

Central Hardwoods

In southern New England the topography and climate were more equitable and the prominent latitudinal/climatic gradient of the north faded. Oaks became pervasive, but did not separate into discrete contiguous geographical units. Although some clusters had geographical centres (e.g. cluster 8 in eastern Connecticut and cluster 6 on Cape Cod), they also included multiple widely scattered towns mixed among other clusters. Earlier maps delimited a distinct northern boundary of oak or 'sprout hardwoods' (i.e. chestnut, hickory, oak) at varying locations across Connecticut (Hawley & Hawes, 1912; Bromley, 1935; Braun, 1950; Westveld *et al.*, 1956). These latitudinal zones were inconsistent with the grouping of clusters that, if anything, had only a weak east–west division in Connecticut. The prevalence of oak mixed with some hickories and chestnut was obviously related to the drier conditions and perhaps disturbance (Curtis, 1959; Grimm, 1984). Precipitation was actually higher, but the temperature regime was markedly warmer than in northern New England (Table 4). More importantly, the moisture needing ('mesic') species, particularly beech (Fig. 3a), were rare except near the water (lakes, rivers or the coast). The less discrete vegetation pattern on a regional scale and the lack of a climatic gradient imply that another factor, perhaps operating at a smaller scale, is influencing the vegetation. The prominence of oaks and other 'sprout' species, the drier soils and higher temperatures, and perhaps the proximity to large indigenous populations, indicate that fire might have been the disturbance factor which was causing the non-zonal, patchy, oak vegetation (Abrams, 1992; Foster *et al*., 2002).

Central Pine

The most distinctive species pattern in central New England was the prominence of pine at the northern edge of oak dominance (Hawley & Hawes, 1912; Bromley, 1935; Jorgensen, 1971). This formed the pine-oak cluster 6 on Cape Cod and scattered towns, especially in the Merrimack Valley [also recognized by Westveld *et al*. (1956) as a unit on Cape Cod]. The more variable mixed oak transition cluster 5 was scattered from the Hudson Valley to coastal Maine. The centre of this transition was on the edge of the Seaboard Lowland where there were widespread sandy outwash soils, perhaps linking these clusters to substrate conditions. The vegetation also included scattered pine (predominantly pitch) plains (cluster 6), which presumably had a high fire regime (Parshall & Foster, 2002). This area, particularly central Massachusetts, had a considerable presettlement mixed pine component and was later the centre of the old-field white pine region (Bromley, 1935; Westveld *et al*., 1956; Foster *et al*., 1998). Interestingly, the southern edge of this oak-pine region had been proposed earlier as a prominent vegetation boundary across New England (Raup, 1940; Braun, 1950; Bailey, 1976). The presettlement surveys clearly showed (Fig. 9) the strongest vegetation boundary was near the northern edge of Westveld *et al*.'s (1956) transition zone, and that this 'transition' was vegetationally closer to oak types than to northern hardwoods.

Temperature regimes

As vegetation is a product of the climate of previous centuries, its association with current climate is indirect and dependent on a temporal equilibrium of climate patterns. Furthermore, as all presettlement surveys were carried out at the end of the 'Little Ice Age' (1450–1850), they are further removed from a modern environmental baseline. Significantly, historic climatic records from New England document a predominantly cool and somewhat wet regime from 1640 to 1820, with only short-term variability (Jones & Bradley, 1992). The coolest decade was the 1810s and the relatively stable conditions of the previous three centuries ended with a dramatic warming starting roughly in 1850 (Baron, 1992). Preliminary comparisons of early nineteenth century temperature regimes with modern averages quantify this significant increase in mean annual temperature: New Haven, CT (+1.2 °C since 1780 s); Portland, ME (+1.2 °C since 1820s); Hanover, NH (+1.7 °C since 1830s); and Amherst, MA (+1.4 °C since 1840s) (Bradley *et al*., 1987; Hamburg & Cogbill, 1988; Baron, 1992). Despite a possible slight decrease in the strength of the sea-to-upland gradient, the historic temperature changes were relatively consistent across the study area. Thus although the average climate regime has varied temporally, the spatial patterns across the region appear to be reasonably robust.

The modern climate record, at least spatially, still represents the environment which framed the geographical distributions in the presettlement surveys. The quantitative values underlying these patterns, however, must be corrected for the temporal change in the climate. The modern temperature regime can be recalibrated by −1.4 °C (average of four sites cited above) to yield an estimate of the mean annual temperature 200 years ago. For example, normalization of the current average temperature of the tension zone (7.6 °C T_{bar}), yielded an approximation of 6.2 °C for this primary boundary in the presettlement regime. Similarly, the mean annual temperature of coniferous cluster 1 (4.3 °C T_{bar}; Table 4) normalized to the eighteenth century yielded a value of 2.9 °C, which is colder (found at higher altitude) than the 3.4 °C found at the modern coniferous ecotone (Cogbill & White, 1991). If the average temperature of the boundary of the mountain zone (5.1 °C T_{bar}; Table 4, Fig. 8) is corrected by the 1.7 °C change at Hanover, it yields the same estimate of 3.4 °C for the historic coniferous boundary. Thus it appears that the contemporary temperature at the historic vegetation boundary is still appropriate today. Remarkably the presettlement vegetation distributions imply that the historic coniferous/deciduous ecotone was in a lower altitudinal position [historic crossover at 150 m below the current FCE-elevation (Fig. 6)]. This is additional evidence for the long-term decline of red spruce in the mixed forests just below the ecotone (Hamburg & Cogbill, 1988). It also indicates that significant climate change has already occurred and presettlement data are a useful quantitative baseline to document these environmental changes in the region.

Very interestingly the vegetationally discrete tension zone is typified less by temperature regime than is the mountain zone. Despite a change in climate, modern vegetation boundaries, such as the old-field pine transition zone, remains tightly bound to the historic position of the tension zone. Thus the two ecotones in New England are fundamentally different; the mountain zone is a division of a continuum which easily responds to climate, while the tension zone is sharp and responds to enduring non-climatic parameters. Apparently the mountain zone is determined by the varying abundance of a single species (spruce) that is more fluid through time than a complex vegetational tension zone involving multiple species and dynamic processes, perhaps involving fire.

Historical baseline

The witness tree sample gives both spatial and temporal perspective to the vegetation of New England that is difficult to get from previous ecological or geographical studies. Historical methods can also elucidate traditional questions. The 1700s are the appropriate baseline for judging the changes in the forest, be it the effects of logging (Williams, 1989), the role of fire (Day, 1953; Cronon, 1983), or the shifting species compositions, such as spruce or chestnut

(Hamburg & Cogbill, 1988; Paillet 2002). Significantly many prominent regional preconceptions are inconsistent with the historic data that show white pine as a minor component of the forest; spruce as prominent in the northern hardwood zone; chestnut as very restricted (< 10% of the forest); and fire as an important disturbance process throughout southern New England. Further expanded is an enigma first noted by Siccama (1963): an amazingly high presettlement proportion of beech in Vermont, a location which now supports much more maple. The presettlement data base clearly indicates this tremendous dominance of beech over all northern New England. In addition, the data base quantitatively documents many intriguing patterns of both abundance and range not seen in the previous broad-scale isowit maps (Whitney, 1994). For example, local details emerge, such as the abundance of spruce in the swamps of south-eastern New England, more than its current distribution suggests (Bromley, 1935), or the large patch of hemlock in the eastern Berkshire Hills of Massachusetts. This sample of towns is dense enough to display fine details and the extent is wide enough to show the patterns at many scales.

CONCLUSIONS

Town-wide resolution of samples and their expansive coverage were critical in documenting intermediate-scale geographical patterns in the region. The towns covered a wide range of sizes; however, all were large enough to encompass a variety of forest types and landforms, but not so large as to span major differences in physiography or climate. The town-wide scale was ideal to represent species composition of the landscape and this also matched a scale appropriate for distinguishing processes such as responses to glacial substrates or climate (Delcourt & Delcourt, 1988). Thus regional factors, such as species abundances across forest types or response to geomorphology, were detected to various degrees. For example, oak and pine vegetation, together with hickory and chestnut taxa extended northward in the large river valleys in New England. Valley influences on the presettlement vegetation, however, were more prominent in mid-river sections as the zones narrowed and the boundaries or range limits became unclear further upriver. For all the advantages of a scale which generalizes by averaging local variable composition, this method is also limited by its scale. Thus some of the valley attenuation could be the result of past species migration patterns or geomorphology in the valley itself, but much is because of the shrinking of the vegetation scale to a point at which the town-wide sample cannot detect restricted or rare elements. The town-wide surveys are ideal for describing vegetation patterns at the regional scale, but they are only an adjunct to other ecological studies of flora, palaeoecology or phytosociology.

An historical–geographical analysis of presettlement witness tree surveys gives an unparalleled picture of New England's forests before European settlement. The unbiased sampling, comprehensive spatial coverage and temporal control produce a sample that is arguably more accurate and detailed than any current description of the vegetation today (e.g. Iverson *et al*., 1999). Analyses of the extensive data base produce maps showing spatial patterns in the vegetation as several scales. Beyond serving as direct quantitative observations of the forest unconfounded by land use history, the town-wide presettlement surveys yield a new perspective on the New England vegetation. The dominant species, oaks, beech, and to a degree spruce, determine differences in the vegetation types. Significantly hemlock, white pine and hickory seemingly are not discriminators between associations currently bearing their names (Hawley & Hawes, 1912; Braun, 1950; Westveld *et al*., 1956). The eight clusters of towns distinguished by vegetation form a clear climatic/latitudinal series, with broad overlap of species abundances. With the exception of the dramatic oak/beech tension zone across the centre of New England uplands, the changes are gradual, the hallmark of a vegetation continuum.

Reconstructing the nature of the original forests is not just an academic exercise in historical ecology, phytogeography or vegetation ecology, but should be applied to future educational, management and conservation activities. Most importantly this historical–geographical approach establishes an empirical baseline of past species and vegetation distributions that can be used to judge both present and future changes induced by human land use or environmental change.

ACKNOWLEDGMENTS

This work is a collaborative project between Harvard Forest and the Hubbard Brook Long-Term Ecological Research Projects. David Foster has been an enthusiastic supporter and was instrumental in expanding the scope of the study to a regional scale. We thank the many town clerks, archivists, librarians, and repository staff, for helping to use the documents in their care. We are especially grateful for the assistance with valuable collections by the staffs of the Vermont Historical Society, the New Hampshire, Connecticut, and Rhode Island State Libraries, and the Massachusetts and New York State Archives. Ben Slater and other researchers from the Harvard Forest also helped collate some of the Massachusetts records. We thank Brian Hall, Jerry Jenkins and Tom Siccama for valuable discussions and analytical advice and Laura Conkey, Lawrence Walker and Rachel Walker Cogbill for their helpful comments on earlier versions of this manuscript. This work was supported by grants from the National Science Foundation (DEB-98-10221 and DEB-99-11975), the A. W. Mellon Foundation, and the Vermont Biodiversity Project.

REFERENCES

Abrams, M.D. (1992) Fire and the development of oak forests. *Bioscience*, **42**, 346–353.

Abrams, M.D. (2001) Eastern white pine versatility in the presettlement forest. *Bioscience*, **51**, 967–979.

Abrams, M.D. & McCay, D.M. (1996) Vegetation–site relationships of witness trees (1780–1856) in the presettlement forests of eastern West Virginia. *Canadian Journal of Forest Research*, **26**, 217–224.

Abrams, M.D. & Ruffner, C.M. (1995) Physiographic analysis of witness-tree distribution (1765–1798) and present forest cover through north central Pennsylvania. *Canadian Journal of Forest Research*, **25**, 659–668.

Bailey, R.G. (1976) *Ecoregions of the United States* (Map). USDA Forest Service, Intermountain Region. Ogden, UT.

Bailey, R.G. (1996) *Ecosystem Geography*. Springer-Verlag, New York.

Baron, W.R. (1992) Historical climate records from the northeastern United States, 1640 to 1900. *Climate since A.D. 1500* (ed. R.S. Bradley and P.D. Jones), pp. 74–91. Routledge, London.

Black, B.A. & Abrams, M.D. (2001) Analysis of temporal variation and species–site relationships of witness tree data in southeastern Pennsylvania. *Canadian Journal of Forest Research*, **31**, 419–429.

Bonnicksen, T.M. (2000) *America's ancient forests*. John Wiley & Sons, New York.

Boose, E.R., Chamberlain, K.E. & Foster, D.R. (2001) Landscape and regional impacts of hurricanes in New England. *Ecological Monographs*, **71**, 27–48.

Bormann, F.H. & Buell, M.F. (1964) Old-age stand of hemlock-northern hardwood forest in central Vermont. *Bulletin of the Torrey Botanical Club*, **91**, 451–465.

Bormann, F.H. & Likens, G.E. (1979) *Pattern and process in a forested ecosystem*. Springer-Verlag, New York.

Bourdo, E.A. Jr (1956) A review of the general land office survey and of its use in quantitative studies of former forests. *Ecology*, **37**, 754–768.

Bradley, R.S., Eischeid, J.K. & Ives, P.T. (1987) *The climate of Amherst, Massachusetts, 1836–1985*. Department of Geology, University of Massachusetts, MA, Contribution no. 50.

Braun, E.L. (1950) *Deciduous forests of eastern North America*. Hafner Press, New York.

Bromley, S.W. (1935) The original forest types of southern New England. *Ecological Monographs*, **5**, 61–89.

Brown, D.G. (1998) Mapping historical forest types in Baraga County, Michigan, USA as fuzzy sets. *Plant Ecology*, **134**, 97–111.

Burgi, M. & Russell, E.W.B. (2000) Evaluating accuracy of two types of early land survey records in the northeastern United States. *Journal of the Torrey Botanical Society*, **127**, 94–98.

Clark, C.E. (1983) *The eastern frontier, the settlement of northern New England, 1610–1763*. University Press of New England, Hanover, NH.

Clements, F.E. (1905) *Research methods in ecology*. University Publishing Co, Lincoln, NE.

Cline, A.C. & Spurr, S.H. (1942) *The virgin upland forest of central New England: a study of old growth stands in the Pisgah Mountain section of southwestern New Hampshire*. Harvard Forest Bulletin, no. 21. Petersham, MA.

Cogbill, C.V. (1996) Black growth and fiddlebutts: the nature of old-growth red spruce. *Eastern old-growth forests, prospects for rediscovery and recovery* (ed. M.B. Davis), pp. 113–125. Island Press, Covelo, CA.

Cogbill, C.V. (2000) Vegetation of the presettlement forests of northern New England and New York. *Rhodora*, **102**, 250–276.

Cogbill, C.V. & White, P.S. (1991) The latitude–elevation relationship for spruce-fir forest and treeline along the Appalachian mountain chain. *Vegetatio*, **94**, 153–175.

Cowell, C.M. (1995) Presettlement Piedmont Forests: patterns of composition and disturbance in central Georgia. *Annals of the Association of American Geographers*, **85**, 65–83.

Cronon, W. (1983) *Changes in the land, Indians, colonists and the ecology of New England*. Hill and Wang, New York.

Curtis, J.T. (1959) *The vegetation of Wisconsin*. University of Wisconsin press, Madison, WI.

Davis, M.B. (1996) *Eastern old-growth forests, prospects for rediscovery and recovery*. Island Press,. Covelo, CA.

Day, G.M. (1953) The Indian as an ecological factor in the northeastern forest. *Ecology*, **34**, 329–346.

Delcourt, H.R. & Delcourt, P.A. (1988) Quaternary landscape ecology: relevant scales in space an time. *Landscape Ecology*, **2**, 23–44.

Delcourt, H.A. & Delcourt, P.A. (1996) Presettlement landscape heterogeneity: evaluating grain of resolution using General Land Office Survey data. *Landscape Ecology*, **11**, 363–381.

Delcourt, H.R. & Delcourt, P.A. (2000) Eastern Deciduous forests. *North American vegetation*, edition 2nd edn (ed. M.G. Barbour and W.D. Billings), pp. 355–395. Cambridge University Press, New York.

Delcourt, H.R., Delcourt, P.A. & Webb, T. III (1983) Dynamic plant ecology: the spectrum of vegetational change in space and time. *Quaternary Science Reviews*, **1**, 153–175.

Edmonds, M. (2001) The pleasures and pitfalls of written records. *The historical ecology handbook, a restorationist's guide to reference ecosystems* (ed. D. Egan, E.A. Howell and C. Meine), pp. 73–99. Island Press, Covelo, CA.

Egan, D., Howell, E.A. & Meine, C. (eds) (2001) *The historical ecology handbook, a restorationist's guide to reference ecosystems*. Island Press, Covelo, CA.

Fenneman, N.M. (1938) *Physiography of eastern United States*. McGraw-Hill, New York.

Foster, D.R., Clayden, S., Orwig, D.A., Hall, B. & Barry, S.L. (2002) Oak, chestnut and fire: climatic and cultural controls of long-term forest dynamics in New England, USA. *Journal of Biogeography*, **29**, 1359–1379.

Foster, D.A., Motzkin, G. & Slater, B. (1998) Land-use history as long-term broad scale disturbance: regional forest dynamics in central New England. *Ecosystems*, **1**, 96–119.

Frank, E.C. & Lee, R. (1966) *Potential solar beam irradiation on slopes*. United States Forest Service, USA, Research Paper RM-18.

Gleason, H.A. & Cronquist, A. (1964) *The natural geography of plants*. Columbia University Press, New York.

Gleason, H.A. & Cronquist, A. (1991) *Manual of vascular plants of northeastern United States and adjacent Canada*, 2nd edn. New York Botanical Garden, Bronx, NY.

Glitzenstein, J.S., Canham, C.D., McDonnell, M.J. & Streng, D.R. (1990) Effects of environment and land-use history on upland forests of the Cary Arboretum, Hudson Valley, New York. *Bulletin of the Torrey Botanical Club*, **117**, 106–122.

Gosz, J.R. (1991) Fundamental ecological characteristics of landscape boundaries. *Ecotones, the role of landscape boundaries in the management and restoration of changing* *environments* (ed. M.M. Holland, P.G. Risser and R.J. Naiman), pp. 8–30. Chapman & Hall, New York.

Grimm, E.C. (1984) Fire and other factors controlling the Big Woods vegetation of Minnesota in the mid-nineteenth century. *Ecological Monographs*, **54**, 291–311.

Hamburg, S.P. (1984) Organic matter and nitrogen accumulation during 70 years of old-field succession in central New Hampshire. PhD Dissertation, Yale University, New Haven, CT.

Hamburg, S.P. & Cogbill, C.V. (1988) Historical decline of red spruce populations and climatic warming. *Nature (London)*, **331**, 428–431.

Harper, R.M. (1918) Changes in the forest area of New England in three centuries. *Journal of Forestry*, **16**, 442–452.

Hawley, R.C. & Hawes, A.F. (1912) *Forestry in New England*. John Wiley & Sons, New York.

Iverson, L.R., Prasad, A.M., Hale, B.J. & Sutherland, E.K. (1999) *Atlas of current and potential future distributions of common trees of the eastern United States*. USDA. Forest Service, Northeastern Research Station. General Technical Report NE-265, Radnor, PA.

Jones, P.D. & Bradley, R.S. (1992) Climatic variations over the last 500 years. *Climate since A.D. 1500* (ed. R.S. Bradley and P.D. Jones), pp. 649–665. Routledge, London.

Jorgensen, N. (1971) *A guide to New England's landscape*. Barre Publishers, Barre, MA.

Keys, J.E. Jr, Carpenter, C., Hooks, S., Koenig, F., McNab, W.H., Russell, W.E. & Smith, M.-L. (1995) *Ecological units of the eastern United States: first approximation*. UDSA Forest Service, Atlanta, GA.

Kuchler, A.W. (1964) *The potential natural vegetation of the conterminous United States*. Special Publication, no. 36. American Geographical Society, New York.

Kuchler, A.W. (1978) Potential natural vegetation (map and essay). *Ecosystems of the United States*. United States Geological Survey, Reston VA.

Legendre, P. & Legendre, L. (1998) *Numerical ecology*, 2nd English edn. Elsevier, Amsterdam.

Leopold, D.J., McComb, W.C. & Muller, R.N. (1998) *Trees of the central hardwood forests of North America*. Timber Press, Portland, OR.

Little, E.L. Jr (1971) *Atlas of the United States trees. Volume 1. Conifers and important hardwoods*. USDA, Forest Service. Misc. Pub. 1146, Washington, DC.

Little, E.L. Jr (1977) *Atlas of the United States trees. Volume 4. Minor eastern hardwoods*. USDA, Forest Service. Misc. Pub. 1342, Washington, DC.

Loeb, R.E. (1987) Pre-European settlement forest composition in east New Jersey and southeastern New York. *American Midland Naturalist*, **118**, 414–423.

Lorimer, C.G. (1977) The presettlement forest and natural disturbance cycle of northeastern Maine. *Ecology*, **58**, 139–148.

MacCleery, D.W. (1992) *American forests: a history of resiliency and recovery*. Forest History Society, Durham, NC.

Manies, K.L. & Mladenoff, D.J. (2000) Testing methods to produce landscape-scale presettlement vegetation maps from the U.S. public land survey records. *Landscape Ecology*, **15**, 741–754.

Manning, W.E. (1973) The northern limits of the distributions of hickories in New England. *Rhodora*, **75**, 33–51.

Marks, P.L. & Gardescu, S. (1992) Vegetation of the central Finger Lakes Region of New York in the 1790s. *New York State Museum Bulletin*, **484**, 1–35.

McCune, B. & Mefford, M.J. (1997) *PC-ORD. Multivariate analysis of ecological data*, Version 3.0. MjM Software Design, Gleneden Beach, OR.

McIntosh, R.P. (1962) The forest cover of the Catskill Mountain region, New York, as indicated by the land survey records. *American Midland Naturalist*, **68**, 409–423.

Melnyk, M.D. (ed.) (1999) *Genealogist's handbook for New England research*, 4th edn. New England Historic Genealogical Society, Boston, MA.

Merriam, C.H. (1898) Laws of temperature control of the geographic distribution of terrestrial animals and plants. *National Geographic Magazine*, **6**, 229–238.

Neilson, R.P. (1991) Climatic constraints and issues of scale controlling regional biomes. *Ecotones, the role of landscape boundaries in the management and restoration of changing environments* (ed. M.M. Holland, P.G. Risser and R.J. Naiman), pp. 31–51. Chapman & Hall, New York.

Nichols, G.E. (1913) The vegetation of Connecticut II. Virgin Forests. *Torreya*, **13**, 199–215.

Nichols, G.E. (1935) The hemlock-white pine-northern hardwood region of eastern North America. *Ecology*, **16**, 403–422.

Ollinger, S.V., Aber, J.D., Federer, C.A., Lovett, G.M. & Ellis, J.M. (1995) *Modeling physical and chemical climate of the northeastern United States for a Geographic information system*. USDA, Forest Service, Northeastern Research Station. General Technical Report NE-191, Radnor, PA.

Pacala, S.W., Canham, C.D. & Silander, J.A. Jr (1993) Forest models by field measurement: I. The design of a northeastern forest simulator. *Canadian Journal of Forest Research*, **23**, 1980–1988.

Paillet, F. (2002) Chestnut: history and ecology of a transformed species. *Journal of Biogeography*, **29**, 1517–1530.

Parshall, T. & Foster, D.R. (2002) Fire on the New England landscape: regional and temporal variation, cultural and environmental controls. *Journal of Biogeography*, **29**, 1305–1317.

Poiani, K.A., Richter, B.D., Anderson, M.G. & Richter, H.E. (2000) Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. *Bioscience*, **50**, 133–146.

Price. E.T. (1995) *Dividing the land*. University of Chicago Press, Chicago, IL.

Raup, H.M. (1940) Old field forest of southern New England. *Journal of the Arnold Arboretum*, **21**, 266–273.

Russell, E.W.B. (1981) Vegetation of northern New Jersey before European settlement. *American Midland Naturalist*, **105**, 1–12.

Russell, E.W.B. (1997) *People and the land through time, linking ecology and history*. Yale University Press, New Haven, CT.

Schwartz, M.W. (1994) Natural distribution and abundance of forest species and communities in northern Florida. *Ecology*, **75**, 687–705.

Seischab, F.K. (1990) Presettlement forests of the Phelps and Gorham Purchase in western New York. *Bulletin of the Torrey Botanical Club*, **117**, 27–38.

Siccama, T.G. (1963) Pre-settlement and present forest cover in Chittenden County, Vermont. MSc Thesis, University of Vermont, Burlington, VT.
Siccama, T.G. (1971) Presettlement and present forest vegetation in northern Vermont with special reference to Chittenden County. *American Midland Naturalist*, **85**, 153–172.
Siccama, T.G. (1974) Vegetation, soil, and climate on the Green Mountains of Vermont. *Ecological Monographs*, **44**, 325–349.
Sokal, R.R. & Rohlf, F.J. (1981) *Biometry*, 2nd edn. W.H. Freeman & Co., San Francisco, CA.
Thompson, E.H. & Sorenson, E.R. (2000) *Wetland, woodland, wildland: a guide to the natural communities of Vermont*. University of. New England Press, Hanover, NH.
Torbert, E. (1935) Evolution of land utilization in Lebanon, New Hampshire. *Geographical Review*, **25**, 209–230.
Trewartha, G.T. (1968) *An introduction to climate*. McGraw-Hill, New York.
Turner, M.G., Gardner, R.H. & O'Neill, R.V. (2001) *Landscape ecology in theory and practice, pattern and process*. Springer-Verlag, New York.
Vankat, J.L. (1979) *The natural vegetation of North America*. J. Wiley & Sons, New York.
Weaver, J.E. & Clements, F.E. (1938) *Plant ecology*. McGraw-Hill Co, New York.
Westveld, M. *et al.* (1956) Natural forest vegetation zones of New England. *Journal of Forestry*, **54**, 332–338.
White, C.A. (1984) *A history of the rectangular survey system*. United States Department of Interior, Bureau of Land Management, GPO, Washington, DC.
White, P.S. & Cogbill, C.V. (1992) Spruce-Fir Forests of Eastern North America. *Ecology and decline of Red Spruce in the Eastern United States* (eds C. Eagar and M.B. Adams), pp. 3–63. Springer Verlag, New York.
Whitney, G.G. (1994) *From Coastal Wilderness to Fruited Plain*. Cambridge University Press, New York.
Whitney, G.G. & Davis, W.C. (1986) From primitive woods to cultivated woodlots: Thoreau and the forest history of Concord, Massachusetts. *Journal of Forest History*, **30**, 70–81.
Whitney, G.G. & DeCant, J. (2001) Government Land Office Survey and other early land surveys, *The historical ecology handbook, a restorationist's guide to reference ecosystems* (eds D. Egan, E.A. Howell and C. Meine), pp. 147–172. Island Press, Covelo, CA.
Whittaker, R.H. (1975) *Communities and ecosystems*, 2nd edn. Macmillan Publishers, New York.
Williams, M. (1989) *Americans and their forests, a historical geography*. Cambridge University Press, New York.
Winer, H. (1955) History of the Great Mountain Forest, Litchfield, County, Connecticut. PhD Dissertation, Yale University, New Haven, CT.
Woodard, F.M. (1936) *The town proprietors of Vermont: the New England town proprietorship in decline*. Columbia University Press, New York.

BIOSKETCHES

Charles V. Cogbill is a freelance ecologist and the historical ecologist on the Hubbard Brook Long-Term Ecological Research Project. He works on all aspects of the history, composition, dynamics, and management of the vegetation of north-eastern North America.

John Burk is a research assistant in historic ecology at the Harvard Forest studying human history and impact on the New England landscape.

Glenn Motzkin is a plant ecologist at Harvard Forest whose work focuses on historical ecology and its application to conservation in New England.

Appendix 1 Names, occurrence, and equivalent synonymy of trees cited in 389 New England presettlement surveys; [ ] indicates possible secondary synonymy; A ? indicates degree of uncertainty in some identifications

Surveyor name	Towns	Stems	Taxa[†] inferred	Genus group this study
Apple	34	40	*Pyrus malus*	Other
Ash*	254	1647	*Fraxinus* sp.	Ash
Ash, Black	149	874	*Fraxinus nigra*	Ash
Ash, Mountain	1	2	*Sorbus americana,* [*S. decora*]	Other
Ash, Red	77	333	*Fraxinus* sp.*?*	Ash
Ash, Water	6	15	*Fraxinus nigra*	Ash
Ash, White	205	1118	*Fraxinus americana*	Ash
Aspen	9	37	*Populus tremuloides & P. grandidentata*	Poplars
Basswood*	182	1085	*Tilia americana*	Basswood
Beech*	293	18,731	*Fagus grandifolia*	Beech
Birch*	330	4312	*Betula* sp.	Birch
Birch, Black	171	1040	*Betula lenta*	Birch
Birch, Red	7	35	*Betula cordifolia,* [*B. alleghaniensis*]	Birch
Birch, Rock	19	78	*Betula* sp.	Birch
Birch, Swamp	4	60	*Betula alleghaniensis?*	Birch
Birch, White	130	577	*Betula papyrifera, B. cordifolia*	Birch
Birch, Yellow	54	315	*Betula alleghaniensis*	Birch
Blue Beech	3	7	*Carpinus caroliniana*	Ironwoods
Boxwood	20	60	*Acer negundo?*	Other
Butternut*	87	272	*Juglans cinerea*	Butternut
Buttonwood*	44	88	*Platanus occidentalis*	Buttonwood
Cedar (Atlantic)*	52	277	*Chamaecyparis thyoides*	Cedar (Atlantic)
Cedar (northern)*	52	230	*Thuja occidentalis*	Cedar (northern)
Cedar, Red	6	7	*Juniperus virginiana*	Other
Cherry*	84	154	*Prunus* sp.	Cherries
Cherry, Black	14	16	*Prunus serotina*	Cherries
Cherry, Red	5	10	*Prunus pensylvanica*	Cherries
Chestnut*	174	6841	*Castanea dentata*	Chestnut
Elm*	228	1281	*Ulmus* sp.	Elms
Elm, Red	7	10	*Ulmus rubra*	Elms
Elm, White	3	6	*Ulmus americana*	Elms
Elm, Witch	10	22	*Ulmus rubra, U. americana*	Elms
Fir*	90	669	*Abies balsamea*	Fir
Hackmetack	35	199	*Picea rubens &* [*Larix larcina*]	Spruces
Hard beam	16	39	*Ostrya virginiana,* [*Carpinus caroliniana*]	Ironwoods
Hardhack	30	171	*Ostrya virginiana,* [*Carpinus caroliniana*]	Ironwoods
Hazel (Witch)	62	354	*Ostrya virginiana?*	Ironwoods
Hemlock*	290	10,281	*Tsuga canadensis*	Hemlock
Hickory	11	50	*Carya* sp.	Hickories
Hornbeam	81	257	*Ostrya virginiana, Carpinus caroliniana*	Ironwoods
Hornpine	6	78	*Pinus* sp.*?*	Pines
Ironwood*	64	384	*Ostrya virginiana*	Ironwoods
Juniper	8	13	*Juniperus* sp.	Other
Leverwood	27	75	*Ostrya virginiana*	Ironwoods
Linden	1	1	*Tilia americana*	Basswood
Linewood	2	3	*Tilia americana?*	Basswood
Maple*	360	9900	*Acer* sp.	Maples
Maple, Hard	62	975	*Acer saccharum*	Maples
Maple, Red	1	1	*Acer rubrum*	Maples
Maple, Rock	57	309	*Acer saccahrum*	Maples
Maple, Soft	58	466	*Acer rubrum,* [*A. saccharinum*]	Maples
Maple, Sugar	5	10	*Acer saccahrum*	Maples
Maple, Swamp	2	3	*Acer rubrum,* [*A. saccharinum*]	Maples
Maple, White	50	190	*Acer saccharinum,* [*A. rubrum*]	Maples
Mulberry	1	2	*Morus rubra*	Other
Oak	235	1961	*Quercus* sp.	Oaks
Oak, Black	264	16,733	*Quercus rubra* & *Q. velutina*	Oaks

Appendix 1 *continued*

Surveyor name	Towns	Stems	Taxa† inferred	Genus group this study
Oak, Chestnut	41	174	*Quercus prinus*	Oaks
Oak, Grey	59	799	*Quercus rubra*?	Oaks
Oak, Mountain	3	7	*Quercus prinus*?	Oaks
Oak, Pin	18	28	*Quercus palustris*	Oaks
Oak, Red	237	7548	*Quercus rubra*	Oaks
Oak, Rock	37	551	*Quercus prinus*?	Oaks
Oak, Shrub	9	15	*Quercus ilicifolia*?	Oaks
Oak, Swamp	46	256	*Quercus bicolor*	Oaks
Oak, White*	271	32,635	*Quercus alba*	Oaks
Oak, White Swamp	18	54	*Quercus bicolor*	Oaks
Oak, Yellow	32	145	*Quercus prinus*	Oaks
Oilnut	5	16	*Juglans cinerea*	Butternut
Peach	2	2	*Prunus* sp.?	Cherries
Pear, Wild	11	23	*Prunus* sp., [*Amelanchier* sp.]	Cherries
Pepperidge*	34	114	*Nyssa sylvatica*	Pepperidge
Pine*	259	9629	*Pinus* sp.	Pines
Pine, Black	3	4	*Pinus* sp.?	Pines
Pine, Candle	4	39	*Pinus* sp.?	Pines
Pine, Norway	16	44	*Pinus resinosa*	Pines
Pine, Pitch	119	2642	*Pinus rigida*	Pines
Pine, Red	2	2	*Pinus resinosa*	Pines
Pine, Spruce	7	13	*Tsuga canadensis*?	Pines
Pine, Swamp	3	3	*Pinus* sp.?	Pines
Pine, White	189	2403	*Pinus strobus*	Pines
Pine, Yellow	15	29	*Pinus rigida*, [*P. resinosa*]	Pines
Plum	9	16	*Prunus* sp., [*Amelanchier* sp.]	Cherries
Poplar*	194	1188	*Populus* sp.	Poplars
Poplar, Water	3	22	*Populus* sp.?	Poplars
Popple	25	86	*Populus tremuloides* & *P. grandidentata*	Poplars
Remmond	4	15	*Ostrya virginiana*?	Ironwoods
Roundwood	4	7	*Sorbus* sp., [*Acer pensylvanicum*]	Other
Shagbark	4	7	*Carya ovata*	Hickories
Spruce*	229	4533	*Picea rubens* [*P. mariana, P. glauca*]	Spruces
Spruce, Black	12	77	*Picea rubens* & [*P. mariana*]	Spruces
Tamarack*	21	58	*Larix laricina*	Tamarack
Walnut*	230	7927	*Carya* sp.?	Hickories
Walnut, Bitter	3	4	*Carya cordiformis*	Hickories
Walnut, Black	2	2	*Juglans nigra*	Other
White Tree	5	6	*Liriodendron tuliperfera*?, [*Tilia americana*] & [*Populus deltoides*?]	Other
Whitewood	46	187	*Liriodendron tuliperfera*?, [*Tilia americana*] & [*Populus deltoides*?]	Other
Wicerpee	10	32	*Tilia americana*?	Basswood
Odd Oaks (*Quercus* sp.): (Name Towns, Trees)	Greene 2, 54; Blue 2, 9; Clapboard 2, 3; Shingle 1, 2; Beach 1, 1; Chasson 1, 1; Live 1, 1; Pinknot 1, 1; Red Rock 1, 1; Ruff 1, 1; Squirrel 1, 1; Swamp Black 1, 1			
Ambiguous species: (Name Towns, Trees)	Swampwood 12, 50; Pegwood 5, 15; Hornwood 4, 8; Beattlewood 3, 18; Bilberry tree 4, 6; Black tree 3, 5; Clapboard 3, 3; Shittum wood 2, 2; Ballwood 2, 2; Jerwood 2, 2; Navewood 2, 2; Feare? 1, 8; Caven 1, 3; Tobaccowood 1, 2; Dogberry 1, 2; Chebalo tree 1, 2; Almond tree 1, 1; Bigwood 1, 1; Greenwood 1, 1; Mastick 1, 1; Nickopwood 1, 1; Pipestaff 1, 1, White Ash Poplar 1, 1; Rauge 1, 1			

*Lead taxa in one of the twenty-two lumped genus categories.
†Nomenclature follows Gleason & Cronquist (1991).

Exhibit D

nature climate change

LETTERS

PUBLISHED ONLINE: 23 OCTOBER 2011 | DOI: 10.1038/NCLIMATE1264

Regional carbon dioxide implications of forest bioenergy production

Tara W. Hudiburg[1]*, Beverly E. Law[1], Christian Wirth[2] and Sebastiaan Luyssaert[3]

Strategies for reducing carbon dioxide emissions include substitution of fossil fuel with bioenergy from forests[1], where carbon emitted is expected to be recaptured in the growth of new biomass to achieve zero net emissions[2], and forest thinning to reduce wildfire emissions[3]. Here, we use forest inventory data to show that fire prevention measures and large-scale bioenergy harvest in US West Coast forests lead to 2–14% (46–405 Tg C) higher emissions compared with current management practices over the next 20 years. We studied 80 forest types in 19 ecoregions, and found that the current carbon sink in 16 of these ecoregions is sufficiently strong that it cannot be matched or exceeded through substitution of fossil fuels by forest bioenergy. If the sink in these ecoregions weakens below its current level by 30–60 g C m^{-2} yr^{-1} owing to insect infestations, increased fire emissions or reduced primary production, management schemes including bioenergy production may succeed in jointly reducing fire risk and carbon emissions. In the remaining three ecoregions, immediate implementation of fire prevention and biofuel policies may yield net emission savings. Hence, forest policy should consider current forest carbon balance, local forest conditions and ecosystem sustainability in establishing how to decrease emissions.

Policies are being developed worldwide to increase bioenergy production as a substitution for fossil fuel to mitigate fossil fuel-derived carbon dioxide emissions, the main cause of anthropogenic global climate change[4,5]. However, the capacity for forest sector bioenergy production to offset carbon dioxide emissions is limited by fossil fuel emissions from this activity (harvest, transport, and manufacturing of wood products) and the lower energy output per unit carbon emitted compared with fossil fuels[6]. Furthermore, forest carbon sequestration can take from decades to centuries to return to pre-harvest levels, depending on the initial conditions and amount of wood removed[7]. The effects of changes in management on CO_2 emissions need to be evaluated against this baseline. Consequently, energy policy implemented without full carbon accounting and an understanding of the underlying processes risks increasing rather than decreasing emissions[4,8].

In North America, there is increasing interest in partially meeting energy demands through large-scale forest thinning[5], with the added benefit of preventing catastrophic wildfire and concurrent carbon loss[3]. Although forest thinning can be economically feasible, sustainable, and an effective strategy for preventing wildfire where risk is high[9,10], it remains unresolved whether this type of forest treatment can satisfy both the aims of preventing wildfire and reducing regional greenhouse gas emissions.

For both aims to be satisfied, it needs to be shown that: (1) reduction in carbon stocks due to thinning and the associated emissions are offset by avoiding fire emissions and substituting fossil fuel emissions with forest bioenergy, (2) the change in management results in less CO_2 emissions than the current or 'baseline' emissions, and (3) short-term emission changes are sustained in the long term. Determination of baseline forest sector carbon emissions can be accomplished by combining forest inventory data and life-cycle assessment (LCA[6]) that includes full carbon accounting of net biome production (NBP) on the land in addition to carbon emissions from bioenergy production and storage in wood products. NBP is the annual net change of land-based forest carbon after accounting for harvest removals and fire emissions.

Our study focused on the US West Coast (Washington, Oregon and California), a diverse region owing to the strong climatic gradient from the coast inland (300–2,500 mm precipitation per year) and a total of 80 associated forest types, ranging from temperate rainforests to semi-arid woodlands (Supplementary Table S1). The region is divided into 19 distinct ecoregions[11] on the basis of climate, soil and species characteristics, and includes a broad range of productivity, age structures, fire regimes and topography. Mean net primary production of the forest types range from 100–900 g C m^{-2} yr^{-1} (this study), falling within the global range of 100 to 1,600 g C m^{-2} yr^{-1} reported for temperate and boreal forests[12]. Forest land ownership is divided fairly evenly between public and private sectors having different management histories and objectives that affect forest carbon dynamics[13].

Carbon sequestration rates vary greatly across the region, with mean net ecosystem production (NEP; photosynthesis minus respiration) ranging from -85 g C m^{-2} yr^{-1} in the dry Northern Basin to more than 400 g C m^{-2} yr^{-1} in the mesic Coast Range. After accounting for fire emissions and substantial harvest removals, regional NBP remains a significant sink of 26 ± 3 Tg C yr^{-1} or 76 ± 9 g C m^2 yr^{-1}, similar to the US average[14] and estimates for the member states of the European Union[15]. Sixteen of the 19 ecoregions, representing 98% of the forest area in the region are estimated to be carbon sinks (Fig. 1a; exceptions are drier ecoregions where annual productivity is low and fire emissions are relatively high). Thus, the observed regional sink is not solely due to the region's highly productive rainforests, which occupy 15% of the area. Within the region, California's NBP is higher than that of Oregon and Washington (107 versus 53–61 g C m^{-2} yr^{-1}), primarily owing to differences in NEP (Supplementary Table S2) and harvest between similar forest types within the same ecoregions that cross state boundaries (Supplementary Discussion and Table S3).

In addition to current management or business as usual (BAU, characterized by current preventive thinning and harvest levels), we designed three treatments (Supplementary Fig. S1a) to reflect the varying objectives of potential forest management systems: forest fire prevention by emphasizing removal of fuel ladders

[1]Department of Forest Ecosystems and Society, 321 Richardson Hall, Oregon State University, Corvallis, Oregon 97331, USA, [2]Department of Systematic Botany and Functional Biodiversity, University of Leipzig, Johannisalle 21-23, 04103 Leipzig, Germany, [3]Laboratoire des Sciences du Climat en de l'Environnement, CEA CNRS UVSQ, Centre d'Etudes Ormes des Merisiers, 91191 Gif Sur Yvette, France. *e-mail: Tara.hudiburg@oregonstate.edu.

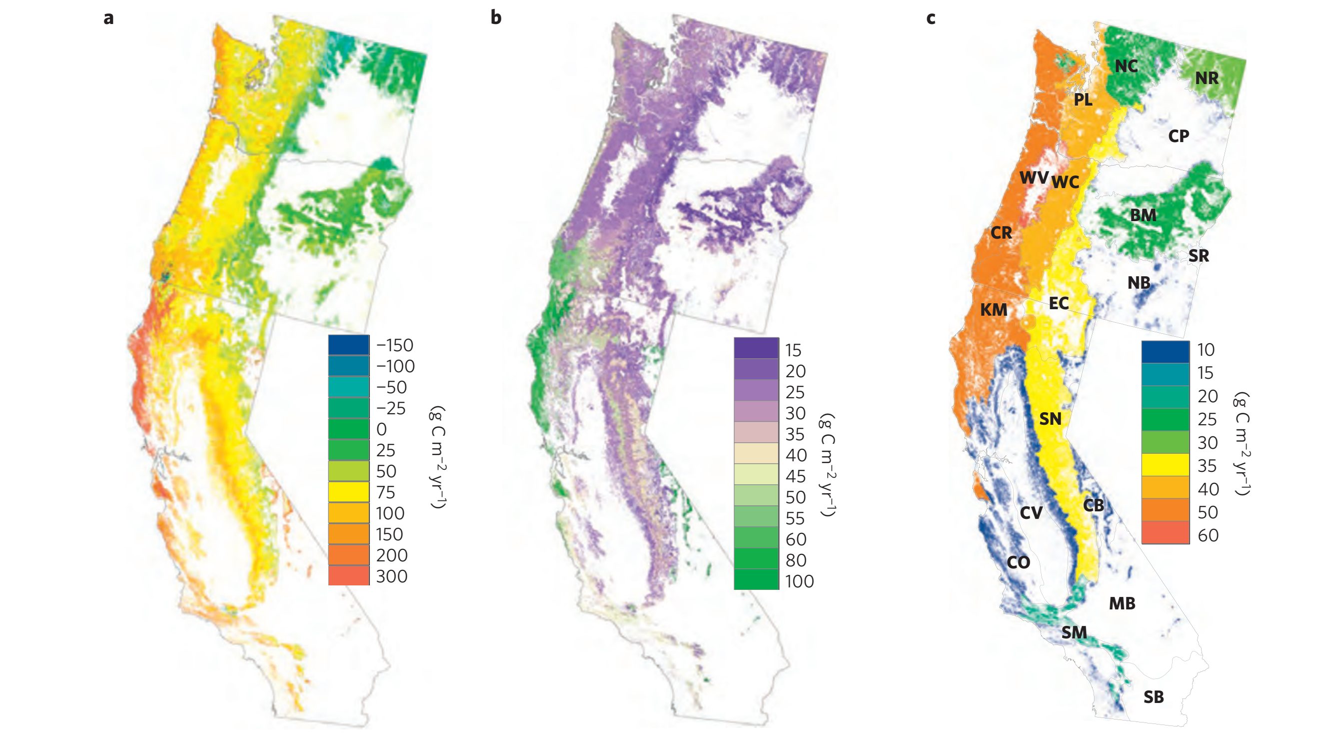


Figure 1 | Maps of US West Coast NBP and uncertainty for current and threshold conditions. a, Current NBP or BAU; positive values (warm colours) indicate forest sinks whereas negative values (cool colours) are carbon sources to the atmosphere. **b**, The current NBP uncertainty estimates that were calculated using Monte Carlo simulations of mean forest type values for the components of NBP (net ecosystem productivity, fire and harvest) combined with the uncertainty associated with remote sensing land cover estimates. **c**, The amount NBP would need to decrease to reach a threshold NBP where bioenergy management may result in emission decreases to the atmosphere. BM, Blue Mountains; CB, Central Basin; CO, California Chaparral and Oak Woodlands; CP, Columbia Plateau; CR, Coast Range; CV, Central California Valley; EC, East Cascades; KM, Klamath Mountains; MB, Mohave Basin; NB, North Basin and Range; NC, North Cascades; NR, Northern Rockies; PL, Puget Lowlands; SB, Sonoran Basin; SM, Southern California Mountains; SN, Sierra Nevada; SR, Snake River; WC, West Cascades; WV, Willamette Valley.

('fire prevention') in fire-prone areas, making fuel ladder removal economically feasible by emphasizing removal of additional marketable wood in fire-prone areas ('economically feasible'), or thinning all forestland regardless of fire risk to support energy production while contributing to fire prevention ('bioenergy production'). Removals are in addition to current harvest levels and are performed over a 20-year period such that 5% of the landscape is treated each year. Our reliance on a data-driven approach versus model simulations strengthens our analysis in the short term, but limits our ability to make long-term predictions. Extending our study beyond a 20-year timeframe would overstretch data use because current forest growth is unlikely to represent future growth due to changes in climate, climate-related disturbance, and land use[16,17].

In our study region, we found that thinning reduced NBP under all three treatment scenarios for 13 of the 19 ecoregions, representing 90% of the region's forest area. The exceptions where NBP was not reduced were primarily due to high initial fire emissions compared to NEP (for example, Northern Basin and North Cascades; Supplementary Fig. S2). The dominant trend at the ecoregion level was mirrored at the regional level, with the bioenergy production scenario (highest thinning level) resulting in the region becoming a net carbon source (Supplementary Table S2 and discussion of state-level estimates). Regionally, forest biomass removals exceeded the potential losses from forest fires, reducing the *in situ* forest carbon sink even after accounting for regrowth, as found in previous studies with different approaches or areas of inference[8,18]. Because we have assumed high reductions in fire emissions for the areas treated in each scenario, it is unlikely we are underestimating the benefit of preventive thinning on NBP.

It is important to recognize that even if the land-based flux is positive (a source) or zero (carbon neutral), decreases in NBP from BAU can increase CO_2 emissions to the atmosphere. LCA was used to estimate the net emissions of carbon to the atmosphere in each treatment scenario (Supplementary Fig. S1b and Tables S4 and S5). LCA at the ecoregion level revealed that emissions are increased for 10 out of 19 of the ecoregions (Fig. 2), representing 80% of the forest area in the region. The combination of *in situ* and wood-use carbon sinks and sources emit an additional 46, 181 and 405 Tg C to the atmosphere over a 20-year period (2–14% increase) above that of the BAU forest management scenarios for the fire prevention, economically feasible, and bioenergy production treatments, respectively (Fig. 3).

Sensitivity analysis of our results to a range of fire emission reductions, energy conversion efficiencies, wood product decomposition rates and inclusion of wood substitution showed that carbon emissions varied by −10 to 28% from the optimum values across the scenarios, depending on the combination of assumptions (Supplementary Discussion and Table S6). The analysis revealed that an increase in estimated current fire emissions (which effectively reduces the baseline sink) may decrease total atmospheric C emissions in the fire prevention scenario, but only given optimum conditions for all of the other parameters (for example 100% energy efficiency). Nevertheless, if fire frequency and intensity increase in the future[19], emissions savings through forest bioenergy production may become possible, especially in ecoregions where the sink is already weak.

Previous case studies showed that harvesting an old-growth forest in the Pacific Northwest[20] or increasing the thinning removals of temperate forests is likely to deteriorate the forest and wood

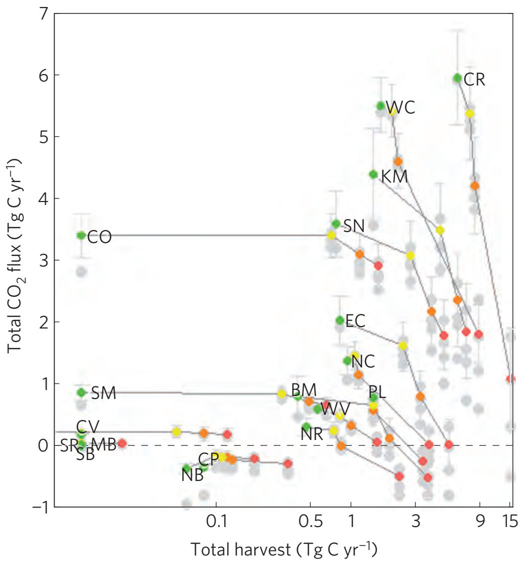


Figure 2 | Life-cycle assessment carbon emission trends by ecoregion under various management scenarios. The *x* axis is the total harvest (BAU + treatment) and the *y* axis is the total CO_2 flux in Tg C yr^{-1} for each ecoregion. Coloured circles represent each scenario (green, BAU; yellow, fire prevention; orange, economically feasible; red, bioenergy production). Grey circles are the values for each sensitivity analysis set of parameters and the error bars represent the estimate uncertainty. The locations of the ecoregions indicated by labels are shown in Fig. 1a. For most ecoregions, the treatments increase emissions to the atmosphere.

product carbon stock[21]. However, these studies were limited to a handful of sites, relied primarily on modelled results[3,18] and did not account for the energy requirements of forest management and wood processing nor for the potential to substitute fossil fuels with bioenergy. We build on these results by including all ecoregions, all age classes (not just old-growth), three treatments including bioenergy production, and sector-based LCA. We found that even though forest sector emissions are compensated for by emission savings from bioenergy use, fewer forest fires, and wood product substitution, the end result is an increase in regional CO_2 emissions compared to BAU as long as the regional sink persists.

To determine a threshold NBP for which bioenergy management reduces atmospheric CO_2 emissions compared with BAU, we applied the same assumptions as used in the LCA. We found that if the NBP drops by 50–60 g C m^{-2} yr^{-1} in currently productive ecoregions or 15–30 g C m^{-2} yr^{-1} in currently less productive ecoregions, bioenergy management would come with CO_2 emissions savings compared to BAU (Fig 1c). Aggregating the ecoregion thresholds translates into a regional mean NBP of 45 g C m^{-2} yr^{-1} or a 41% reduction on average. Reductions in NBP may occur due to increased mortality and/or decreased growth due to climate, fire, or insect outbreaks. However, reductions in NBP from increased harvest do not qualify because harvest increases emissions; wood carbon enters the products/bioenergy chain, where subsequent losses occur. We cannot predict from the data when the threshold NBP would occur because a high resolution process-based model with the ability to incorporate future climate, nitrogen deposition, age dynamics, disturbance and management would need to be used, which is beyond the scope of this study.

Ecoregion threshold NBP is dependent on the scenario treatment removals and area because the fire prevention treatment targets only those areas most likely to burn. For example, to reduce emissions in the Sierra Nevada, baseline NBP would have to decrease by as much as 84 g C m^{-2} yr^{-1} for the bioenergy production scenario versus only 13 g C m^{-2} yr^{-1} for the fire prevention scenario. In ecoregions where current sinks are marginal or weakened by climate, fire, or insect outbreaks there may be a combination of harvest intensity and bioenergy production that reduces forest sector emissions. In nine of the ecoregions where forests are carbon neutral or a source of CO_2 to the atmosphere and/or fire emissions are high for BAU, total CO_2 emissions under the fire prevention scenario could be reduced compared with BAU. They provide examples where management strategies for carbon emission reduction or sequestration should differ from the majority of the region; a one-size-fits-all approach will not work[22]. Also, large areas in the Northern Rockies (for example, Colorado and Wyoming) are at present experiencing increases in forest mortality due to beetle-kill, a trend which could continue in a warmer climate[23]. These areas may already be at or below the threshold NBP; if so, they could benefit from targeted bioenergy implementation. However, simply lowering current regional harvest intensities in areas where NBP is not weakened also reduces emissions (Supplemental Discussion and Fig. S3). Finally, as we have assumed large-scale implementation of these strategies in addition to BAU harvest, we may be overestimating future harvest even though harvest has declined significantly since 1990 because of restrictions placed on harvest on federal lands as part of the Northwest Forest Plan. If the strategies were used to substitute for BAU harvest, the outcome on NBP would be much different (that is, increased for the fire prevention scenario).

Our study is one of the first to provide full carbon accounting, including all of the sinks and sources of carbon emissions from the

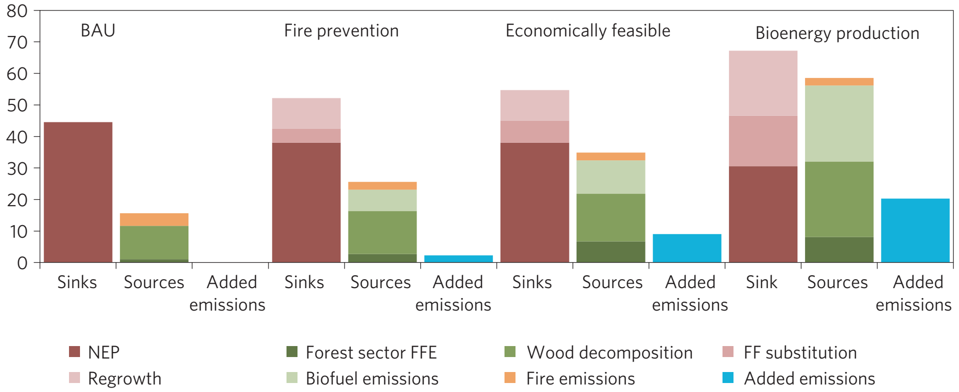


Figure 3 | Total US West Coast forest sector carbon sinks, sources and added emissions relative to BAU under various management scenarios. Units are in Tg C yr^{-1}. Life-cycle assessment estimates account for changes in carbon on land in addition to emissions associated with production, transport and usage of wood, and substitution and displacement of fossil fuel emissions associated with use and extraction. BAU results in the lowest anthropogenic emissions from the forest sector.

forestry sector and the current *in situ* sink, for such a large area. Given the diversity of woody ecosystems in the study region, ranging from highly productive temperate rainforests to less productive semi-arid woodlands, the trends in response probably apply to other temperate regions globally (Supplementary Table S1) where forests are at present a strong net carbon-sink (for example, Eastern US, China and Europe), although the extent of the effect remains to be established.

Greenhouse-gas reduction plans call for up to 10% reductions in emissions by 2020 and forest-derived fuels are being proposed as a carbon-neutral solution to reducing energy emissions. In all of our proposed scenarios, increases in harvest volume on the US West Coast will on average result in regional emission increases above current levels, although there are a few ecoregions where the tested scenarios could result in emission savings. As long as the current *in situ* NBP persists, increasing harvest volumes in support of bioenergy production is counterproductive for reducing CO_2 emissions. In this study region, the current *in situ* NBP in tree biomass, woody detritus and soil carbon is more beneficial in contributing to reduction of anthropogenic carbon dioxide emissions than increasing harvest to substitute fossil fuels with bioenergy from forests.

Although large uncertainty remains for regional forecasts to year 2050 or 2100, it is expected that forest carbon sinks will diminish over time because of ageing of the forests, saturation of the CO_2-fertilization and N-deposition effects, and increased mortality due to climate or insects[24,25]. This would require new assessments to identify management options appropriate for each situation. Carbon-management is not the sole criteria that should be considered when planning forest management. Our findings should thus also be evaluated against other ecosystem services, such as habitat, genetic and species diversity, watershed protection, and natural adaptation to climate change.

Methods

We quantified forest sequestration rates and test forest thinning scenarios across the region using a data-intensive approach which, for the first time, takes into account the diversity of forest characteristics and management. We combined Landsat remote sensing data with inventories and ancillary data to map current forest NEP, NBP, and changes in NBP with three thinning scenarios. The approach can be applied at multiple scales of analysis in other regions.

We combined spatially representative observational data from more than 6,000 federal Forest and Inventory Analysis plots (see Supplementary Methods and Table S7) with remote-sensing products on forest type, age and fire risk[26], a global data compilation of wood decomposition data and 200 supplementary plots[13] to provide new estimates of US West Coast (∼34 million hectares) forest biomass carbon stocks (Supplementary Table S8), NEP (the balance of photosynthesis and respiration) and NBP (the *in situ* net forest carbon-sink accounting for removals). We included all forestland in our analysis, across all age classes (20–800 years old) and management regimes. Plot values were aggregated by climatic region (ecoregion), age class and forest type, and this look-up table was used to assign a value to each associated 30 m pixel.

We use regional combustion coefficients to determine fire emissions. Only 3–8% of live tree biomass is actually combusted and emitted in high severity fire in the Pacific Northwest[28], contrary to other studies that report much higher emissions because they assume 30% of all aboveground woody biomass is consumed[27]. Although the latter contradicts extensive field observations[28,29] and modelling studies[30] in the region, we included 30% as the upper-end combustion factor in our sensitivity analysis (Supplementary Table S9).

In addition to the spatially explicit estimates of stocks and fluxes under current management or BAU (current forest harvest), three treatments were designed (fire prevention, economically feasible and bioenergy production; Supplementary Fig. S1a) to reflect the varying objectives of potential future forest management over the next 20 years; within the proposed time period for CO_2 reductions in the US. Areas were prioritized for treatment by fire risk and frequency. The proposed treatments result in additional harvest removals because we assume the current harvest rate for wood products will continue in the future. We limit our specific analysis to the short term because this is the timeframe suitable for policymakers, effectiveness of fire protection treatments, and an appropriate use of the data-driven approach. However, to investigate conditions (for example, sink saturation) that could invalidate our short-term results in the long term, we also calculated the *in situ* NBP at which the atmosphere may benefit from bioenergy removals.

Last, we studied the net effects of the thinning treatments on atmospheric CO_2 by LCA of carbon sources and sinks that includes the post-thinning NBP and wood use (harvest, transport, manufacturing, decomposition, wood product substitution, conversion and use of bioenergy, and displacement of fossil fuel extraction emissions; Supplementary Fig. S1b and Table S4,S5).

Received 19 July 2011; accepted 30 September 2011; published online 23 October 2011

References

1. Buyx, A. & Tait, J. Ethical framework for biofuels. *Science* **332,** 540–541 (2011).
2. Gustavsson, L., Börjesson, P., Johansson, B. & Svenningsson, P. Reducing CO_2 emissions by substituting biomass for fossil fuels. *Energy* **20,** 1097–1113 (1995).
3. Hurteau, M. D. & North, M. Carbon recovery rates following different wildfire risk mitigation treatments. *Forest Ecol. Manag.* **260,** 930–937 (2010).
4. Fargione, J., Hill, J., Tilman, D., Polasky, S. & Hawthorne, P. Land clearing and the biofuel carbon debt. *Science* **319,** 1235–1238 (2008).
5. Richter, D. Jr *et al.* Resource policy: Wood energy in America. *Science* **323,** 1432–1433 (2009).
6. Law, B. E. & Harmon, M. E. Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. *Carbon Manag.* **2,** 73–84 (2011).
7. Harmon, M. E. & Marks, B. Effects of silvicultural practices on carbon stores in Douglas-fir—western hemlock forests in the Pacific Northwest, USA: Results from a simulation model. *Can. J. Forest Res./Rev. Can. Rech. Forest* **32,** 863–877 (2002).
8. Searchinger, T. D. *et al.* Fixing a critical climate accounting error. *Science* **326,** 527–528 (2009).
9. Evans, A. & Finkral, A. From renewable energy to fire risk reduction: A synthesis of biomass harvesting and utilization case studies in US forests. *GCB Bioenergy* **1,** 211–219 (2009).
10. Huggett, R. J. Jr, Abt, K. L. & Shepperd, W. Efficacy of mechanical fuel treatments for reducing wildfire hazard. *Forest Policy Econ.* **10,** 408–414 (2008).
11. Omernik, J. M. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). *Ann. Assoc. Am. Geogr.* **77,** 118–125 (1987).
12. Luyssaert, S. *et al.* Old-growth forests as global carbon sinks. *Nature* **455,** 213–215 (2008).
13. Hudiburg, T. *et al.* Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage. *Ecol. Appl.* **19,** 163–180 (2009).
14. Birdsey, R. A. *et al.* in *North American Forests in the First State of the Carbon Cycle Report (SOCCR): The North American Carbon Budget and Implications for the Global Carbon Cycle* (eds King, A. W. *et al.*) (US Climate Change Science Program and the Subcommittee on Global Change Research, 2007).
15. Luyssaert, S. *et al.* The European carbon balance: part 3: Forests. *Glob. Change Biol.* **16,** 1429–1450 (2009).
16. Battles, J. *et al.* Climate change impacts on forest growth and tree mortality: A data-driven modeling study in the mixed-conifer forest of the Sierra Nevada, California. *Climatic Change* **87,** 193–213 (2008).
17. Ryan, M. G. Temperature and tree growth. *Tree Physiol.* **30,** 667–668 (2010).
18. Mitchell, S. R., Harmon, M. E. & O'Connel, K. E. B. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. *Ecol. Appl.* **19,** 643–655 (2009).
19. Rogers, B. *et al.* Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. *J. Geophys. Res.* **116,** G03037 (2011).
20. Harmon, M. E., Ferrell, W. K. & Franklin, J. F. Effects on carbon storage of conversion of old-growth forests to young forests. *Science* **247,** 699–702 (1990).
21. Nunery, J. S. & Keeton, W. S. Forest carbon storage in the northeastern United States: Net effects of harvesting frequency, post-harvest retention, and wood products. *Forest Ecol. Manag.* **259,** 1363–1375 (2010).
22. Marland, G. & Schlamadinger, B. Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. *Biomass Bioenergy* **13,** 389–397 (1997).
23. Evangelista, P. H., Kumar, S., Stohlgren, T. J. & Young, N. E. Assessing forest vulnerability and the potential distribution of pine beetles under current and future climate scenarios in the Interior West of the US. *Forest Ecol. Manag.* **262,** 307–316 (2011).
24. van Mantgem, P. J. *et al.* Widespread increase of tree mortality rates in the Western United States. *Science* **323,** 521–524 (2009).
25. Stinson, G. *et al.* An inventory-based analysis of Canada's managed forest carbon dynamics, 1990 to 2008. *Glob. Change Biol.* **17,** 2227–2244 (2011).
26. LANDFIRE Data Distribution Site, (US Department of Interior, Geological Survey, 2009); available at http://gisdata.usgs.net/website/landfire/.
27. Wiedinmyer, C. & Hurteau, M. D. Prescribed fire as a means of reducing forest carbon emissions in the western United States. *Environ. Sci. Technol.* **44,** 1926–1932 (2010).

28. Campbell, J., Donato, D., Azuma, D. & Law, B. Pyrogenic carbon emission from a large wildfire in Oregon, United States. *J. Geophys. Res.* **112,** G04014 (2007).
29. Meigs, G., Donato, D., Campbell, J., Martin, J. & Law, B. Forest fire impacts on carbon uptake, storage, and emission: The role of burn severity in the Eastern Cascades, Oregon. *Ecosystems* **12,** 1246–1267 (2009).
30. Ottmar, R. D., Pritchard, S. J., Vihnanek, R. E. & Sandberg, D. V. *14 Final Report JFSP Project 98-1-9-06* (Pacific Northwest Research Station, Pacific Wildland Fire Sciences Laboratory, 2006).

Acknowledgements
This research was supported by the Office of Science (BER), US Department of Energy (DOE, Grant no. DE-FG02-07ER64361), for the North American Carbon Program study, 'Integrating Remote Sensing, Field Observations, and Models to Understand Disturbance and Climate Effects on the Carbon Balance of the West Coast US'. We further thank M. Harmon for discussions of wood product life-cycle assessment. T.W.H. is funded by a DOE global change education program PhD fellowship (GREF). S.L. is funded by ERC Starting Grant 242564.

Author contributions
T.W.H. designed and implemented the study with guidance from B.E.L. and S.L. T.W.H., S.L. and B.E.L. co-wrote the paper and S.L. contributed to parts of the analysis. C.W. provided essential data and methods for the analysis and valuable comments on the manuscript.

Additional information
The authors declare no competing financial interests. Supplementary information accompanies this paper on www.nature.com/natureclimatechange. Reprints and permissions information is available online at http://www.nature.com/reprints. Correspondence and requests for materials should be addressed to T.W.H.

Exhibit E

Agriculture, Ecosystems and Environment 200 (2015) 33–41

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Review

Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis

Christopher Poeplau [a,b,*], Axel Don [a]

[a] Thuenen Institute of Climate-Smart Agriculture, Bundesallee 50, 38116 Braunschweig, Germany
[b] Swedish University of Agricultural Sciences (SLU), Department of Ecology, Box 7044, 75007 Uppsala, Sweden

ARTICLE INFO

Article history:
Received 22 January 2014
Received in revised form 17 October 2014
Accepted 29 October 2014
Available online 14 November 2014

Keywords:
Soil carbon sequestration
Agricultural management
Cover crops
Catch crops
Green manure
Land use
Carbon response function

ABSTRACT

A promising option to sequester carbon in agricultural soils is the inclusion of cover crops in cropping systems. The advantage of cover crops as compared to other management practices that increase soil organic carbon (SOC) is that they neither cause a decline in yields, like extensification, nor carbon losses in other systems, like organic manure applications may do. However, the effect of cover crop green manuring on SOC stocks is widely overlooked. We therefore conducted a meta-analysis to derive a carbon response function describing SOC stock changes as a function of time. Data from 139 plots at 37 different sites were compiled. In total, the cover crop treatments had a significantly higher SOC stock than the reference croplands. The time since introduction of cover crops in crop rotations was linearly correlated with SOC stock change ($R^2 = 0.19$) with an annual change rate of 0.32 ± 0.08 Mg ha^{-1} yr^{-1} in a mean soil depth of 22 cm and during the observed period of up to 54 years. Elevation above sea level of the plot and sampling depth could be used as explanatory variables to improve the model fit. Assuming that the observed linear SOC accumulation would not proceed indefinitely, we modeled the average SOC stock change with the carbon turnover model RothC. The predicted new steady state was reached after 155 years of cover crop cultivation with a total mean SOC stock accumulation of 16.7 ± 1.5 Mg ha^{-1} for a soil depth of 22 cm. Thus, the C input driven SOC sequestration with the introduction of cover crops proved to be highly efficient. We estimated a potential global SOC sequestration of 0.12 ± 0.03 Pg C yr^{-1}, which would compensate for 8% of the direct annual greenhouse gas emissions from agriculture. However, altered N_2O emissions and albedo due to cover crop cultivation have not been taken into account here. Data on those processes, which are most likely species-specific, would be needed for reliable greenhouse gas budgets.

Contents

* Corresponding author at: Swedish University of Agricultural Sciences (SLU), Department of Ecology, Box 7044, 75007 Uppsala, Sweden. Tel.: +49 1782078951.
E-mail address: Christopher.poeplau@ti.bund.de (C. Poeplau).

http://dx.doi.org/10.1016/j.agee.2014.10.024
0167-8809/

1. Introduction

Cropland soils are depleted in soil organic carbon (SOC) as compared to soils under natural vegetation. Cultivation leads to SOC losses of 30–40% compared to natural or semi-natural vegetation (Don et al., 2011; Poeplau et al., 2011). In a regional study, Wiesmeier et al. (2013) calculated a mean carbon saturation for Bavarian cropland soils of 53%. Historically, between 32.5 and 35.7 million km^2 of natural vegetation, encompassing forests, woodlands, savannas, grasslands and steppes have been converted to croplands (DeFries et al., 1999). As a consequence, cropland soils are a huge potential global carbon sink. The world wide necessity of food production is growing due to an increasing world population and increasing wealth of emerging economies. The potential area of cultivated land, which can be reconverted to natural vegetation or grassland is thus limited. In the southern hemisphere, the proportion of agricultural land is still growing (McGuire et al., 2001). It is thus crucial to find effective measures to increase SOC stocks while simultaneously enhancing and maintaining high agricultural productivity.

Changes in SOC stocks are a result of the imbalance between carbon inputs, mainly in the form of dead plant material or manure and outputs, mainly caused by decomposition, leaching and erosion. An often recommended approach for decreasing SOC decomposition in agricultural soils is to reduce or desist from soil tillage to maintain the natural aggregation of soils to protect SOC from microbial consumption. However, it has been shown that the benefit of conservation or no till agriculture was often an artifact of shallow sampling and is smaller than previously thought (Luo et al., 2010). The main management option to increase SOC storage should therefore be to increase carbon inputs. Commonly suggested approaches for increasing carbon inputs are organic manure or sewage sludge amendments, incorporation of straw and extensification through arable–ley rotations (Smith et al., 1997), and lately also the cultivation of winter cover crops (Mazzoncini et al., 2011). Cover crops, also named inter-crops or catch crops, are crops that replace bare fallow during winter period and are ploughed under as green manure before sowing of the next main crop. In contrast to the introduction of arable–ley rotations or periodic green fallows (extensification), cover crops do not necessarily reduce the amount of agricultural products that can be harvested. Furthermore their cultivation does not exclude the possibility of organic manure applications. They just replace the annual fallow period, which is beneficial for the soil in many aspects (Dabney et al., 2001). Besides an increased carbon input, cover crops have been shown to increase biodiversity (Lal, 2004) as well as to reduce soil erosion and drought stress for the following crop when used as mulch cover in water limited systems (Frye et al., 1988). Cultivated in autumn and winter, cover crops are able to take up excess N from the soil and reduce N leaching (Blombäck et al., 2003). To date, cover crops have been in the scientific focus mainly for their capacity to improve soil quality and thereby to foster crop production. The potential of cover crops to increase SOC stocks and thus to mitigate climate change has been highlighted in very few studies (Lal, 2004). Comprehensive quantitative evidence about the mid- to long-term effect of cover crops on SOC storage is lacking. Furthermore, the control of environmental parameters such as soil properties and climatic conditions on SOC stock changes after the introduction of cover crops cannot be evaluated in isolated case studies. After all, most existing case studies evaluated a change in carbon concentration, while soil bulk density, which is needed to calculate SOC stocks, was not measured. For these reasons, a comprehensive meta-analysis to quantitatively evaluate the effect of cover crops on SOC stocks is needed.

West et al. (2004) introduced the concept of carbon management functions as simple regression models to describe and predict the temporal dynamics of soil organic carbon after management changes. In a meta-analysis approach, available field observations were compiled to derive such functions. The advantage of simple empirical models as compared to process-based models was seen in the more practical, functional and transparent way to apply them, especially with regard to the application in carbon accounting frameworks. Based on this concept, Poeplau et al. (2011) developed carbon response functions (CRF) for the major land-use changes in the temperate zone. They improved the existing concept by adding further environmental parameters as explanatory variables to the temporal function.

The aim of this meta-analysis was to quantify SOC stock changes as a consequence of winter cover crop cultivation and to derive a specific carbon response function for SOC accumulation due to cover crops. This function would be useful for carbon accounting and upscaling. Furthermore we aimed to estimate the global potential of cover crop cultivation to sequester SOC.

2. Materials and methods

2.1. Dataset

We compiled existing available studies on the effect of cover crops on SOC using the following strict quality criteria for a study to be included in the analysis: (1) a reference cropland with winter fallow had to be present. The reference had to be directly adjacent to the winter cover crop field and its SOC stock should have been approximately in equilibrium. Therefore we excluded studies, where the cropland was used as grassland or for ley production before the experiment started to avoid legacy effects, such as a strong SOC stock depletion during the experiment. Ideally, a part of a long term conventional cropland was planted with winter cover crops, while a reference part of this cropland remained fallow in winter while the main crop or crop rotation remained the same for all treatments. (2) The winter cover crop was not harvested, but was used as green manure or mulch with the whole plant remaining on the field as additional carbon input. (3) The treatment was not a combination of carbon inputs such as cover crop plus farmyard manure plus straw residues, but cover crops were the only additional carbon input. (4) The treatment duration was reported.

A total of 30 studies comprising 37 different sites were included in the dataset for the analysis (Table 1). Twenty seven of these studies were peer-reviewed, two were Ph.D. theses and one was a scientific report with a comparable quality. A total of 24% of all studies (27% of all sites) were conducted in the tropics, 76% of all studies (73% of all sites) in the temperate zone. The majority of the studies used a randomized block design (90%) to investigate different treatments at once and to minimize initial differences. The treatments differed in fertilization, tillage, cover crop type or a combination of these factors. A total of 27 different species were grown as cover crops with legumes ($n = 61$) and non legumes ($n = 66$) being equally distributed over the dataset. The total number of sampled plots (only cover crop plots) matching the mentioned quality criteria was 139. Sampling depth ranged from

Table 1
List of the included sites with country, elevation, mean annual precipitation (MAP), mean annual temperature (MAT), soil type according to the used soil taxonomy, sampling depth, time since introduction of cover crops (treatment time) and Refs.

Country	Elevation m a.s.l.	MAP mm	MAT °C	Soil type WRB/US soil tax.	Sampling depth cm	Treatment time years	Refs.
Brazil	96	1440	19.4	Acrisol	30	9	Bayer et al., 2000
Brazil	770	1500	22.5	Oxisol	30	4	Metay et al., 2007
Brazil	96	1769	19.7	Paleudalf	20	10	Amado et al., 2006
Brazil	96	1440	19.4	Paleudalf	20	15	Amado et al., 2006
Canada	583	429.3	0.6	Regosol	15	30	Campbell et al., 1991
Canada	817	350	3.3	Orthic brown chernozem	15	9	Curtin et al., 2000
Canada	2	1167	10	Rego humic gleysol	5	1	Hermawan and Bomke, 1997
Canada	152	817	1	Humic gleysol	20	5	N'Dayegamiye and Tran, 2001
Denmark	18	858	7.9	Ortic haplohumod	20	23	Hansen et al., 2000
Denmark	24	626	7.8	Mollic luvisol	13	15	Schjønning et al., 2012
Denmark	45	862	7.6	Alfisol	20	9	Thomsen and Christensen, 2004
France	512	604	11.5	Haplic luvisol	28	16	Constantin et al., 2010
France	105	1213	12.1	Dystric cambisol	30	13	Constantin et al., 2010
France	203	605	10.8	Rendzina	23.5	13	Constantin et al., 2010
Germany	44	495	8.9	Cambisol	20	10	Barkusky et al., 2009
Germany	47	586	9	Luvisol	20	18	
Germany	95	485	11.1	Cambisol	30	15	Ganz, 2013
Germany	152	600	8	Luvisol	28	8	Sadat-Dastegheibi, 1974
Germany	243	600	8	Cambisol	20	3	von Boguslawski, 1959
India	215	400	28.3	Ustochrept	15	6	Chander et al., 1997
India	215	400	28.3	Ustochrept	15	11	Goyal et al., 1999
India	217	710	25.5	Ustochrept	30	2	Mandal et al., 2003
India	247	500	24	Ustipsamment	15	15	
India	244	1350	24	Hapludoll	15	15	
India	129	818	25.8	Ustochrept	15	14	
India	105	850	24	Chromustert	15	13	
India	113	1100	24	Udic fluvent	15	13	
India	37.2	1358	24	Ustochrept	15	12	
India	10	1360	25.2	Ustochrept	15	12	
Italy	1	900	14.3	Typic xerofluvent	30	15	Mazzoncini et al., 2011
Japan	12	1250	13	Loam	20	13	Ishikawa, 1988
Japan	12	1250	13	Loam	20	54	Ishikawa, 1988
Mexico	2298	1100	14	Andisol	25	2	Astier et al., 2006
Sweden	41	545	5	Eutric cambisol	20	38	
Sweden	41	545	5	Eutric cambisol	15	35	
USA	108	1134	10.8	Fragiudalf	30?	15	Drinkwater et al., 1998
USA	350	960	9	Aquic fragiudalf	20	4	Eckert, 1991
USA	30	1264.9	17.4	Typic hapludults	30	3	Hargrove, 1986
USA	109	1201	18.9	Kandiudult	2.5	3	Hubbard et al., 2013
USA	114	1633	10.7	Aquic xerofluvent	30	6	Kuo et al., 1997
USA	196	1300	15	Typic hapludult	30	3	McVay et al., 1989
USA	255	1299	16.8	Rhodic paleudult	30	3	McVay et al., 1989
USA	16	1258	18.1	Plinthic paleudults	30	3	Sainju et al., 2006
USA	16	1258	18.1	Rhodic kandiupults	30	3	Sainju et al., 2002
USA	16	1258	18.1	Orthic luvisols	20	5	Sainju et al., 2002
USA	294	1148.1	12.8	Maury silt loam	7.5	2	Utomo et al., 1990

2.5 cm to 120 cm and from single layer to multilayer sampling with up to four depth increments. However, only three studies investigated the effect of cover crops on SOC stocks below the plough layer. Bulk density, which is needed to calculate SOC stocks or readily calculated SOC stocks was only given in 13 studies (30%). Missing bulk densities were derived using a pedotransfer function based on the negative correlation of organic carbon concentration and bulk density (Tranter et al., 2007). Instead of using a published pedotransfer function which is biased by the data it is derived from, we used the data from the collected studies, which reported both, bulk density and C concentration to derive a transfer function (R^2 = 0.90, Fig. 1). We then calculated the SOC stock (Mg ha^{-1}) as:

$$\text{SOCstock} = \text{SOCconc} \times \text{BD} \times D$$

where SOCconc is the SOC concentration (%), BD is bulk density of the individual depth increment (g cm^{-3}) and D is the length of the individual depth increment (cm). Comparing SOC stocks of different treatments with different bulk densities to a fixed depth implies an unequal soil mass comparison. However, investigating absolute changes in SOC stocks requires the analysis of equivalent soil masses (Ellert and Bettany, 1995). This is usually achieved by mass correction (Poeplau et al., 2011). After applying the pedotransfer function to estimate missing bulk density values, we thus compared bulk densities of reference and cover crop treatment and found a very small mean difference of 0.014 g cm^{-3} (1%). We corrected that difference by using the bulk density of the cover crop treatment, which was usually lower, to calculate the SOC stocks of the cover crop and the reference treatment. This correction was described by Poeplau et al. (2011) as a simple ratio correction. Finally, we summed the different depth increments of all multilayer plots to one layer per plot with the exception of the three studies, in which the subsoil was assessed. Here we calculated a plough layer SOC stock (usually 0–30 cm) and a subsoil SOC stock. Since only three studies assessed subsoil SOC changes, we excluded subsoils from the analysis. If a site was sampled more than once at different points in time, we used the results of the last reported sampling.

We collected the following environmental, management and sampling design parameters as potential explanatory variables: elevation above sea level; mean annual precipitation (MAP); meanannual temperature (MAT); soil type; soil texture with sand, silt and clay content; sampling depth; time since introduction of

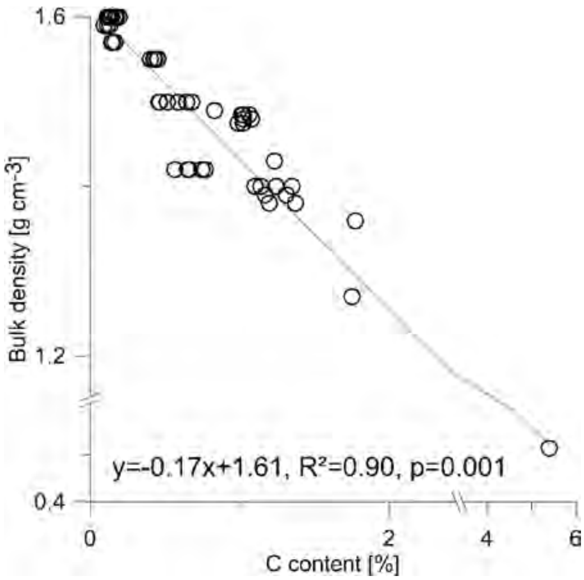


Fig. 1. Correlation of carbon content (%) and bulk density ($g\,cm^{-3}$) with regression line and equation for the compiled studies which reported both parameters.

cover crops; main crop type or crop rotation; cover crop type; cover crop frequency and cover crop above and below ground biomass; C:N ratio of the cover crop; amount and type of fertilization; type of tillage, number of soil samples. Initial carbon concentration was also used as potential explanatory variable. Data gaps in elevation were filled using Google Earth when coordinates were given. Missing MAT and MAP values were filled with data from weatherbase.com, where the closest weather station was selected. When soil texture was only given as texture class, we used the class means for clay, silt and sand (%) according to the used soil classification.

2.2. Statistics

Initially, a *t*-test was conducted to test whether the SOC stocks of the two treatments (cover crops and no cover crops) were significantly different. This was done by testing the SOC stock change against zero. The level of acceptance was 0.05. Subsequently, we plotted SOC stock change against the "time since introduction of cover crops" to assess whether the concept of a temporal carbon response function (CRF) would be applicable for the derived dataset. A significant correlation was found and thus we used linear mixed effect models to explain the SOC stock changes after the introduction of cover crops in two steps: (1) time since introduction as single fixed effect (general CRF) and (2) time since conversion and the above mentioned environmental, management and sampling design parameters as explanatory variables (sSpecific CRF) (Poeplau et al., 2011). Unfortunately, the variables cover crop-biomass and cover crop C:N ratio, which might both have had a high potential explanatory power, were only given for 56% and 25% of all plots, respectively. Furthermore, a huge variability among studies was found for how biomass was determined and which parts of the plants were considered. Therefore both variables could not be used in the analysis. As random effects we used author and site. Author as random effect accounts for the fact that all plots which have been investigated by the same author might be biased by author-specific sampling and analysis methods. Site as a random effect accounts for the fact, that the results of different plots with varying treatments at the same site are not independent from each other. Models were fit by maximum likelihood using the nlme package of the statistical software R (Pinheiro et al., 2009). Model selection was based upon the Akaike information criterion (AIC) as a measure of expected predictive performance of the different candidate models. The accuracy of the models was evaluated by comparing observed and predicted values using the modeling efficiency (EF) described by (Loague and Green, 1991). If EF <0 then the predicted values are worse than simply the mean of all observations. The maximum EF is one. Errors given in the text are 95% confidence intervals.

2.3. Modeling

The distribution of the observed data did not show any SOC saturation with time since introduction. The linear regression model or carbon response function (CRF) as the best fit can thus be used to calculate an annual change rate. However, it is likely that also a system with cover crops will reach a new steady state after a certain time. To estimate the mean overall effect of cover crops as a measure to sequester SOC, a potential steady state needs to be determined. Thus we used the Rothamsted carbon model (RothC) (Coleman and Jenkinson, 1996) to simulate an "average scenario" based on the observed values. For this purpose we used the average values of all plots as model input parameters (MAP, MAT, clay content, sampling depth), calculated plant inputs by backwards simulation and conducted a spin-up run to receive the starting conditions for the fictive average cropland. A mean C input of $1.87 \pm 0.22\,Mg\,ha^{-1}\,yr^{-1}$, was dedicated to the cover crops. We derived this value from the studies ($n = 12$) that reported above- and below-ground cover crop biomass. This average additional C input was used to simulate the average cover crop effect on the SOC stock for 500 years. The lower and upper limits of the 95% confidence interval of the mean C input were used as inputs for additional model runs ($1.65\,Mg\,ha^{-1}\,yr^{-1}$ and $2.09\,Mg\,ha^{-1}\,yr^{-1}$, respectively) to estimate the input associated uncertainty. We defined the new steady state to be reached after the annual change in SOC stock fell below $0.01\,Mg\,ha^{-1}\,yr^{-1}$.

2.4. Estimation of the global potential to cultivate cover crops

Based on remote sensing products, Siebert et al. (2010) estimated the total global cropland area to recently cover $16\,m\,km^2$. Statistics on the actual global cover crop acreage do not exist. For the US and Europe, different surveys (farmer surveys, seed dealer surveys, crop surveys) with different spatial extents (from regional to continental) have been conducted (Singer et al., 2007; CEAP, 2012; CTIC and SARE, 2013; EUROSTAT, 2013). The results range from 1 to 10% of the total cropland area that is already used for cover crop cultivation. Furthermore, a huge share of agricultural area is used for winter crops, which precludes the cultivation of cover crops without a major change of cropping system. In Europe (EU28), 50% of the total cropland area is actually covered each winter, mostly by winter cereals (EUROSTAT, 2013). Siebert et al. (2010) calculated very similar mean crop duration ratios for all continents (0.41 for Africa, 0.47 for America, 0.5 for Asia, 0.56 for Europe and 0.42 for Oceania), which allows the assumption that the European 50% value for winter or off-season fallows can be used for a global cover crop scenario. Additionally, environmental constraints for the growth of cover crops, such as too low winter temperature and water limitations have to be considered (Dabney et al., 2001), but are difficult to estimate spatially. Finally, some crops, such as potatoes or sugarbeet, which are harvested late, do not allow the cultivation of a cover crop. In our scenario we conservatively assume that half of the global winter or off-season fallows could be used for cover crop cultivation, which would result in 25% of the total cropland area ($4\,m\,km^2$).

3. Results

3.1. SOC stock change after introduction of cover crops

The use of cover crops as green manure led to a significant increase in SOC stocks ($p < 0.001$). Time since introduction of cover crops had a significant influence on the SOC stock change (Fig. 2) with a mean annual carbon sequestration rate of 0.32 ± 0.08 Mg ha^{-1} yr^{-1} ($R^2 = 0.17$). Between 1 and 54 years of cover crop cultivation, we did not see any indication of saturation, however, there was very limited data from longer term experiments that improve the ability to demonstrate saturation.

Thus, SOC stock change as a function of time was best described by a linear model. This model was forced through zero, assuming that the cover crop effect on the SOC stock at the start of the experiment was zero. Unfortunately, only 8 plots exceeded the age of 20 so the few long term observations might have strongly influenced the linear response function. However, leaving out the four oldest observations only slightly affected the regression (sequestration rate of 0.35 Mg ha^{-1} yr^{-1}). 102 out of 139 observations had an annual change rate between 0 and 1 Mg ha^{-1} yr^{-1}, while at 13 plots a SOC stock depletion was measured and at 24 plots the change rate exceeded 1 Mg ha^{-1} yr^{-1} (Fig. 3).

3.2. Influence of environmental and management variables

The different environmental and management variables were assessed regarding their explanatory power in the linear mixed effect model to improve the simple response function. The basic model with "time since introduction" as the only fixed effect had a modeling efficiency of EF = 0.23, which resembles the R^2 (0.17) of the simple linear regression (Fig. 2,). Elevation ($p = 0.037$) and sampling depth ($p = 0.049$) were the only variables which could add some explanatory power to the model. However, the EF value increased only slightly (EF = 0.26), since most of the higher values were almost unaffected by the model improvement. However, since all fixed effects (time since introduction, elevation and sampling depth) did not vary within a site, we conducted the same analysis with the site means and only "author" as random effect (Fig. 4a and b). We thus reduced the dataset by the part of the variance, which is treatment specific within the sites and could not be explained with the used variables. The general response function improved considerably (EF = 0.44). Again, elevation and sampling depth were found as significant explanatory variables increasing the model accuracy to EF = 0.56. Thus, the specific CRF which explained the SOC accumulation due to cover crops best for the given dataset was:

$$\Delta\text{SOCstock} = 0.227 \times \text{time} - 0.003 \times \text{elevation} + 0.108 \times \text{depth}$$

with the units years, m a.s.l. and cm for time, elevation and depth, respectively.

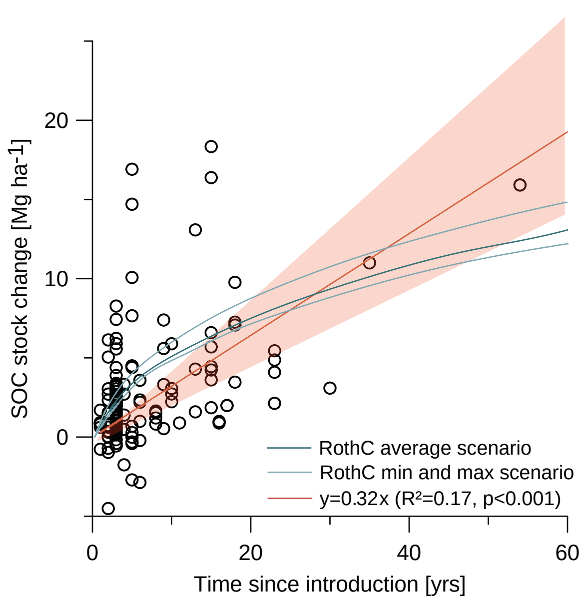


Fig. 2. SOC stock change after cover crop introduction as a function of time with linear regression (with 95% confidence interval) and the RothC simulated average cropland (with min and max scenario).

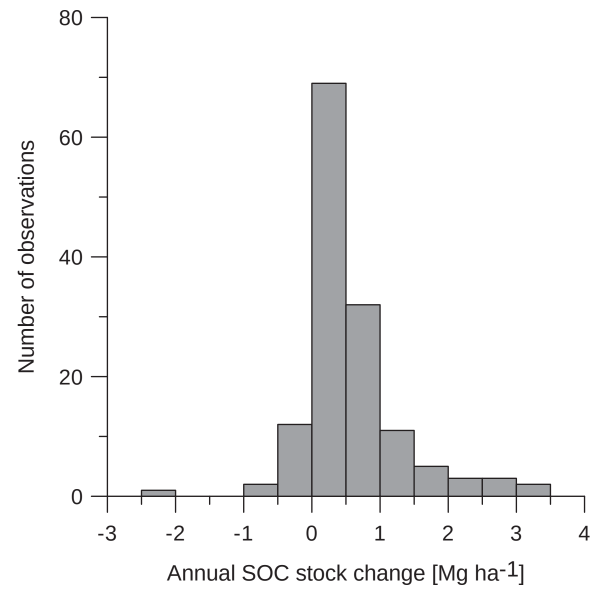


Fig. 3. Histogram of annual change of soil carbon due to cover cropping in comparison to fallow winter.

3.3. Influence of categorical variables

Two categorical variables which could not be used in the response function were tillage regime (no tillage vs. tillage) and cover crop type. Since a total of 27 different species were used as cover crops, we categorized them into the plant functional types legumes and non-legumes. The tillage categories were unbalanced with 41 untilled and 90 tilled plots, while the plant functional types were balanced with 66 non-legume and 61 legume plots (Fig. 5). Both variables were not given for all studies. A third variable tested was climatic zone, (temperate vs. tropic) for which the categories were not balanced (124 and 15 plots). We did not find any significant differences between the categories for any variable.

3.4. Steady state modeling for the average cropland and global C sequestration potential

The average cropland, which was initialized in RothC with a cover crop-derived C input of 1.87 Mg ha^{-1} yr^{-1}, showed a comparable SOC accumulation as the simple linear regression during the first decades (Fig. 2). After 54 years, the predicted SOC accumulation was 12.7 Mg C ha^{-1}, which corresponded to an average annual carbon sequestration rate of 0.23 Mg ha^{-1} yr^{-1}. The low input scenario predicted an SOC accumulation of 11.77 Mg ha^{-1} (0.21 Mg ha yr^{-1}) and the high input scenario predicted an SOC accumulation of 14.12 (0.26 Mg ha yr^{-1}). Those values resemble the found sequestration rate and therefore the simulation can be accepted as a realistic average scenario.

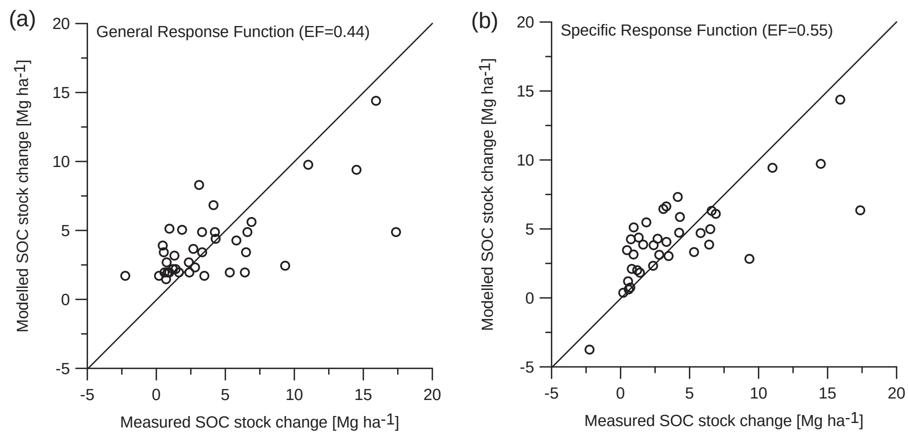


Fig. 4. Modeled vs. measured SOC stock changes for (a) the general "carbon response function" (CRF) applied to site means and (b) the specific CRF applied to site means.

The uncertainty dedicated to C input was less than half of the uncertainty of the linear regression. This is due to the fact, that the variation found in the observations has diverse additional sources, such as abiotic site conditions, degree of carbon saturation, nitrogen fertilization, tillage regime, cover crop–main crop interactions and finally methodological differences among studies. A new equilibrium was reached after 155 years with a total SOC stock change of $16.7 \pm 1.5\,\mathrm{Mg\,ha^{-1}}$ and a relative SOC change of +58 ± 5%. However, half of this new equilibrium (a change of $8.35\,\mathrm{Mg\,ha^{-1}}$) was reached already after 23 years.

Under the assumption that the analyzed dataset of this study is a representative sample of global croplands and with the described 25% scenario, the average potential to sequester SOC via cover crops would be $0.12 \pm 0.03\,\mathrm{Pg\,C\,yr^{-1}}$ (annual change rate for about 50 years) or in total $6.7 \pm 0.6\,\mathrm{Pg\,C}$ (RothC predicted mean steady-state scenario).

4. Discussion

Inclusion of cover crops rather than allowing a fallow period increases the SOC stock of cropland soils and can thus be an effective measure to compensate anthropogenic greenhouse gas emissions (Lal, 2004). Cover cropping improves the net ecosystem carbon balance of a cropland by replacing the bare fallow period (carbon source) by an additional period of carbon assimilation (Lal, 2001). In contrast to other organic amendments, a large part of the C input from cover crop is added as roots, which was found to contribute more effectively to the relatively stable carbon pool than above ground C-input (Kätterer et al., 2011). Additionally, increased SOC might lead to a positive feedback on plant growth and thus increase the C input of the main crop (Brock et al., 2011). Smith et al. (2005) synthesized different agricultural management practices leading to increased SOC stocks. In this study, the different practices were given with a range (low estimate–best estimate–high estimate) as derived from different field studies or other synthesis works. Cover crops were not among the 22 different management options examined by Smith et al. (2005). However, the accumulation rate of $0.32 \pm 0.08\,\mathrm{Mg\,C\,ha^{-1}\,yr^{-1}}$ measured in the current study compares well to other highly effective land use changes and management practices (low estimate of cropland to forest or grassland conversion, high estimate of sewage sludge applications).

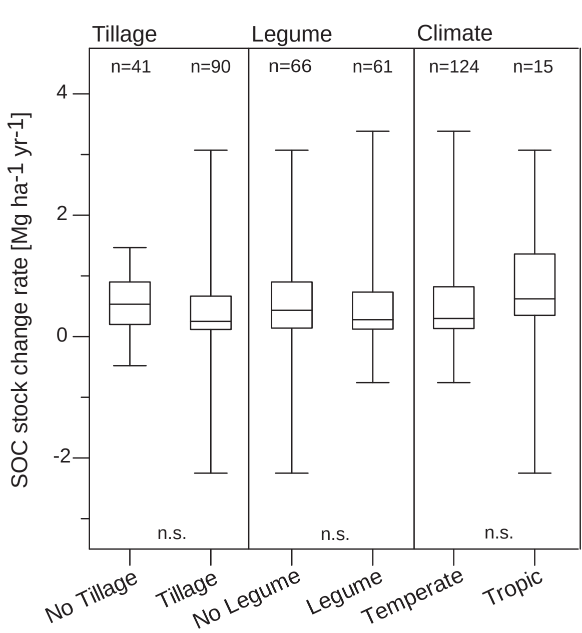


Fig. 5. Boxplots with number of observations for the categorical variables tillage vs. no tillage, plant functional type (no legume vs. legume) and climate zone (temperate vs. tropic).

In our study, we found 13 out of 139 plots with a SOC stock depletion after introducing cover crops. This SOC depletion might have two different reasons; (1) priming: the addition of rapidly decomposable plant material (low C:N ratio) leads to microbial community growth and enough energy became available to break up more stable compounds of old SOC as compared to the no cover crop treatment. Fontaine et al. (2004) showed under controlled laboratory conditions that the addition of fresh C can lead to accelerated C decomposition and thus C losses. (2) Spatial heterogeneity of SOC at sampling sites: the naturally occurring spatial variability of SOC concentrations may overlay the relatively small effects of cover crops on SOC. The latter explanation becomes more likely, when the time since cover crop introduction of the 13 plots with SOC depletion is considered: the oldest plot showing SOC stock depletion was six years old. The same problem does probably apply for those plots, which showed an annual accumulation rate of $>2\,\mathrm{Mg\,C\,ha^{-1}\,yr^{-1}}$. The oldest of these eight plots were five years old. The SOC accumulation rate in these specific plots even exceeded the annual cover crop biomass production (in C) (N'Dayegamiye and Tran, 2001). Spatial heterogeneity is a major problem when assessing or monitoring SOC stock changes over time, even in real time series (resampling of the same plot) (Cambardella et al., 1994). Small differences between treatments are difficult to detect. Goidts et al. (2009) found an average SOC variability at the plot scale of 20% and concluded that only changes >20% could be detected. A large part of the analysed dataset consists of young treatments: the mean time since introduction of the cover crops for all plots was 6.8 years with 108 plots (78%) having a cover crop history of <10 years. It is likely, that a considerable part of the observed variability of SOC stock changes for these plots could be explained by initial differences in SOC between the plots and insufficient time since experiments establishment to be able to detect treatment effects against the background noise due to spatial SOC variability and measurement errors. However, the mean annual change rate, which was derived by the slope of the response function, remained relatively unaffected by those over- and underestimations of SOC stock changes in the early years of the experiments.

Consequently, the investigated explanatory variables did not add much power to the general response function. Only elevation and sampling depth could slightly increase the modeling efficiency, while elevation was negatively correlated and sampling depth was positively correlated with the annual change in SOC stock. Elevation as such does not add much information to the model and it might be a dataset specific phenomenon that elevation enhances a model explaining SOC dynamics. In fact, the highest SOC stock depletion was found at the highest plot in the dataset (2298 m) (Astier et al., 2006) and the highest annual SOC stock accumulation has been found at one of the lowest plots (16 m) (Sainju et al., 2006). Those extreme values might have strongly determined the explanatory power of the variable "elevation". However, running the model without these two extremes did even slightly improve the model (AIC of 203 decreased to 194). The explanatory power of elevation can thus not be ignored. Elevation was slightly negatively correlated with MAT and MAP (not significant), which suggests that elevation in this dataset could be interpreted as a meteorological proxy for temperature and rainfall. With increasing elevation, the climatic conditions, at least in the present dataset, get colder and dryer. However, both variables individually did not add any explanation to the model.

The explanatory power of the variable sampling depth can be explained by the fact that the mass of the soil that is sampled determines the proportion of the total SOC stock that is sampled. This is especially important for experiments, in which ploughing occurs every year and carbon inputs are equally distributed to a depth of usually about 30 cm. The carbon concentration in the plough layer is thus rather homogeneous. If only the upper 20 cm of soil are sampled, while the soil is ploughed to a depth of 30 cm, one third of the cover crop effect could be missed. Poeplau et al. (2011), who derived carbon response functions for different land use changes found a negative correlation between sampling depth and SOC stock change, which is in contrast to the result of the present study. The difference is that they derived functions for the relative SOC stock change [%], while in the present study absolute changes are assessed.

A difference in annual SOC stock change between tilled and untilled plots was not observed. A difference was expected since the input pathways and thus the distribution and potentially the decomposition rate of freshly derived SOC differ considerably between both treatments (Don et al., 2013). In untilled systems, the aboveground residues of both, main crop and cover crop remain on the surface while tillage leads to a regular mixture of residues and mineral soil. Neither the plant functional type (legume vs. non legume) nor the climatic zone (temperate zone vs. tropics) influenced the annual SOC stock change, which might again be explicable by an insufficient size of the dataset.

Finally, it is likely that the absolute carbon input via cover crops would be a major explanatory variable (Hubbard et al., 2013). However, we were not able to use it in the response function, since reliable information about above- and below-ground carbon inputs were scattered. The same is true for the C:N ratio of the biomass, which might determine the turnover time of the material in the soil and the proportion of carbon that is used for microbial growth and thus remains in the soil in more stable organic forms (Schimel and Weintraub, 2003).

SOC stock change is often given in an annual change rate to obtain a direct measure for the climate change mitigation effect of a certain management or land use change (Smith et al., 2005). However a change rate is often not entirely correct or can be even misleading, since it implies either that the change in SOC stock is infinitely linear or that the exact time to reach a new steady state (transition time) and the new steady state SOC stock is known (West and Six, 2007). In fact, SOC accumulation tends to reach a dynamic steady state after a certain time (homeostasis) (West and Six, 2007; Barbera et al., 2012). If a sequestration or depletion of SOC is not linear and saturation occurs, the annual change rate must decrease annually. In addition to the linear regression, we therefore used the RothC model to estimate the total mean SOC stock change. A new steady state occurred after 155 years, which is similar to the curve progression found by Poeplau et al. (2011) for the conversion from cropland to grassland, which showed no steady state after 120 years. In the traditional Rothamsted long term experiments such as Broadbalk and Hoosfield, C sequestration after organic manuring continues since 1844 (Johnston et al., 2009). SOC sequestration of more than a century before a new steady-state is reached is thus realistic. The total mean SOC stock change of $16.7 \pm 1.5\,\mathrm{Mg\,ha^{-1}}$ in the upper 22 cm of the soil resembles the SOC stock change that was reported by Poeplau and Don (2013) for the conversion from cropland to grassland at six different sites across Europe. They observed a mean SOC stock change of $17 \pm 5\,\mathrm{Mg\,ha^{-1}}$ in the 0–30 cm soil increment and $18 \pm 9\,\mathrm{Mg\,ha^{-1}}$ in the 0–80 cm increment after a mean time of 33 years since grassland establishment. Relatively, the estimated SOC stock change of $16.7\,\mathrm{Mg\,ha^{-1}\,yr^{-1}}$ accounts for an increase of 50%. This is well within the range of observed changes in the Ultuna frame trial, a 59 year old long term experiments with a large amount of different organic amendments (Kätterer et al., 2011). When deep rooting cover crops are used, the widespread topsoil sampling might not be sufficient to capture the total effect of cover crops on SOC stocks. The found and estimated average accumulation of SOC might thus resemble a slight underestimation due to shallow sampling in the majority of studies.

Estimates on the potential global C sink due to cover crops are mainly hampered by missing data on the recent area of cover crop cultivation and on environmental, economic and technical constraints that limit cover crop cultivation on a global scale. There are only regional assessments on cover crop cultivation based on proxies such as cover crop seed production and farmers surveys (Bryant et al., 2013). The scenario we used in this study in order to estimate the global potential of cover crops for C sequestration (25% of the total cropland area) was rather conservative but might serve as a rough estimate. The calculated potential of $0.12 \pm 0.03\,\mathrm{Pg\,C\,yr^{-1}}$ ($0.44\,\mathrm{Pg\,CO_2\,yr^{-1}}$) would compensate for 8% of the annual direct greenhouse gas emissions from agriculture (IPCC, 2007) or about 70% of the global greenhouse gas emissions from aviation. This offsetting would potentially last for several decades and slowly decrease until saturation occurs. In practice some farmers might be discouraged from higher costs or the higher presence of volunteer weeds. However, on top of the potential SOC sequestration, cover crops take up excess soil nitrogen, which prevents N leaching and may thus reduce N_2O emissions. Furthermore, cover crops reduce the albedo of croplands with possible effects on climate forcing. To our knowledge both issues have not been addressed sufficiently. Finally, carbon costs for soil preparation and during cover crop elimination and incorporation were also not considered in the scenario.

5. Conclusions

Cover crops used as green manure are an important management option to increase SOC stocks in agricultural soils. So far, this has been insufficiently quantified and widely overlooked. In this meta-analysis we quantified for the first time the general potential of cover crops to enhance SOC. We comprised the majority of available cover crop studies worldwide and found a mean annual SOC sequestration of $0.32 \pm 0.08\,\mathrm{Mg\,ha^{-1}\,yr^{-1}}$ to an average maximum increase of $16.7\,\mathrm{Mg\,ha^{-1}}$. This is in the same order of magnitude as found for other organic-input-related C sequestration management options in agricultural soils and almost as effective as land-use changes like afforestations of croplands. Carbon sequestration could potentially last for more than 100 years, while 50% of the total effect on SOC stocks is likely to occur in the first two decades. The relatively high sequestration rate combined with the large spatial extent of potential cultivation areas allows the conclusion that cover crop cultivation is a sustainable and efficient measure to mitigate climate change. Moreover, cover crops can contribute to reduced nutrient leaching and enhanced nutrient efficiency, reduced wind and water erosion and pest control, which would make cover crops environmental beneficial and long-term economically sound. The predictive power of the derived response function remained comparatively low, mainly due to a limited number of datasets and a high proportion of very short experiments (<10 years). More work is needed to understand e.g., species-specific effects on SOC stocks, but also the effects of cover crops on albedo and N_2O emissions.

Acknowledgements

For data collection we thank Birk Stern. For fruitful discussions we thank Tobias KD Weber. This study was financed by the FP7 project GHG-Europe (grant no. 244122).

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2014.10.024.

References

Amado, T.J.C., Bayer, C., Conceicao, P.C., Spagnollo, E., de Campos, B.H.C., da Veiga, M., 2006. Potential of carbon accumulation in no-till soils with intensive use and cover crops in southern Brazil. J. Environ. Qual. 35, 1599–1607.

Astier, M., Maass, J.M., Etchevers-Barra, J.D., Peña, J.J., González, F.d.L., 2006. Short-term green manure and tillage management effects on maize yield and soil quality in an Andisol. Soil Tillage Res. 88, 153–159.

Barbera, V., Poma, I., Gristina, L., Novara, A., Egli, M., 2012. Long-term cropping systems and tillage management effects on soil organic carbon stock and steady state level of C sequestration rates in a semiarid environment. Land Degrad. Dev. 23, 82–91.

Barkusky, D., Baumecker, M., Chmielewski, F.-M., Ellmer, F., Gagern, W., Hierold, W., Käding, H., Köhn, W., Ruehlmann, J., 2009. Dauerfeldversuche in Brandenburg und Berlin- Beiträge fuer eine nachhaltige landwirtschaftliche Bodennutzung. Schriftenreihe des Landesamtes fuer Verbraucherschutz, Landwirtschaft und Flurneuordnung Abteilung Landwirtschaft und Gartenbau, Reihe Landwirtschaft, 10, pp. 4.

Bayer, C., Mielniczuk, J., Amado, T.J.C., Martin-Neto, L., Fernandes, S.V., 2000. Organic matter storage in a sandy clay loam Acrisol affected by tillage and cropping systems in southern Brazil. Soil Tillage Res. 54, 101–109.

Blombäck, K., Eckersten, H., Lewan, E., Aronsson, H., 2003. Simulations of soil carbon and nitrogen dynamics during seven years in a catch crop experiment. Agric. Syst. 76, 95–114.

Brock, C., Fließbach, A., Oberholzer, H.-R., Schulz, F., Wiesinger, K., Reinicke, F., Koch, W., Pallutt, B., Dittman, B., Zimmer, J., Hülsbergen, K.-J., Leithold, G., 2011. Relation between soil organic matter and yield levels of nonlegume crops in organic and conventional farming systems. J. Plant Nutr. Soil Sci. 174, 568–575.

Bryant, L., Stockwell, R., White, T., 2013. Counting Cover Crops. In: Federation, N.W. (Ed.), http://www.nwf.org/news-and-magazines/media-center/reports/archive/2013/10–01-13-counting-cover-crops.aspx

Cambardella, C.A., Moorman, T.B., Parkin, T.B., Karlen, D.L., Novak, J.M., Turco, R.F., Konopka, A.E., 1994. Field-scale variability of soil properties in central Iowa soils. Soil Sci. Soc. Am. J. 58, 1501–1511.

Campbell, C.A., Biederbeck, V.O., Zentner, R.P., Lafond, G.P., 1991. Effect of crop rotations and cultural practices on soil organic-matter: microbial biomass and respiration in a thin black chernozem. Can. J. Soil Sci. 71, 363–376.

CEAP, 2012. Assessment of the effects of conservation practices on cultivated cropland in the upper Mississippi river basin. In: Service, N.R.C. (Ed.), http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1042093.pdf.

Chander, K., Goyal, S., Mundra, M.C., Kapoor, K.K., 1997. Organic matter, microbial biomass and enzyme activity of soils under different crop rotations in the tropics. Biol. Fertil. Soils 24, 306–310.

Coleman, D.C., Jenkinson, D.S., 1996. RothC-26.3 – a model for the turnover of carbon in soil. In: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of Soil Organic Matter Models Using Existing Long-term Datasets. Springer, Berlin, pp. 237–246 NATO ASI Series I.

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P., Beaudoin, N., 2010. Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. Agric. Ecosyst. Environ. 135, 268–278.

CTIC, SARE, 2013. Cover crop survey, http://www.northcentralsare.org/Educational-Resources/From-the-Field/Cover-Crops-Survey.

Curtin, D., Wang, H., Selles, F., Zentner, R.P., Biederbeck, V.O., Campbell, C.A., 2000. Legume green manure as partial fallow replacement in semiarid Saskatchewan: effect on carbon fluxes. Can. J. Soil Sci. 80, 499–505.

Dabney, S.M., Delgado, J.A., Reeves, D.W., 2001. Using winter cover crops to improve soil and water quality. Commun. Soil Sci. Plant Anal. 32, 1221–1250.

DeFries, R.S., Field, C.B., Fung, I., Collatz, G.J., Bounoua, L., 1999. Combining satellite data and biogeochemical models to estimate global effects of human-induced land cover change on carbon emissions and primary productivity. Glob. Biogeochem. Cycle 13, 803–815.

Don, A., Schumacher, J., Freibauer, A., 2011. Impact of tropical land-use change on soil organic carbon stocks – a meta-analysis. Glob. Change Biol. 17, 1658–1670.

Drinkwater, L.E., Wagoner, P., Sarrantonio, M., 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396, 262–265.

Eckert, D.J., 1991. Chemical attributes of soils subjected to no-till cropping with rye cover crops. Soil Sci. Soc. Am. J. 55, 405–409.

Ellert, B.H., Bettany, J.R., 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. Can. J. Soil Sci. 75, 529–538.

EUROSTAT, 2013. Agri-environmental indicator – soil cover. In: Commission, E. (Ed.), http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Agri-environmental_indicator_-_soil_cover.

Fontaine, S., Bardoux, G., Abbadie, L., Mariotti, A., 2004. Carbon input to soil may decrease soil carbon content. Ecol. Lett. 7, 314–320.

Frye, W.W., Blevins, R.L., Smith, M.S., Corak, S.J., 1988. Role of Annual Legume Cover Crops in Efficient Use of Water and Nitrogen. American Society of Agronomy ASA Special Publication.

Ganz, M., 2013. Kohlenstoffvorrat und Stratifikation der Pflanzennährstoffe im Boden unter dem Einfluss langjährig Pflugloser Bodenbe-arbeitung mit und ohne Gründüngung. University of Applied Science Bingen, Bingen.

Goidts, E., Van Wesemael, B., Crucifix, M., 2009. Magnitude and sources of uncertainties in soil organic carbon (SOC) stock assessments at various scales. Eur. J. Soil Sci. 60, 723–739.

Goyal, S., Chander, K., Mundra, M.C., Kapoor, K.K., 1999. Influence of inorganic fertilizers and organic amendments on soil organic matter and soil microbial properties under tropical conditions. Biol. Fertil. Soils 29, 196–200.

Hansen, E.M., Kristensen, K., Djurhuus, J., 2000. Yield parameters as affected by introduction or discontinuation of catch crop use. Agron. J. 92, 909–914.

Hargrove, W.L., 1986. Winter legumes as a nitrogen source for no-till grain sorghum. Agron. J. 78, 70–74.

Hermawan, B., Bomke, A.A., 1997. Effects of winter cover crops and successive spring tillage on soil aggregation. Soil Tillage Res. 44, 109–120.

Hubbard, R.K., Strickland, T.C., Phatak, S., 2013. Effects of cover crop systems on soil physical properties and carbon/nitrogen relationships in the coastal plain of southeastern USA. Soil Tillage Res. 126, 276–283.

IPCC, et al., 2007. IPCC Climate Change: The Physical Science Basis. In: Solomon, S. (Ed.), Cambridge University Press, Cambridge.

Ishikawa, M., 1988. Green manure in rice – the Japan experience. In: Institute, T.i.R. R. (Ed.), The Role of Green Manure Crops in Rice Farming Systems. Lagunas, Phillipines.

Johnston, A.E., Poulton, P.R., Coleman, K., 2009. Soil organic matter: its importance in sustainable agriculture and carbon dioxide fluxes. Advances in Agronomy, Vol. 101, Elsevier Academic Press Inc., San Diego, pp. 1–57.

Kätterer, T., Bolinder, M.A., Andrén, O., Kirchmann, H., Menichetti, L., 2011. Roots contribute more to refractory soil organic matter than above-ground crop residues as revealed by a long-term field experiment. Agric. Ecosyst. Environ. 141, 184–192.

Kuo, S., Sainju, U.M., Jellum, E.J., 1997. Winter cover crop effects on soil organic carbon and carbohydrate in soil. Soil Sci. Soc. Am. J. 61, 145–152.

Lal, R., 2001. World cropland soils as a source or sink for atmospheric carbon. Advances in Agronomy. Academic Press, pp. 145–191.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123, 1–22.

Loague, K., Green, R.E., 1991. Statistical and graphical methods for evaluating solute transport models overview and application. J. Contam. Hydrol. 7, 51–74.

Luo, Z., Wang, E., Sun, O.J., 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agric. Ecosyst. Environ. 139, 224–231.

Mandal, U.K., Singh, G., Victor, U.S., Sharma, K.L., 2003. Green manuring: its effect on soil properties and crop growth under rice–wheat cropping system. Eur. J. Agron. 19, 225–237.

Mazzoncini, M., Sapkota, T.B., Bàrberi, P., Antichi, D., Risaliti, R., 2011. Long-term effect of tillage, nitrogen fertilization and cover crops on soil organic carbon and total nitrogen content. Soil Tillage Res. 114, 165–174.

McGuire, A.D., Sitch, S., Clein, J.S., Dargaville, R., Esser, G., Foley, J., Heimann, M., Joos, F., Kaplan, J., Kicklighter, D.W., Meier, R.A., Melillo, J.M., Moore, B., Prentice, I.C., Ramankutty, N., Reichenau, T., Schloss, A., Tian, H., Williams, L.J., Wittenberg, U., 2001. Carbon balance of the terrestrial biosphere in the twentieth century: analyses of CO_2, climate and land use effects with four process-based ecosystem models. Glob. Biogeochem. Cycle 15, 183–206.

McVay, K.A., Radcliffe, D.E., Hargrove, W.L., 1989. Winter legume effects on soil properties and nitrogen fertilizer requirements. Soil Sci. Soc. Am. J. 53, 1856–1862.

Metay, A., Moreira, J.A.A., Bernoux, M., Boyer, T., Douzet, J.-M., Feigl, B., Feller, C., Maraux, F., Oliver, R., Scopel, E., 2007. Storage and forms of organic carbon in a no-tillage under cover crops system on clayey Oxisol in dryland rice production (Cerrados, Brazil). Soil Tillage Res. 94, 122–132.

N'Dayegamiye, A., Tran, T.S., 2001. Effects of green manures on soil organic matter and wheat yields and N nutrition. Can. J. Soil Sci. 81, 371–382.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., 2009. nlme: Linear and Nonlinear Mixed Effects. Models. R package version 3, 1–96.

Poeplau, C., Don, A., 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. Geoderma 192, 189–201.

Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J., Gensior, A., 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone – carbon response functions as a model approach. Glob. Change Biol. 17, 2415–2427.

Sadat-Dastegheibi, B., 1974. Untersuchungen zur Stoffdynamik in Ackerboeden in Abhaengigkeit von verschiedener organischer Duengung und Stickstoffduengung. [s.n.], Giessen.

Sainju, U.M., Singh, B.P., Whitehead, W.F., 2002. Long-term effects of tillage, cover crops, and nitrogen fertilization on organic carbon and nitrogen concentrations in sandy loam soils in Georgia, USA. Soil Tillage Res. 63, 167–179.

Sainju, U.M., Singh, B.P., Whitehead, W.F., Wang, S., 2006. Carbon supply and storage in tilled and nontilled soils as influenced by cover crops and nitrogen fertilization. J. Environ. Qual. 35, 1507–1517.

Schimel, J.P., Weintraub, M.N., 2003. The implications of exoenzyme activity on microbial carbon and nitrogen limitation in soil: a theoretical model. Soil Biol. Biochem. 35, 549–563.

Schjønning, P., de Jonge, L.W., Munkholm, L.J., Moldrup, P., Christensen, B.T., Olesen, J.E., 2012. Clay Dispersibility and Soil Friability—Testing the Soil Clay-to-carbon Saturation Concept. Gsvadzone, pp. 11.

Siebert, S., Portmann, F.T., Döll, P., 2010. Global patterns of cropland use intensity. Remote Sens. 2, 1625–1643.

Singer, J.W., Nusser, S.M., Alf, C.J., 2007. Are cover crops being used in the US corn belt? J. Soil Water Conserv. 62, 353–358.

Smith, P., Andrén, O., Karlsson, T., Perälä, P., Regina, K., Rounsevell, M., Van Wesemael, B., 2005. Carbon sequestration potential in European croplands has been overestimated. Glob. Change Biol. 11, 2153–2163.

Smith, P., Powlson, D.S., Glendining, M.J., Smith, J.U., 1997. Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. Glob. Change Biol. 3, 67–79.

Thomsen, I.K., Christensen, B.T., 2004. Yields of wheat and soil carbon and nitrogen contents following long-term incorporation of barley straw and ryegrass catch crops. Soil Use Manag. 20, 432–438.

Tranter, G., Minasny, B., McBratney, A.B., Murphy, B., McKenzie, N.J., Grundy, M., Brough, D., 2007. Building and testing conceptual and empirical models for predicting soil bulk density. Soil Use Manag. 23, 437–443.

Utomo, M., Frye, W.W., Blevins, R.L., 1990. Sustaining soil nitrogen for corn using hairy vetch cover crop. Agron. J. 82, 979–983.

von Boguslawski, E., 1959. Das zusammenwirken von gründüngung und stickstoffdüngung auf den C- und N-umsatz im boden. Z. Pflanzenernähr. Düng. Bodenkd. 84, 85–93.

West, T., Six, J., 2007. Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. Clim. Change 80, 25–41.

West, T.O., Marland, G., King, A.W., Post, W.M., Jain, A.K., Andrasko, K., 2004. Carbon management response curves: estimates of temporal soil carbon dynamics. Environ. Manag. 33, 507–518.

Wiesmeier, M., Hübner, R., Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., von Lützow, M., Kögel-Knabner, I., 2013. Carbon sequestration potential of soils in southeast Germany derived from stable soil organic carbon saturation. Glob. Change Biol.

Exhibit F

Soil Biology and Biochemistry 143 (2020) 107735

Contents lists available at ScienceDirect

Soil Biology and Biochemistry

journal homepage: http://www.elsevier.com/locate/soilbio

A meta-analysis of global cropland soil carbon changes due to cover cropping

Jinshi Jian[a,b], Xuan Du[c], Mark S. Reiter[a,d], Ryan D. Stewart[a,*]

[a] School of Plant and Environmental Sciences, Virginia Tech, Blacksburg, VA, USA
[b] Pacific Northwest National Laboratory-University of Maryland Joint Global Change Research Institute, 5825 University Research Court, Suite, 3500, College Park, MD, USA
[c] Department of Hydraulic Engineering, Yangling Vocational & Technical College, Yang Ling, Shaanxi, China
[d] Eastern Shore Agricultural Research and Extension Center, Virginia Tech, Painter, VA, USA

ARTICLE INFO

Keywords:
Cover crop
Soil organic carbon
Soil quality
Soil health
Meta-analysis
Database

ABSTRACT

Including cover crops within agricultural rotations may increase soil organic carbon (SOC). However, contradictory findings generated by on-site experiments make it necessary to perform a comprehensive assessment of interactions between cover crops, environmental and management factors, and changes in SOC. In this study, we collected data from studies that compared agricultural production with and without cover crops, and then analyzed those data using meta-analysis and regression. Our results showed that including cover crops into rotations significantly increased SOC, with an overall mean change of 15.5% (95% confidence interval of 13.8%–17.3%). Whereas medium-textured soils had highest SOC stocks (overall means of 39 Mg ha^{-1} with and 37 Mg ha^{-1} without cover crops), fine-textured soils showed the greatest increase in SOC after the inclusion of cover crops (mean change of 39.5%). Coarse-textured (11.4%) and medium-textured soils (10.3%) had comparatively smaller changes in SOC, while soils in temperate climates had greater changes (18.7%) than those in tropical climates (7.2%). Cover crop mixtures resulted in greater increases in SOC compared to mono-species cover crops, and using legumes caused greater SOC increases than grass species. Cover crop biomass positively affected SOC changes while carbon:nitrogen ratio of cover crop biomass was negatively correlated with SOC changes. Cover cropping was associated with significant SOC increases in shallow soils ($\leq$30 cm), but not in subsurface soils ($>$30 cm). The regression analysis revealed that SOC changes from cover cropping correlated with improvements in soil quality, specifically decreased runoff and erosion and increased mineralizable carbon, mineralizable nitrogen, and soil nitrogen. Soil carbon change was also affected by annual temperature, number of years after start of cover crop usage, latitude, and initial SOC concentrations. Finally, the mean rate of carbon sequestration from cover cropping across all studies was 0.56 Mg ha^{-1} yr^{-1}. If 15% of current global cropland were to adopt cover crops, this value would translate to 0.16 $\pm$ 0.06 Pg of carbon sequestered per year, which is ~1–2% of current fossil fuels emissions. Altogether, these results indicated that the inclusion of cover crops into agricultural rotations can enhance soil carbon concentrations, improve many soil quality parameters, and serve as a potential sink for atmosphere CO_2.

1. Introduction

Many woodlands and grasslands are being converted to cropland due to increasing world population and food production requirements. Between 1850 and 1980, at least 900 million hectare (Mha) of naturally vegetated lands were converted to croplands and pastures across the globe (Houghton, 1995), with the conversion process continuing today in many parts of the world (McGuire et al., 2001). Converting lands from natural vegetation to cropland leads to soil organic carbon (SOC) losses. For example, a large-scale study in Germany (Wiesmeier et al., 2013a,

Abbreviations: ha, hectare; Mha, million hectares; SD, Standard deviation; RR, Response ratio; CC(s), Cover crop(s); NC, No cover crops; BD, Bulk density [M L^{-3}]; SOC_{stock}, Soil organic carbon stock [M L^{-2}]; $SOC_{\%}$, Soil organic carbon concentration [M M^{-1}]; C_{rate}, Rate of change of soil organic carbon [M L^{-2} T^{-1}]; $C_{sequestration}$, Carbon sequestered in soil due to cover crop usage [M T^{-1}].

* Corresponding author.
E-mail address: ryan.stewart@vt.edu (R.D. Stewart).

https://doi.org/10.1016/j.soilbio.2020.107735
Received 14 September 2019; Received in revised form 17 January 2020; Accepted 25 January 2020
Available online 27 January 2020
0038-0717/

2013b) showed that cropland soils stored lower amounts of SOC (90 Mg ha^{-1}) than either forest soils (98 Mg ha^{-1}) or grassland soils (118 Mg ha^{-1}). Based on a meta-analysis of 95 studies covering 322 sites with temperate climates, converting grassland to cropland led to mean SOC losses of 36% and converting from forest to cropland caused SOC to decrease by an average of 32% (Poeplau et al., 2011) Another meta-analysis conducted using 385 studies found that converting from primary forest to cropland yielded SOC losses of 25%–30% (Don et al., 2011).

Reversing carbon losses in cropland soils can be a means to sequestrate atmospheric carbon (Gattinger et al., 2012). Elevated SOC is also associated with improved soil health and fertility; therefore, increasing SOC may help to enhance agricultural productivity (Stewart et al., 2018; Van Eerd et al., 2014). Shifts in management practices, including the use of cover crops (CCs) within rotations, has been proposed as a way to increase SOC stocks (Kaye and Quemada, 2017). However, not all studies found CCs to result in SOC accumulation, with some demonstrating SOC losses after introduction of CCs (Bandick and Dick, 1999; Idowu et al., 2009; Ndiaye et al., 2000). Due to differences in climate and management, CCs may need to be used for decades in some systems to cause significant SOC increases (Poeplau and Don, 2015). Results also vary depending on soil texture and type, as some fine-textured soils can help to physically protect SOC from decomposition (Callesen et al., 2003; Krull et al., 2003), depending on factors such as mineralogy and amount of soil aggregation (Hassink and Whitmore, 1997; Schmidt et al., 2011).

Cover crop and rotation types can both affect SOC changes. For example, some studies found grass CCs, including cereal rye [*Secale cereal*] and annual ryegrass [*Lolium multiflorum*], cause greater SOC increases than leguminous CCs like cowpea [*Vigna unguiculate*] and hairy vetch [*Vicia villosa Roth*] (Mazzoncini et al., 2011; O'Dea et al., 2013). Other studies found greater SOC increases after legume CCs compared to grass CCs (Utomo et al., 1990), with mixtures often causing the greatest increases of all (Sainju et al., 2006). One reason for these incongruous results may result from differences in biomass and carbon:nitrogen (C:N) ratios between CC species. Legume CCs typically have low C:N ratios, due in part to their ability to fix atmospheric nitrogen, while grass CCs often have higher biomass but also higher C:N ratios. Crop rotations may also influence SOC accumulation from CCs, both by affecting CC planting and harvesting dates (thus varying the time for biomass accrual) and by altering soil properties such as nutrient availability, soil structure, and soil microbial properties (Bandick and Dick, 1999; Campbell et al., 1991; Sainju et al., 2006). However, it is not well understood whether and how cash crop rotations interact with CCs to alter SOC in agricultural soils.

To resolve uncertainties and discrepancies that can emerge from single-site studies, modern techniques such as meta-analysis have been used to compile and compare results from various investigations of CCs. For example, a meta-analysis for the Pampas region of Argentina showed that SOC significantly increased when CCs were grown in coarse- and fine-textured soils (Alvarez et al., 2017). Aguilera et al. (2013) found that practices combining external organic amendments with CCs caused significant increases in SOC concentrations. Blanco-Canqui et al. (2015) determined that introducing CCs resulted in an SOC increases of 0.1–1.0 Mg ha^{-1} yr^{-1}. While the rate of accumulation slowed through time, the results of that study suggested that more than a century may be needed for SOC concentrations to reach new equilibria. Despite generating such insights, however, most existing meta-analyses focused on a particular region and have not considered climatic influences and environmental factors in the response. These regional responses may therefore have limited applicability towards understanding SOC dynamics at global scales.

Meta-analysis has also not yet been used to comprehensively assess interactions between SOC and other soil properties (e.g., soil penetration resistance, soil nitrogen, soil microbial activity), even though individual studies identified different (and at times contradictory) parameter responses to CCs. For instance, CC introduction led to increases in soil aggregate stability (Marques et al., 2010; Stavi et al., 2012; Tesfahunegn et al., 2016) and water infiltration rates (Haruna et al., 2018), yet other studies demonstrated no effect or even decreased infiltration rates after introduction of CCs (Abdollahi and Munkholm, 2014; Steele et al., 2012). Likewise, some studies found that bulk density (BD) decreased after CC usage (Stavi et al., 2012; Spargo et al., 2008), while others found no effect (Blanco-Canqui et al., 2011, 2013; Jiang et al., 2007). To examine past and current practices that best quantify properties associated with soil health, Stewart et al. (2018) collected historical publications examining CC and no-till practices and integrated those data into a global soil health database called SoilHealthDB (Jian et al., 2019, 2020). That analysis determined that 13 out of 42 soil health indicators showed >10% difference from no cover crop (NC) controls in the first 1–3 years after CCs were introduced. Responsive parameters included soil aggregate stability, nitrogen mineralization rate, and microbial biomass nitrogen and carbon, whereas SOC did not show a consistent short-term response to CC. The question thus remains whether these changes in soil properties help to drive longer-term changes in SOC.

In this present study, we used the SoilHealthDB to further explore SOC dynamics after CCs. The study objectives were to: 1) quantify CC usage effects on SOC concentrations; 2) evaluate how climate type, CC species, and cash crop rotation affected SOC dynamics using meta-analysis; 3) identify possible mechanisms for SOC changes via correlation with other soil/agronomic variables; and 4) estimate carbon sequestration potential as CCs become applied to various extents within cropland across the globe.

2. Methods

2.1. Data collection

To gather data for analysis, we collected publications from three sources: (1) the "Research Landscape Tool" that compiled soil health-related publications and research projects into a searchable database (http://www.soilhealthinstituteresearch.org/); (2) cited papers from previous meta-analyses and review papers, specifically Poeplau and Don (2015), Alvarez et al. (2017), Sileshi (2009), and Gattinger et al. (2012); and (3) scholarship-focused search engines, including ISI Web of Science, Google Scholar, and the China National Knowledge Infrastructure (CNKI). We accessed the Research Landscape Tool (http://www.soilhealthinstituteresearch.org/Home/Search/Cover-Crop) on April 25, 2018, and found all peer-reviewed journal articles listed there under "cover crops". For search engines, we used the keywords "soil health" or "soil quality" and "cover crop" or "green manure" or "organic farm" to find relevant publications. We searched and downloaded more than 500 papers, and then used the following criteria to determine whether a publication would be included in this study: (1) experiments were conducted in the field or at a research station; (2) the publications reported comparisons between controls (e.g., NC control plots; baseline SOC) and treatments (i.e., with CCs); (3) the publications were either peer-reviewed journal articles, conference collections, theses, or dissertations; and (4) the publications were written in English or Chinese. With these constraints, 1195 comparisons were digitized from 131 studies across the globe. More than half (60%) of comparisons were from North America, with the remainder from the other five habitable continents (Fig. 1). Each study reported an average of 9 comparisons, and some comparisons in each study were not independent from one another. Following the method provided by Alvarez et al. (2017), we allocated a unique experiment ID to a comparison if the CC group, cash crop group, site, tillage, fertilization, soil depth, termination date, or rotation type were different from other comparisons (Fig. A1). Our methodology resulted in 581 independent experiments, of which 144 experiments included SD information (Fig. 1).

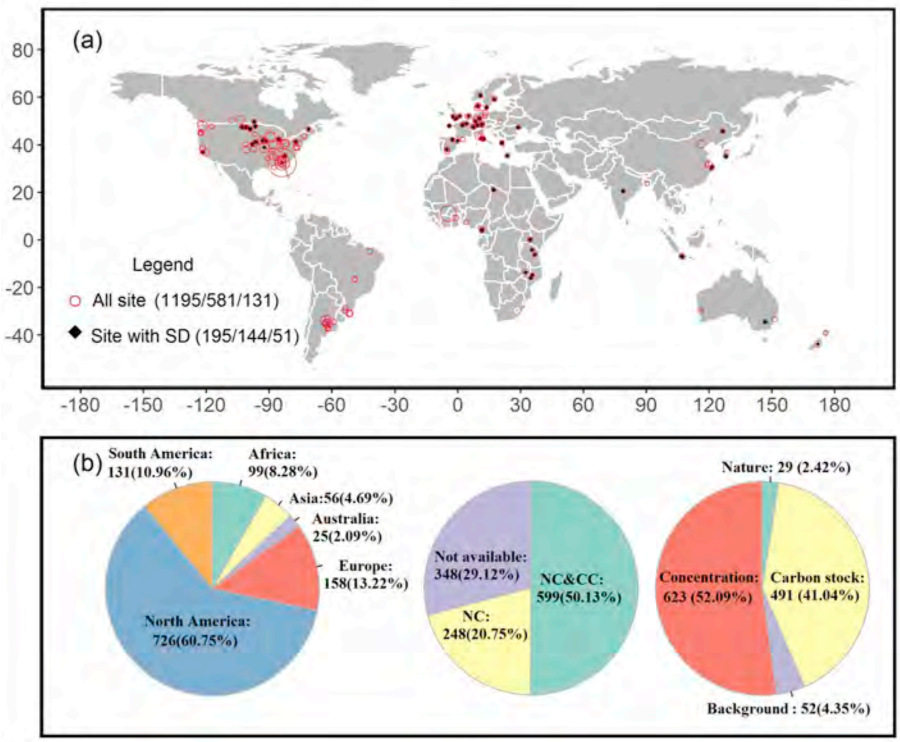


Fig. 1. (a) Site spatial distribution from cover crop studies compiled in the meta-analysis, which included 1195 pairwise comparisons from 131 papers and represented 581 independent experiments (All sites); of those, 195 comparisons from 51 papers reported standard deviations (Sites with SD), representing 144 independent experiments. (b) Breakdown of study comparisons by: (left) continent, (center) whether bulk density data was reported, and (right) type of carbon data reported. NC&CC = bulk density was reported for both control and cover crops; NC = bulk density was only reported for the control; Not available = bulk density was not reported; Carbon stock = carbon was reported as mass of carbon per area [M L^{-2}]; concentration = carbon was reported as mass of carbon per mass of soil [M M^{-1}]; background = background soil carbon was reported; nature = carbon concentration was compared against nearby soils under natural vegetation.

2.2. Data processing

Data were digitized from tables directly and from figures using the software *Data Thief* (version III, http://datathief.org/). After digitization, meta (background) data were extracted from publications and coded as being one of 38 types of information. For more details on these categories, please refer to Stewart et al. (2018).

Whenever latitude and longitude were not reported, we estimated that information based on the name and location of the site, using the website https://www.findlatitudeandlongitude.com/http://www.findlatitudeandlongitude.com/. Elevation was identified by latitude and longitude using https://www.freemaptools.com/elevation-finder.htm/http://www.freemaptools.com/elevation-finder.htm/ whenever it was not reported in the original source.

We applied several quality control tests to verify data quality. First, we checked original source material for all data reported in the meta-analyses/reviews to make sure that the information was translated correctly. Next, we used the following equation to convert any data reported as soil organic matter (SOM) to SOC (Gattinger et al., 2012; Nelson and Sommers, 1996).

$$SOC = SOM/1.72 \tag{1}$$

Once all data were collected and converted, we mapped sites by country to confirm that latitude and longitude were collected correctly.

The number of replications and standard deviation (SD) values were recorded from publications when possible. Most studies reported replication number (1076 out of 1195). When applicable, SD values were calculated from reported standard error (SE), coefficient of variation (CV), or confidence interval (CI) values according to the following equations:

$$SD = SE \times \sqrt{n} \tag{2}$$

$$SD = CV \times mean \tag{3}$$

$$SD = |CI - mean|/(2Z_{\alpha/2}) \times \sqrt{n} \tag{4}$$

where $Z_{\alpha/2} = 1.96$ when the significance level $\alpha = 0.05$.

For SOC data, only 195 out of 1195 comparisons (16%) reported SD information (Fig. 1). Considering only studies that reported SD, SE, CV, or CI, we determined that the ratio of the SD to the mean did not follow a normal distribution. We therefore used a bootstrap approach to resample the SD/mean with replacement 10,000 times. The mean ratio of SD/mean was 0.12 (with confidence interval of 0.10–0.14) for the control data and 0.13 (with confidence interval of 0.11–0.15) for the CC data (Fig. A4), which closely resembled findings of Luo et al. (2006). We thus assigned SD as 0.12 of the reported mean for all control data that did not report SD values, and as 0.13 of the reported mean for all CC data that did not report the SD values.

Only 599 comparisons (50.1%) reported BD for both the NC controls and CC treatments (Fig. 1b). A regression analysis on those comparisons showed that the BD of the CC treatments was highly correlated with the control BD (adjusted $R^2 = 0.92$; Fig. S5a). We thus estimated soil BD of CCs using the control BD for the 248 comparisons (20.80%, Fig. 1b) that only reported background BD. Finally, our analysis showed that BD was moderately correlated with $SOC_{\%}$ (adjusted $R^2 = 0.47$; Fig. A5b), so we estimated BD based on the SOC information for the 348 comparisons (29.10%, Fig. A5a) that did not include any BD measurements.

Carbon stocks were reported for 491 studies (41.0%), while carbon concentration was reported for 623 comparisons (52.1%; Fig. 1b). Converting carbon concentration values into carbon stock required the following equation:

$$SOC_{stock} = SOC_{\%} \times h \times BD \tag{5}$$

where SOC_{stock} represents soil organic carbon stock [M L^{-2}], $SOC_{\%}$ represents soil organic concentration [M M^{-1}], h represents the soil sampling depth [L], and BD represents soil bulk density [M L^{-3}].

2.3. Data analysis

After calculating SOC_{stock} for all observations, the carbon sequestration rate, C_{rate} [M L^{-2} T^{-1}], was calculated as:

$$C_{rate} = (SOC_{CC} - SOC_{NC})/y \quad (6)$$

where SOC_{CC} and SOC_{NC} are the respective soil carbon stocks [M L^{-2}] under CCs and NC controls, and y represents time after CC implementation [T]. All C_{rate} values were compiled together to determine the global mean rate of SOC change due to CCs. First, the normality of the distribution of C_{rate} values was tested using the Shapiro-Wilk test. Since that analysis suggested that data were not normally distributed ($p <$ 0.05), we resampled using a bootstrapping approach to generate a normal distribution from data (Fig. A6).

The overall mean C_{rate} value [M L^{-2} T^{-1}] determined using this approach was then used to estimate total carbon sequestration, $C_{sequestration}$ [M T^{-1}], associated with incorporating CCs into agricultural rotations:

$$C_{sequestration} = C_{rate} \text{ x } A \text{ x } f \quad (7)$$

where A represents the global cropland area [L^2], and f is the proportion of cropland being managed with CCs [L^2 L^{-2}]. A was estimated as the mean of values reported in six studies found in the literature. Three values of f were analyzed: 1) $f = 0.08$, which represented the approximate amount of CCs currently planted in the United States (Tellatin and Myers, 2018); 2) $f = 0.15$, which represented the proportion of CCs that might be planted by 2040 assuming similar rates of adoption (Poeplau and Don, 2015; Tellatin and Myers, 2018); and 3) $f = 1.0$, which represented that total possible carbon that could be sequestered if all cropland was managed using CCs. For more details on how values for C_{rate}, A and f were selected, please refer to the Supplemental Information.

We also computed a response ratio (RR_{SOC}) for each pairwise comparison between SOC stocks in the CC treatment versus NC control:

$$RR_{SOC} = \ln (SOC_{CC} / SOC_{NC}) \quad (8)$$

as well as response ratios (RR_x) for all pairwise values reported for 32 different soil properties and agronomic variables:

$$RR_x = \ln (x_{cc}/x_{nc}) \quad (9)$$

where x_{cc} is the parameter value in the CC treatments and x_{nc} is the parameter value in the NC controls. Specific parameters analyzed included: 1) BD; 2) aggregate stability; 3) porosity; 4) penetration resistance; 5) infiltration rates; 6) saturated hydraulic conductivity; 7) erosion; 8) runoff; 9) leaching; 10) soil temperature; 11) soil water content; 12) available water holding capacity; 13) soil nitrogen; 14) phosphorus; 15) potassium; 16) pH; 17) cation exchange capacity; 18) electricity conductivity; 19) base saturation; 20) soil fauna; 21) fungal indicators; 22) other microbial indicators; 23) enzymatic assays; 24) mineralizable carbon; 25) mineralizable nitrogen; 26) N_2O gas emission; 27) burst test CO_2; 28) field-measured soil CO_2 efflux; 29) microbial biomass carbon; 30) microbial biomass nitrogen; 31) cash crop biomass not including yield (e.g., leaf, steam, root biomass); and 32) cash crop yield.

We next divided data into different categories to explore how climate, cash crop type(s), soil texture, CC type(s), and soil depth affected SOC dynamics under CCs. The climate type of each site was identified based on the Koppen climate classification (Kottek et al., 2006), with relevant categories including tropical, arid, temperate, and snowy climates. Cash crops were grouped into corn, soybean, wheat, other monoculture, corn-soybean rotation, corn-wheat-soybean rotation, and other rotations of more than two cash crops. The CCs were grouped as broadleaf, grass, legume, mixtures of two legumes, mixtures of a legume and a grass, and other mixtures of more than two CCs. Soil texture was grouped as being coarse (sand, loamy sand, and sandy loam), medium (sandy clay loam, loam, silt loam, and silt), or fine (clay, sandy clay, clay loam, silty clay, and silty clay loam) based on The Cornell Framework of Soil Health Manual (Moebius-Clune et al., 2016). Soil sampling depths were grouped into surface, i.e., ≤30 cm, and subsurface, i.e., >30 cm, depth increments (Fig. A2).

After grouping data, SOC_{stock} and C_{rate} values were compiled for the CC versus NC data in each category (reported here as means and SDs). RR_{SOC} values were used in a meta-analysis to compare SOC changes due to CCs for each category. Note that for ease of interpretation, RR_{SOC} values from the meta-analysis are presented as percent change, calculated as 100 * [e^{RRSOC} – 1].

Simple linear regression was used to analyze the relationship between RR_{SOC} and RR_x for the 32 different soil/agronomic variables recorded from the studies. Simple linear regression was also applied to explore the relationship between RR_{SOC} and many other environmental conditions. We specifically analyzed 16 environmental variables: 1) elevation; 2) latitude; 3) carbon to nitrogen ratio of CC biomass; 4) CC biomass returned to the field; 5) soil background total carbon content; 6) soil background pH; 7) clay content; 8) silt content; 9) sand content; 10) soil background BD; 11) duration of CCs; 12) years after CC implementation; 13) mean annual precipitation between 1960 and 2015 (MAP); 14) annual precipitation during the study period (Pannual); 15) mean annual temperature between 1960 and 2015 (MAT); and 16) annual temperature during the study period (Tannual). Note that the number of samples differed between different soil properties, agronomic variables, and environmental factors. Adjusted R^2 was used to evaluate the goodness of fit for the linear models used in this analysis.

All statistics were conducted using R (version 3.5.1, R Core Team, 2014). The meta-analysis was applied using '*metafor*' package, and the simple linear regressions were applied under R using the linear model (*lm*) function. We used '*ggmap*' to generate site spatial distribution (Kahle and Wickham, 2013).

3. Results

3.1. Carbon stocks under CCs versus NC controls

Cropland had greater SOC stocks when managed with CCs as compared to NC controls (Fig. 2). Soil carbon stocks were greater in regions characterized by a snowy climate (i.e., ≥1 month with average air temperature < -3 °C; Kottek et al., 2006) compared to temperate and tropic regions. Soil carbon stocks were similar across cash crop systems, except for the corn-wheat-soybean rotation, which had the largest SOC stocks. Likewise, medium-textured soils had the greatest SOC stocks, with mean SOC values of 39 Mg ha^{-1} under CC and 37 Mg ha^{-1} under NC. Soil carbon sequestration rates were similar across climates and cash crop systems; however, fine-textured soils showed larger carbon sequestration rates than the medium- and coarse-textured soils, and surface soils had larger sequestration rates than subsurface layers.

A meta-analysis was applied to analyze RR_{SOC} for different categories (Fig. 3). When cover crops were included in rotations, SOC increased by an average of 15.5% across all experiments (Fig. 3a); however, including only those experiments that reported SD in the original publication showed a 30% increase in SOC under CCs (Fig. 3b). When separated by categories, similar trends can be identified from both methods. Soil carbon stocks were significantly greater in CC treatments versus NC controls for all soil texture groups, but fine-textured soils showed larger increases than medium- and coarse-textured soils. Legume CCs, mixtures of two legumes, and mixtures of more than two other CCs showed significant increases in SOC relative to controls, while grass CCs or grass-legume mixtures did not show significant changes. Corn, wheat, and vegetables all showed significant carbon increases while soybeans, corn-soybean rotations, and corn-wheat-soybean rotations did not. Other rotations of two or more crops (i.e., those that did not include corn and soybeans) also showed significant SOC increases when using CCs. When separated into different soil sampling depths, surface soils showed

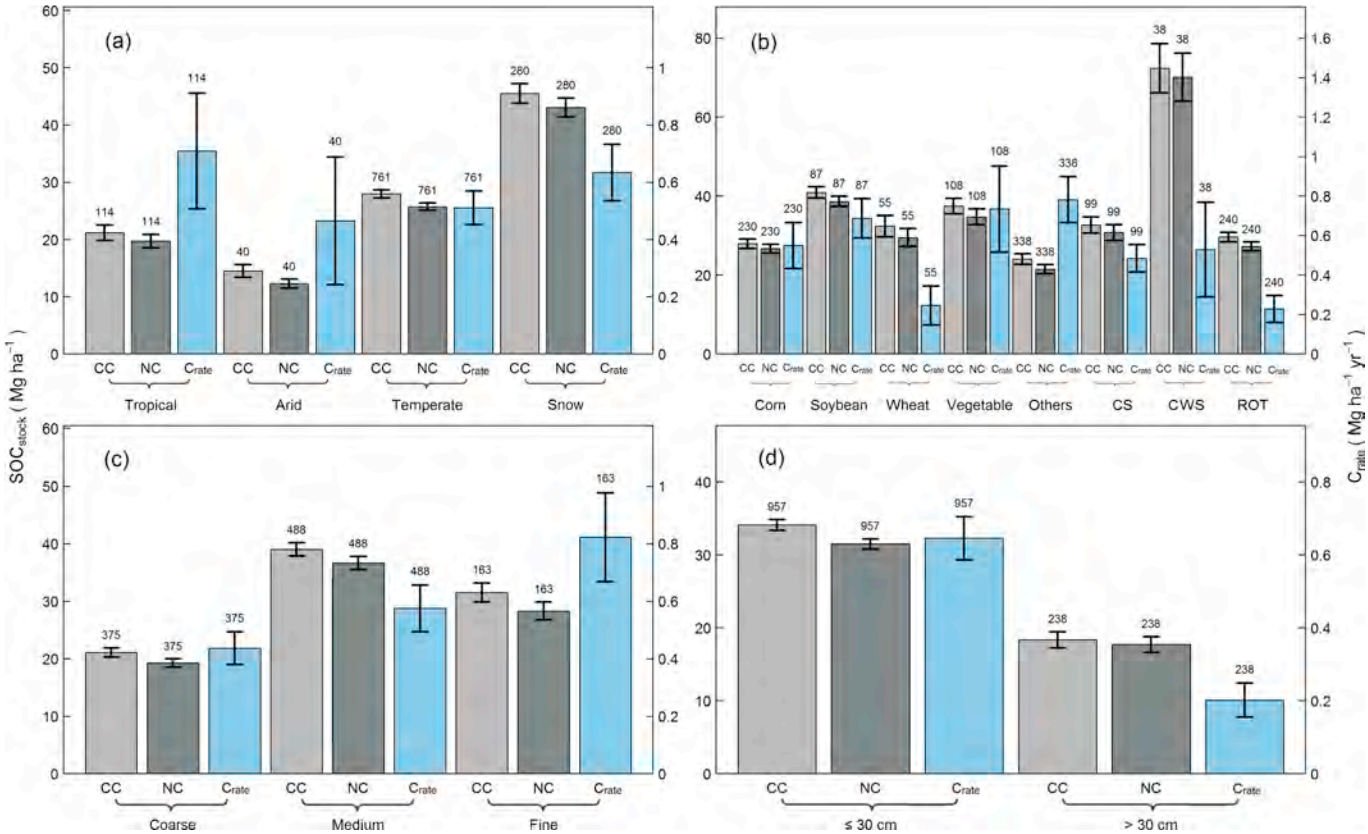


Fig. 2. Soil carbon stocks, SOC_{stock} (Mg ha^{-1}, y-axis on the left) and carbon sequestration rates, C_{rate} (Mg ha^{-1} yr^{-1}, y-axis on the right) in croplands under different (a) climatic regions, (b) cash crop types, (c) soil texture groups, and (d) soil sampling depth increments. Black bars represent standard errors and numbers above the bars are the number of pairwise comparisons. CC: cover crop; NC: no cover crop (i.e., control); C_{rate}: soil carbon sequestration rate; CS: rotation of corn and soybean; CWS: rotation of corn, soybean, and wheat; Other: other monoculture cash crop; ROT: other cash crop rotation.

significant carbon increases when CCs were used, whereas subsurface soils did not.

3.2. Carbon sequestration potential of CCs

The areal extent of global cropland was constrained from six different studies as $A = 1960 \pm 680$ Mha (Table A1). Using the overall mean C_{rate} value of 0.56 Mg ha^{-1} yr^{-1} (Fig. A6b), average global $C_{sequestration}$ values ranged from 0.09 ± 0.03 Pg yr^{-1} (for $f = 0.08$, which represented the current proportion of acreage managed using CCs in the United States) to 0.16 ± 0.06 Pg yr^{-1} (for $f = 0.15$) to 1.1 ± 0.4 Pg yr^{-1} (for $f = 1.0$, which represented all cropland being managed using CCs). Carbon sequestration values thus represented 0.5% on the low end to 16% on the high end of current fossil fuel emissions (Table 1).

3.3. Interactions between SOC changes and soil/agronomic variables

Other than SOC, many other physical, chemical, and biological soil properties can be affected by including CCs as a rotation. Linear regression between the RR_{SOC} (Equation (6)) and RR_x (Equation (7)) showed that SOC changes were negatively correlated with changes in BD, erosion, runoff, soil water content, and burst test CO_2 emissions, which are determined by rewetting air-dried soil to 50% water holding capacity and then measuring soil respiration for at least 2 h (Franzluebbers et al., 2000). SOC changes were positively correlated with changes in soil aggregation, porosity, soil nitrogen, phosphorus, potassium, cation exchange capacity, electrical conductivity, enzymatic assays, mineralizable carbon, mineralizable nitrogen, microbial biomass nitrogen, and biomass of cash crop yield ($p < 0.05$; Fig. 4). Correlations were highest between SOC and runoff (adjusted $R^2 = 0.86$), erosion (adjusted $R^2 = 0.47$), mineralizable carbon (adjusted $R^2 = 0.48$), mineralizable nitrogen (adjusted $R^2 = 0.22$), and soil nitrogen (adjusted $R^2 = 0.21$). SOC was not significantly correlated with other soil properties.

3.4. Interactions between soil carbon changes and environmental factors

Linear regression was applied to analyze the relationship between SOC response from CCs and sixteen other environmental variables (Fig. 5). Results showed that soil carbon stock had significant and positive correlations with annual temperature, years after CC implementation, and CC duration ($p < 0.05$ and slope > 0, labeled as orange in Fig. 5), and significant but negative correlations with latitude and soil background carbon stocks ($p < 0.05$ and slope < 0, labeled as blue in Fig. 5). Individual factors explained little of the total variability in SOC, with only annual temperature and soil background carbon having adjusted R^2 values > 0.05. All other factors were not significantly correlated with SOC changes caused by CCs ($p \geq 0.05$, labeled as pink in Fig. 5). It should be noted that CC biomass ($p = 0.15$) and CC biomass C:N ratio ($p = 0.24$) had the highest adjusted R^2 relationships with SOC changes, even though relationships were not significant. This result may be due to the small number of samples reported for biomass ($n = 28$) and C:N ratios ($n = 22$); which are much fewer than the number of observations reported for other factors.

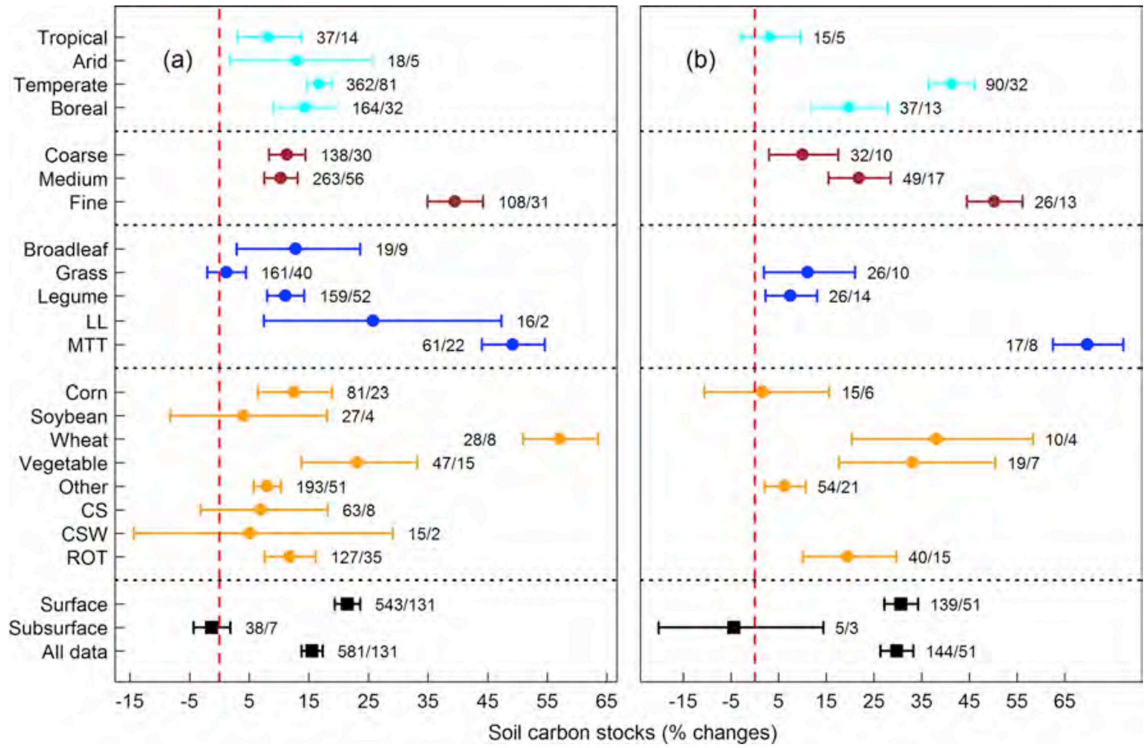


Fig. 3. Meta-analysis results showing change in soil carbon stocks due to implementation of cover crops in four climatic regions, three soil texture groups, six cover crop types, eight cash crop types, and two soil sampling depths. Panel (a) presents results from all comparisons; panel (b) presents meta-analysis results from comparisons that reported sufficient information to calculate standard deviations (SD). Circles or squares with error bars represent the overall mean RR_{SOC} values ± 95% confidence intervals (scaled to %). Categories whose 95% confidence intervals do not cross 0 (represented by the vertical red lines) have significant differences between cover crop treatments and controls. The number of experiments followed by the number of studies are listed for each category. Surface = samples collected from ≤30 cm depth; subsurface = samples collected from depths >30 cm. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

Table 1
Estimates of global soil carbon sequestration potential, $C_{sequestration}$, and % of current fossil fuel emissions, assuming three different levels of cover crop adoption: $f = 0.08$; $f = 0.15$; and $f = 1.0$. Note that C_{rate} was assumed to equal 0.56 Mg ha^{-1} yr^{-1}, A was assumed to equal 1960 ± 680 Mha, and current carbon emissions from fossil fuels were assumed to range from 9.0 to 11.0 Pg yr^{-1} (Jackson et al., 2017). The $C_{sequestration}$ values represent near-surface (≤30 cm depth) soils.

	$f = 0.08$	$f = 0.15$	$f = 1.0$
$C_{sequestration}$ (Pg yr^{-1}) = C_{rate} x A x f	0.09 ± 0.03	0.16 ± 0.06	1.1 ± 0.4
% of current fossil fuel emissions	0.52–1.3	0.98–2.5	6.5–16

4. Discussion

4.1. SOC changes due to cover crop usage

In this study we compiled data from 131 publications that compared SOC concentrations when CCs were included in rotations versus SOC in NC controls. In total, 1195 comparisons were included (Fig. 1). We first examined how different categorical environmental factors influenced SOC stocks and sequestration rates, specifically examining the role of climate, CC type, cash crop type, soil texture, and soil sampling depth (Fig. 2). We then used a meta-analysis to quantify SOC changes under CCs using RR_{SOC} as the response variable (Fig. 3). We also performed linear regressions to examine correlations between changes in SOC and continuous environmental factors (e.g., temperature, precipitation, CC biomass; Figs. 4) and 32 other soil/agronomic variables (Fig. 5). While previous studies examined mechanisms of SOC changes under CCs at various scales (Aguilera et al., 2013; Olson et al., 2014; Schmidt et al., 2017; Stavi et al., 2012; Tian et al., 2018; Poeplau and Don, 2015), this study for the first time analyzed how SOC dynamics correlated with other soil/agronomic variables.

Our analysis compared SOC stocks for CC treatments versus NC controls both in terms of absolute magnitudes (Fig. 2) and relative changes (i.e., Fig. 3). The former approach allowed us to compute the rate of carbon change, C_{rate}, whereas the latter quantified the total change, regardless of time elapsed. Results from these two approaches agreed for many factors, yet some discrepancies arose. As an example, tropical and snowy regions had higher C_{rate} values than temperate and arid climates (Fig. 2), while temperate regions had overall higher RR_{SOC} values than the tropics (Fig. 3). Of all cash crop rotations, wheat monocultures had the highest RR_{SOC} values but relatively low results for C_{rate}. These disparities may reflect variations in time of CC usage, as C_{rate} can change through time (Blanco-Canqui et al., 2015). Differences in sampling depths between studies could also be a factor, as approximately 20% of all observations came from deeper than 30 cm, where SOC changes were negligible (Figs. 2 and 3).

The meta-analysis showed that incorporation of CCs into rotations caused a mean SOC increase of 15.5% (when all comparisons were included) or 30.0% (when only comparisons with SD values were included; Fig. 3). Here we note that including calculated SD data is commonly used in meta-analyses that examine SOC dynamics (Aguilera et al., 2013; Alvarez et al., 2017; Don et al., 2011; Lesur-Dumoulin et al., 2017; Luo et al., 2006; Poeplau and Don, 2015; Sileshi, 2009; Tian et al., 2018; Tonitto et al., 2006). While the approach including all data provided a more conservative estimate of overall SOC changes due to CCs in the present study, it also resulted in higher estimates of SOC change under certain conditions (Fig. 3). The 15.5% mean change in SOC translates to a carbon sequestration value of C_{rate} = 0.56 Mg ha^{-1} yr^{-1} (Fig. A6), while the 30% mean change in SOC translates to C_{rate} = 1.05 Mg ha^{-1} yr^{-1}. Historical SOC losses when converting from natural

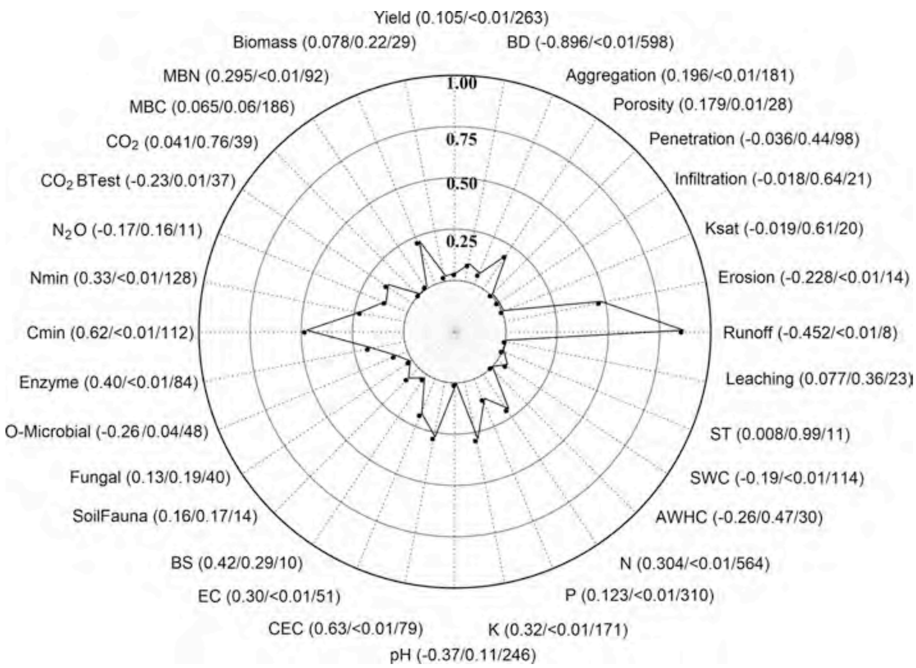


Fig. 4. Correlations between SOC changes due to cover crops and changes in other physical, chemical, biological properties of soil, cash crop biomass, and yield. The dots indicate adjusted R^2 values (each concentric circle represents an R^2 increment of 0.25). The values in the parentheses represent the slope/p-value of slope/number of samples of regression between the SOC response ratio (RR_{SOC}) and response of specific indicators (RR_x). NS means that the correlation was not significant ($p < 0.05$). Acronyms: BD – bulk density, Ksat – saturated hydraulic conductivity, ST – soil temperature, SWC – soil water content, AWHC – available water holding capacity, N – soil nitrogen, P – soil phosphorus, K – soil potassium, CEC – cation exchange capacity, EC – electricity conductivity, BS – base saturation, O-Microbial – other microbial indicator, Cmin – mineralizable carbon, Nmin – mineralizable nitrogen, CO_2BTest – CO_2 burst test, CO_2 – field-measured soil CO_2 efflux, MBC – microbial biomass carbon, MBN – microbial biomass nitrogen, Biomass – cash crop biomass not including yield (e.g., leaf, steam, root biomass).

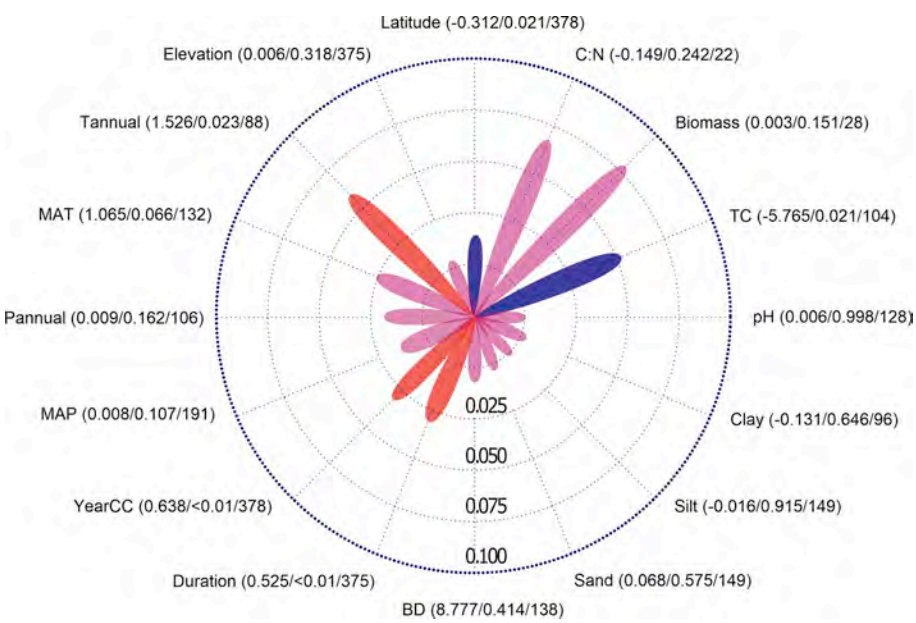


Fig. 5. Adjusted R^2 values from simple linear regression between SOC change under cover crops and interactions with 16 environmental factors. The blue color represents a significant negative correlation between a variable and the SOC response ratio (RR_{SOC}); the orange color represents a significant positive correlation between a variable and RR_{SOC}; and the pink color indicates no significant relationship between a variable and RR_{SOC} ($p > 0.05$). The adjusted R^2 values of 0, 0.025, 0.050, 0.075, and 0.100 are represented by the concentric circles. The values in the parenthesis represents the slope/p-value of slope/number of samples of the regression. Acronyms: MAP – mean annual temperature between 1960 and 2015, Pannual – annual precipitation during study period, MAT – mean annual temperature between 1960 and 2015, Tannual – annual temperature during study period, YearCC – years after cover crop implementation, BD – bulk density, TC – total soil carbon, Biomass – cover crop biomass, C:N – carbon to nitrogen ratio of cover crop biomass. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

vegetation to cropland were estimated to range from −25% to −36% (Don et al., 2011; Poeplau and Don, 2015), so SOC gains under CCs (assuming a 15.5% increase) may recover approximately one-quarter to one-third of this overall SOC loss.

The C_{rate} value of 0.56 Mg ha^{-1} yr^{-1} equals approximately 0.06–0.12 Pg yr^{-1} of sequestered carbon, under the assumption that approximately 8% of cropland is currently managed using CCs worldwide (Table 1). If all cropland were to adopt cover crops (i.e., $f = 1.0$), the $C_{sequestration}$ potential could be as large as 1.5 Pg yr^{-1}. This latter amount would account for 13–16% of annual carbon emissions from fossil fuel combustion, or approximately 1/2 of the terrestrial carbon sink (Stocker et al., 2013). Planting anywhere near 100% of global cropland with CCs is impractical due to numerous reasons, including expenses associated with planting and managing CCs (Zhou et al., 2017), potential water limitations (Reese et al., 2014), lack of suitable growing windows in certain crop rotations (Clark, 2008), and disruptions to cash crop planting and harvesting times. However, this number does provide an upper limit for the amount of atmospheric carbon that may be sequestered via CCs. At the more realistic adoption level of $f = 0.15$, carbon sequestered by CCs could represent ∼1–2% of current yearly emissions from fossil fuel combustion.

4.2. Correlations between SOC accumulation and other factors

Our results also showed that different environmental and management factors affected SOC accumulation. For instance, SOC accumulation was found to vary by soil textural class. Medium-textured soils had the highest overall SOC stocks, including both CC and NC data (Fig. 2). Fine-textured soils resulted in the highest carbon increase after introduction of CCs, while medium and coarse-textured soils had lowest SOC increases with CCs (Fig. 3). These results could partially reflect study location, with many studies coming from the Midwestern U.S. (Fig. A7), a region that is characterized by organic-rich, medium-textured soils (Nachtergaele et al., 2010; Reynolds et al., 2000; Scharlemann et al., 2014). Having relatively high SOC stocks under NC conditions likely caused lower carbon sequestration rates relative to fine-textured soils that had relatively low initial SOC stocks.

These findings may also reflect the ability of some fine-textured soils to provide physical protection to SOC (Hassink et al., 1997; Zinn et al., 2005). Clays and silt-sized particles are more likely to form stable aggregates than sands (Gyawali and Stewart, 2019; Sollins et al., 1996), which can protect SOC by isolating it from microbial access (Puget et al., 2000; Six et al., 2004). Clay- and silt-sized particles provide the majority of sorbent surface area within soils and thereby affect SOC sorption and availability (Sollins et al., 1996). Clay particles can also alter microbial metabolism pathways and enzymatic activity, both of which can affect SOC (Huang et al., 1986).

At the same time, factors such as physical heterogeneity and root distributions can be more important than texture (Schmidt et al., 2011), and may help explain seemingly contradictory results in which coarse-textured soils have greater SOC changes than fine-textured soils. For instance, a meta-analysis in the Pampas region of Argentina found that SOC increased more in coarse-textured (+9%) than in fine-textured (+4%) soils (Alvarez et al., 2017). That particular study differed from ours in several other key aspects: 1) the two analyses included different number of studies; 2) Alvarez et al. (2017) focused on the Pampas region, which has a temperate climate, whereas our analysis collected results from many parts of the globe and considered four different climate types with associated effects on plant productivity; and 3) we separated soils into three texture groups (i.e., coarse, medium, and fine) following the Cornell Framework of Soil Health manual (Moebius-Clune et al., 2016), while Alvarez et al. (2017) grouped soils into two groups (i.e., coarse and fine) based on soil family information.

Our analysis also showed that SOC change under CCs was negatively correlated with total soil carbon content. This result may reflect an upper limit in the amount of carbon that can be stored in a soil matrix given the surrounding environmental conditions (e.g., soil temperature and soil moisture; Clark, 2008). SOC increases are often greatest in formerly degraded soils, e.g., soils that have experienced high erosion (Berhe et al., 2007). Therefore, condition of the physical substrate may be an important factor influencing SOC dynamics.

Runoff (adjusted $R^2 = 0.86$), mineralizable carbon (adjusted $R^2 = 0.48$), erosion (adjusted $R^2 = 0.47$), potassium (adjusted $R^2 = 0.29$), CEC (adjusted $R^2 = 0.28$), and mineralizable nitrogen (adjusted $R^2 = 0.22$) were indicators that correlated best with SOC change under CCs. SOC increases after CC introduction were associated with significant decreases in runoff and erosion (Fig. 4). Lower rates of runoff and erosion can reduce SOC losses from the field and thereby form a positive feedback (Berhe et al., 2007; Kaye and Quemada, 2017; Meyer et al., 1997). Likewise, root biomass, rhizo-deposits, and soil microbes are all important sources of SOC (Kutsch et al., 2009) and greater mass and activity of these groups under CCs may help to explain significant positive relationships seen with mineralizable carbon, mineralizable nitrogen, and potassium CCs (Araujo et al., 2012; Balakrishna et al., 2017). Enhanced soil nitrogen, phosphorus, and potassium content may have helped increase CC biomass by providing ample fertility from deeper within the soil profile. This process is especially important in coarse-textured or well-structured soils, where nutrients can experience rapid transport through the soil profile (Tremblay et al., 2012; Zhu et al., 2016).

The results in this study showed that soil carbon stock changes after CCs had significant positive correlations with annual temperature and precipitation (Fig. 5), matching observations reported in other studies. For example, in semiarid areas, SOC can increase when using CCs, but limited biomass production due to low precipitation means that accumulation can take longer than in wetter climates (Blanco-Canqui et al., 2013). At the same time, temperature is a key factor controlling plant growth and is often positively correlated with plant productivity (Churkina and Running, 1998), yet higher temperatures also typically cause higher decomposition rates (Lloyd and Taylor, 1994). The positive relationship between annual temperature and SOC changes after CCs indicates that rate of carbon accumulation from higher plant productivity exceeded any increases in decomposition rates. Our analysis also established that latitude was negatively correlated to changes in SOC_{stock} after cover cropping. Higher latitude locations have short growing seasons, which can reduce CC growth rates and limit plant residue inputs into soils (Mirsky et al., 2017).

When regressed against SOC changes, CC biomass and C:N ratios had relatively high R^2 values, with CC biomass having a positive correlation with SOC and C:N ratio having a negative one. However, these relationships were not significant (Fig. 5). Previous work has found that using CCs during fallow seasons can increase carbon returned to the soil via residue (O'Dea et al., 2013), with the rate of residue return scaling with aboveground biomass (Kuo et al., 1997). Other studies have also established that greater aboveground CC biomass can translate to more root biomass, enhanced rhizo-deposition, and greater diversity of soil microbes (Araujo et al., 2012; Balakrishna et al., 2017). All of these factors can increase the pool of belowground carbon (Kutsch et al., 2009). We thus speculate that the non-significant relationships determined by our study were likely influenced by the limited number of observations reported for each factor ($N = 28$ for biomass and $N = 22$ for C:N ratio).

Our analysis did reveal that SOC changes varied between different types of CC species (Fig. 3), which may indirectly reflect differences in CC biomass and C:N ratios. Specifically, legume and mixed CCs both caused significant SOC increases, while grass species did not. One possible reason for this particular outcome is that grass CCs often have high C:N ratios, which can increase the amount of time needed for biomass to be converted into SOC (Jani et al., 2016; Kaye and Quemada, 2017; O'Dea et al., 2013). High C:N ratios can also cause more carbon to be lost as respiration versus stabilized in soil, due to lower carbon use efficiency by decomposer organisms (Manzoni et al., 2012; 2008).

Perhaps as a result of these tradeoffs between maximum biomass and optimum C:N ratios in single-species CCs, mixtures of CCs provided the greatest overall increases in SOC. Similar results were reported by other studies; for example, Faé et al. (2009) found that CC mixtures led to greater SOC increases compared to single species CCs. Zhou et al. (2019) showed that increased plant species richness can reduce the litter C:N ratio and thus promote SOC accumulation. Likewise, Stavi et al. (2012) found that CC mixtures led to greater SOC concentrations (19.4 g kg^{-1}) than single-species CCs (15.9–17.6 g kg^{-1}) for farmland in Ohio. Using CC mixtures may therefore offer a reliable strategy for increasing SOC.

4.3. Comparison with other conservation agriculture practices

Cover crops represent only one conservation strategy for cropland management. It is well recognized that agro-forest systems usually have higher SOC compared with nearby cropland or pastures (Shi et al., 2018). Conservation tillage, including no-till, is another practice that can affect SOC concentrations in soil. No-till management often increases SOC near the soil surface ($\leq$30 cm) compared with conventional tillage systems (Cooper et al., 2016). However, the increase in SOC under no-till has been challenged by recent analyses (Baker et al., 2007; Luo et al., 2010), which showed that SOC increases in soils can be offset by decreases in the subsurface layer. Further, few studies have performed factorial comparisons of cover cropping and tillage practices, limiting our ability to analyze interactions between these management strategies. Other cropland conservation management techniques include alley cropping and inter-seeding, diverse crop rotations, forage and biomass planting, contour farming, mulching, riparian herbaceous cover, and contour buffer strips. The ability of these other cropland conservation management strategies to increase SOC and sequester atmospheric carbon should be investigated in future studies.

4.4. Limitations and perspective

Most existing comparisons reported soil carbon concentration (i.e., $SOC_{\%}$) changes without reporting soil BD. Since BD is necessary to calculate SOC stocks, we had to estimate BD for many instances based on correlations developed using reported values (Fig. A5). As this process added additional uncertainty to the dataset, we recommend that soil carbon stock and BD should be reported in the future studies. Likewise, it is important that studies report SD values whenever possible.

Substantially fewer data were available for the subsurface layers (>30 cm) compared with the surface layers (≤30 cm). Our results suggested that CCs may not change SOC concentrations in subsurface soil layers; however, this conclusion may be due to the small sample size (only 38 experiments from 7 studies and only 5 experiments from 3 studies have SD information; Fig. 3). Because of this uncertainty, future experiments should strive to include samples from subsurface layers, which should help evaluate subsoil benefits of CCs in the future.

Comparisons collected in this study covered a wide time period (1960–2014) and included samples from various depths and sampling increments. We did not attempt to account for any sampling differences in this study. In addition, most comparisons were reported after less than 5 years of data collection, even though CC effects on SOC are often not detectable in the first few years after establishment, due to high spatial field variability or soil heterogeneity (Olson et al., 2014; Poeplau and Don, 2015). Future CC experiments should continue to collect data for mid- (e.g., 5–10 years) and long-term (e.g., >10 years) periods to the extent possible. Longer-term data are particularly important to understand maximum sequestration potentials in soils. The ~0.1–~1 Mg ha^{-1} yr^{-1} carbon sequestration rates found in this study (Fig. 3) and in others (e.g., Alvarez et al., 2017) suggest that soils can sequester 10–100 Mg ha^{-1} $century^{-1}$. However, those values approach or exceed existing carbon stocks in many existing systems (Fig. 2), making it probable that sequestration rates will diminish through time (Blanco-Canqui et al., 2015).

Finally, our estimated potential carbon sequestration values relied on a single mean value for SOC change that translated to all land planted with CCs. These estimates therefore represent crude approximations, as our analysis showed that SOC changes from CCs vary depending on numerous environmental and management factors. Future investigations should therefore seek to refine the initial estimates provided here, for example by incorporating global datasets for climate (Kottek et al., 2006), soil texture (Nachtergaele et al., 2010; Reynolds et al., 2000) and cropping systems (Thenkabail et al., 2013). Even so, it is important to consider that our analysis showed that CCs increase SOC concentrations in nearly all conditions. This general finding emphasizes the importance of efforts that encourage farmers to adopt these practices more widely, despite uncertainties regarding the exact amount of carbon that can be sequestered as a result.

5. Conclusion

In this study, we collected 1195 SOC comparisons between CC treatments and NC controls from 131 studies. Using this data, we conducted a meta-analysis to explore SOC changes under CCs and the interactions between SOC changes, soil properties, and environmental factors. Altogether, cover cropping caused a 15.5% increase in SOC (95% confidence interval of 13.8%–17.3%) in near-surface soils (i.e., ≤30 cm depth), indicating that inclusion of CCs into agricultural rotations can potentially increase soil carbon sequestration. As an example, under the reasonable assumption that 15% of worldwide cropland was to be managed using CCs, approximately 0.16 ± 0.06 Pg of carbon could be sequestered per year. These values represent ~1–2% of yearly emissions from fossil fuel combustion.

Subsurface soils (i.e., >30 cm) showed no significant change in SOC, possibly due to the limited number of samples reported for the subsurface. A regression analysis revealed that SOC increases were correlated with runoff, erosion, mineralizable carbon, mineralizable nitrogen, and soil nitrogen. Surrounding environmental conditions also affected SOC changes under CCs, but only explained a small amount of total variability. Similarly, CC biomass was positively correlated and C:N ratio was negatively correlated with changes in SOC, though these relationships were not significant. Based on sources of uncertainty identified in this study, we propose that future CCs studies should: 1) sample both the near surface (e.g., 0–10 cm) and the subsurface (e.g., 40–50 cm) layers of the soil profile; 2) maintain experiments over mid- (e.g., 5–10 year) to long-term (>10 years) periods; and 3) always report soil BD, so that SOC stock can be appropriately estimated.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This material is based upon work supported by the Natural Resources Conservation Service, U.S. Department of Agriculture, under Conservation Innovation Grant 69-3A75-14-260. Jinshi Jian was partially supported by the U.S. Department of Energy, Office of Science, Biological and Environmental Research as part of the Terrestrial Ecosystem Sciences Program. Funding for this work was also provided in part by the Virginia Agricultural Experiment Station and the Hatch program of the National Institute of Food and Agriculture, U.S. Department of Agriculture.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.soilbio.2020.107735.

References

Manzoni, S., Taylor, P., Richter, A., Porporato, A., Ågren, G.I., 2012. Environmental and stoichiometric controls on microbial carbon-use efficiency in soils. New Phytologist 196, 79–91. https://doi.org/10.1111/J.1469-8137.2012.04225.X@10.1002/(ISSN)1469-8137(CAT)VIRTUALISSUES(VI)ECOLOGICALSTOICHIOMETRYGLOBALCHANGE.

Abdollahi, L., Munkholm, L.J., 2014. Tillage system and cover crop effects on soil quality:I. Chemical, mechanical, and biological properties. Soil Science Society of America Journal 78, 262. https://doi.org/10.2136/sssaj2013.07.0301.

Aguilera, E., Lassaletta, L., Gattinger, A., Gimeno, B.S., 2013. Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: a meta-analysis. Agriculture, Ecosystems & Environment 168, 25–36. https://doi.org/10.1016/j.agee.2013.02.003.

Alvarez, R., Steinbach, H.S., De Paepe, J.L., 2017. Cover crop effects on soils and subsequent crops in the pampas: a meta-analysis. Soil and Tillage Research 170, 53–65. https://doi.org/10.1016/j.still.2017.03.005.

Araujo, A.S.F., Leite, L.F.C., De Freitas Iwata, B., De Andrade Lira, M., Xavier, G.R., Do Vale Barreto Figueiredo, M., 2012. Microbiological process in agroforestry systems. A review. Agronomy for Sustainable Development 32, 215–226. https://doi.org/10.1007/s13593-011-0026-0.

R Core Team, 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria, 2013.

Baker, J.M., Ochsner, T.E., Venterea, R.T., Griffis, T.J., 2007. Tillage and soil carbon sequestration—what do we really know? Agriculture, Ecosystems & Environment 118, 1–5.

Balakrishna, A.N., Lakshmipathy, R., Bagyaraj, D.J., Ashwin, R., 2017. Influence of alley copping system on AM fungi, microbial biomass C and yield of finger millet, peanut and pigeon pea. Agroforestry Systems 91, 487–493. https://doi.org/10.1007/s10457-016-9949-4.

Bandick, A.K., Dick, R.P., 1999. Field management effects on soil enzyme activities. Soil Biology and Biochemistry 31, 1471–1479.

Berhe, A.A., Harte, J., Harden, J.W., Torn, M.S., 2007. The significance of the erosion-induced terrestrial carbon sink. BioScience 57, 337–346. https://doi.org/10.1641/B570408.

Blanco-Canqui, H., Mikha, M.M., Presley, D.R., Claassen, M.M., 2011. Addition of cover crops enhances No-till potential for improving soil physical properties. Soil Science Society of America Journal 75, 1471. https://doi.org/10.2136/sssaj2010.0430.
Blanco-Canqui, H., Holman, J.D., Schlegel, A.J., Tatarko, J., Shaver, T.M., 2013. Replacing fallow with cover crops in a semiarid soil: effects on soil properties. Soil Science Society of America Journal 77, 1026. https://doi.org/10.2136/sssaj2013.01.0006.
Blanco-Canqui, H., Shaver, T.M., Lindquist, J.L., Shapiro, C.A., Elmore, R.W., Francis, C. A., Hergert, G.W., 2015. Cover crops and ecosystem services: insights from studies in temperate soils. Agronomy Journal 107, 2449–2474. https://doi.org/10.2134/agronj15.0086.
Callesen, I., Liski, J., Raulund-Rasmussen, K., Olsson, M.T., Tau-Strand, L., Vesterdal, L., Westman, C.J., 2003. Soil carbon stores in Nordic well-drained forest soils-relationships with climate and texture class. Global Change Biology 9, 358–370. https://doi.org/10.1046/j.1365-2486.2003.00587.x.
Campbell, C.A., Biederbeck, V.O., Zentner, R.P., Lafond, G.P., 1991. Effect of crop rotations and cultural practices on soil organic matter, microbial biomass and respiration in a thin Black Chernozem. Canadian Journal of Plant Science 71, 363–376.
Churkina, G., Running, S.W., 1998. Contrasting climatic controls on the estimated productivity of global terrestrial biomes. Ecosystems 1, 206–215.
Clark, A., 2008. Managing Cover Crops Profitably. Diane Publishing.
Cooper, J., Baranski, M., Stewart, G., Nobel-de Lange, M., Bàrberi, P., Fließbach, A., Peigné, J., Berner, A., Brock, C., Casagrande, M., Crowley, O., David, C., De Vliegher, A., Döring, T.F., Dupont, A., Entz, M., Grosse, M., Haase, T., Halde, C., Hammerl, V., Huiting, H., Leithold, G., Messmer, M., Schloter, M., Sukkel, W., van der Heijden, M.G.A., Willekens, K., Wittwer, R., Mäder, P., 2016. Shallow non-inversion tillage in organic farming maintains crop yields and increases soil C stocks: a meta-analysis. Agronomy for Sustainable Development 36. https://doi.org/10.1007/s13593-016-0354-1.
Don, A., Schumacher, J., Freibauer, A., 2011. Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. Global Change Biology 17, 1658–1670. https://doi.org/10.1111/j.1365-2486.2010.02336.x.
Faé, G.S., Mark Sulc, R., Barker, D.J., Dick, R.P., Eastridge, M.L., Lorenz, N., 2009. Integrating winter annual forages into a no-till corn silage system. Agronomy Journal 101, 1286–1296. https://doi.org/10.2134/agronj2009.0144.
Franzluebbers, A.J., Haney, R.L., Honeycutt, C.W., Schomberg, H.H., Hons, F.M., 2000. Flush of carbon dioxide following rewetting of dried soil relates to active organic pools. Soil Science Society of America Journal 64, 613–623. https://doi.org/10.2136/sssaj2000.642613x.
Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mader, P., Stolze, M., Smith, P., Scialabba, N.E.-H., Niggli, U., 2012. Enhanced top soil carbon stocks under organic farming. Proceedings of the National Academy of Sciences 109, 18226–18231. https://doi.org/10.1073/pnas.1209429109.
Gyawali, A.J., Stewart, R.D., 2019. An improved method for quantifying soil aggregate stability. Soil Science Society of America Journal 83, 27–36. https://doi.org/10.2136/sssaj2018.06.0235.
Haruna, S.I., Nkongolo, N.V., Anderson, S.H., Eivazi, F., Zaibon, S., 2018. In situ infiltration as influenced by cover crop and tillage management. Journal of Soil and Water Conservation 73, 164–172. https://doi.org/10.2489/jswc.73.2.164.
Hassink, J., Whitmore, A.P., 1997. A model of the physical protection of organic matter in soils. Soil Science Society of America Journal 61, 131–139. https://doi.org/10.2136/sssaj1997.03615995006100010020x.
Hassink, J., Whitmore, A., Kubát, J., 1997. Size and density fractionation of soil organic matter and the physical capacity of soils to. European Journal of Agronomy 7, 189–199. https://doi.org/10.1016/S0378-519X(97)80025-6.
Houghton, R.A., 1995. Land-use change and the carbon cycle. Global Change Biology 1, 275–287.
Huang, P.M., Schnitzer, M., Stotzky, G., 1986. Influence of soil mineral colloids on metabolic processes, growth, adhesion, and ecology of microbes and viruses. In: Huang, P.M., Schnitzer, M. (Eds.), Interactions of Soil Minerals with Natural Organics and Microbes. Soil Science Society of America Special Publication, Madison, WI, pp. 305–428. https://doi.org/10.2136/sssaspecpub17.c10.
Idowu, O.J., Van Es, H.M., Abawi, G.S., Wolfe, D.W., Schindelbeck, R.R., Moebius-Clune, B.N., Gugino, B.K., 2009. Use of an integrative soil health test for evaluation of soil management impacts. Renewable Agriculture and Food Systems 24, 214–224. https://doi.org/10.1017/S1742170509990068.
Jackson, R.B., Le Quéré, C., Andrew, R.M., Canadell, J.G., Peters, G.P., Roy, J., Wu, L., 2017. Warning signs for stabilizing global CO2 emissions. Environmental Research Letters 12, 110202. https://doi.org/10.1088/1748-9326/aa9662.
Jani, A.D., Grossman, J., Smyth, T.J., Hu, S., 2016. Winter legume cover-crop root decomposition and N release dynamics under disking and roller-crimping termination approaches. Renewable Agriculture and Food Systems 31, 214–229. https://doi.org/10.1017/S1742170515000113.
Jian, J., Stewart, R., Du, X., 2019. SoilHealthDB [WWW document]. Figshare. https://doi.org/10.6084/m9.figshare.8292176.v8.
Jian, J., Du, X., Stewart D., R., 2020. A database for global soil health assessment. Scientific Data 7 (16), 1–8. https://doi.org/10.1038/s41597-020-0356-3.
Jiang, P., Anderson, S.H., Kitchen, N.R., Sadler, E.J., Sudduth, K.a., 2007. Landscape and conservation management effects on hydraulic Properties of a claypan-soil toposequence. Soil Science Society of America Journal 71, 803. https://doi.org/10.2136/sssaj2006.0236.
Kahle, D., Wickham, H., 2013. ggmap: spatial Visualization with ggplot2. R Journal 5, 144–161.
Kaye, J.P., Quemada, M., 2017. Using cover crops to mitigate and adapt to climate change. Rev. Agron. Sustain. Dev. 37 https://doi.org/10.1007/s13593-016-0410-x.
Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. Updated world map of the Köppen-Geiger climate classification. Meteorologische Zeitschrift 15, 259–263.
Krull, E.S., Baldock, J.A., Skjemstad, J.O., 2003. Importance of mechanisms and processes of the stabilisation of soil organic matter for modelling carbon turnover. Functional Plant Biology 30, 207–222.
Kuo, S., Sainju M., U., Jellum J., E., 1997. Winter Cover Crop Effects on Soil Organic Carbon and Carbohydrate in Soil. Soil Science Society of America Journal 61 (1), 145–152.
Kutsch, W.L., Bahn, M., Heinemeyer, A., 2009. Soil Carbon Dynamics: an Integrated Methodology. Cambridge University Press.
Lesur-Dumoulin, C., Malézieux, E., Ben-Ari, T., Langlais, C., Makowski, D., 2017. Lower average yields but similar yield variability in organic versus conventional horticulture. A meta-analysis. Agronomy for Sustainable Development 37. https://doi.org/10.1007/s13593-017-0455-5.
Lloyd, J., Taylor, J.A., 1994. On the temperature dependence of soil respiration. Functional Ecology 8, 315. https://doi.org/10.2307/2389824.
Luo, Y., Hui, D., Zhang, D., 2006. Elevated CO2 stimulates net accumulations of carbon and nitrogen in land ecosystems: a meta-analysis. Ecology 87, 53–63. https://doi.org/10.1890/04-1724.
Luo, Z., Wang, E., Sun, O.J., 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agriculture, Ecosystems & Environment 139, 224–231. https://doi.org/10.1016/j.agee.2010.08.006.
Manzoni, S., Jackson, R.B., Trofymow, J.A., Porporato, A., 2008. The global stoichiometry of litter nitrogen mineralization. Science 321, 684–686. https://doi.org/10.1126/science.1159792.
Marques, M.J., García-Muñoz, S., Muñoz-Organero, G., Bienes, R., 2010. Soil conservation beneath grass cover in hillside vineyards under mediterranean climatic conditions (MADRID, Spain). Land Degradation & Development 21, 122–131. https://doi.org/10.1002/ldr.915.
Mazzoncini, M., Sapkota, T.B., Bàrberi, P., Antichi, D., Risaliti, R., 2011. Long-term effect of tillage, nitrogen fertilization and cover crops on soil organic carbon and total nitrogen content. Soil and Tillage Research 114, 165–174. https://doi.org/10.1016/j.still.2011.05.001.
McGuire, a.D., Sitch, S., Clein, J.S., Dargaville, R., Esser, G., Foley, J., Heimann, M., Joos, F., Kaplan, J., Kicklighter, D.W., Meier, R.a., Melillo, J.M., Moore, B., Prentice, I.C., Ramankutty, N., Reichenau, T., Schloss, a., Tian, H., Williams, L.J., Wittenberg, U., 2001. Carbon balance of the terrestrial biosphere in the Twentieth Century: analyses of CO 2 , climate and land use effects with four process-based ecosystem models. Global Biogeochemical Cycles 15, 183–206. https://doi.org/10.1029/2000GB001298.
Meyer, L.D., Dabney, S.M., Murphree, C.E., Harmon, W.C., Grissinger, E.H., 1997. Effect of cropland management practices on storm runoff and erosion. In: Proceedings of the Conference on Management of Landscapes Distributed by Channel Incision. The University of Mississippi, Oxford Campus, Mississippi, pp. 983–989.
Mirsky, S.B., Ackroyd, V.J., Cordeau, S., Curran, W.S., Hashemi, M., Reberg-Horton, S.C., Ryan, M.R., Spargo, J.T., 2017. Hairy vetch biomass across the eastern United States: effects of latitude, seeding rate and date, and termination timing. Agronomy Journal 109, 1510–1519. https://doi.org/10.2134/agronj2016.09.0556.
Moebius-Clune, B.N., Moebius-Clune, D.J., Gugino, B.K., Idowu, O.J., Schindelbeck, R.R., Ristow, A.J., van Es, H.M., Thies, J.E., Shayler, H.A., McBride, M.B., Kurtz, K.S.M., Wolfe, D.W., Abawi, G.S., 2016. Comprehensive Assessment of Soil Health: the Cornell Framework Manual, Third. ed. Cornell University, Ithaca, New York.
Nachtergaele, F., van Velthuizen, H., Verelst, L., Batjes, N.H., Dijkshoorn, K., van Engelen, V.W.P., Fischer, G., Jones, A., Montanarela, L., 2010. The harmonized world soil database. In: Proceedings of the 19th World Congress of Soil Science. Soil Solutions for a Changing World, Brisbane, Australia, pp. 1–6. August 2010. pp. 34–37.
Ndiaye, E.L., Sandeno, J.M., McGrath, D., Dick, R.P., 2000. Integrative biological indicators for detecting change in soil quality. American Journal of Alternative Agriculture 15, 26. https://doi.org/10.1017/S0889189300008432.
Nelson, D.W., Sommers, L.E., 1996. Total carbon, organic carbon, and organic matter. Methods Soil Anal. Part 3—Chem. Methods 961–1010.
Olson, K., Ebelhar, S.A., Lang, J.M., 2014. Long-term effects of cover crops on crop yields, soil organic carbon stocks and sequestration. Open Journal of Soil Science 4, 284–292. https://doi.org/10.4236/ojss.2014.48030.
O'Dea, J.K., Miller, P.R., Jones, C.A., 2013. Greening summer fallow with legume green manures: on-farm assessment in north-central Montana. Journal of Soil and Water Conservation 68, 270–282. https://doi.org/10.2489/jswc.68.4.270.
Poeplau, C., Don, A., 2015. Carbon sequestration in agricultural soils via cultivation of cover crops - a meta-analysis. Agriculture, Ecosystems & Environment 200, 33–41. https://doi.org/10.1016/j.agee.2014.10.024.
Poeplau, C., Don, A., Vesterdal, L., Leifeld, J., Van Wesemael, B., Schumacher, J., Gensior, A., 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone - carbon response functions as a model approach. Global Change Biology 17, 2415–2427. https://doi.org/10.1111/j.1365-2486.2011.02408.x.
Puget, P., Chenu, C., Balesdent, J., 2000. Dynamics of soil organic matter associated with particle-size fractions of water-stable aggregates. European Journal of Soil Science 51, 595–605. https://doi.org/10.1111/j.1365-2389.2000.00353.x.
Reese, C.L., Clay, D.E., Clay, S.A., Bich, A.D., Kennedy, A.C., Hansen, S.A., Moriles, J., 2014. Winter cover crops impact on corn production in semiarid regions. Agronomy Journal 106, 1479–1488. https://doi.org/10.2134/agronj13.0540.
Reynolds, C.A., Jackson, T.J., Rawls, W.J., 2000. Estimating soil water-holding capacities by linking the Food and Agriculture Organization soil map of the world with global pedon databases and continuous pedotransfer functions. Water Resources Research 36, 3653–3662. https://doi.org/10.1029/2000WR900130.

Sainju, U.M., Singh, B.P., Whitehead, W.F., Wang, S., 2006. Carbon supply and storage in tilled and nontilled soils as influenced by cover crops and nitrogen fertilization. J. Environ. Qual. 35, 1507. https://doi.org/10.2134/jeq2005.0189.
Scharlemann, J.P.W., Tanner, E.V.J., Hiederer, R., Kapos, V., 2014. Global soil carbon: understanding and managing the largest terrestrial carbon pool. Carbon Management 5, 81–91. https://doi.org/10.4155/cmt.13.77.
Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kögel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P., Rasse, D.P., Weiner, S., Trumbore, S.E., 2011. Persistence of soil organic matter as an ecosystem property. Nature. https://doi.org/10.1038/nature10386.
Schmidt, H.P., Pandit, B.H., Cornelissen, G., Kammann, C.I., 2017. Biochar-based fertilization with liquid nutrient enrichment: 21 field trials covering 13 crop species in Nepal. Land Degradation & Development 28, 2324–2342. https://doi.org/10.1002/ldr.2761.
Shi, L., Feng, W., Xu, J., Kuzyakov, Y., 2018. Agroforestry systems: meta-analysis of soil carbon stocks, sequestration processes, and future potentials. Land Degradation & Development. https://doi.org/10.1002/ldr.3136.
Sileshi, G., 2009. In: Evidence for Impact of Green Fertilizers on Maize Production in Sub-saharan Africa.
Six, J., Bossuyt, H., Degryze, S., Denef, K., 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. Soil and Tillage Research. https://doi.org/10.1016/j.still.2004.03.008.
Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. Geoderma 74, 65–105. https://doi.org/10.1016/S0016-7061(96)00036-5.
Spargo, J.T., Alley, M.M., Follett, R.F., Wallace, J.V., 2008. Soil carbon sequestration with continuous no-till management of grain cropping systems in the Virginia coastal plain. Soil and Tillage Research 100, 133–140. https://doi.org/10.1016/j.still.2008.05.010.
Stavi, I., Lal, R., Jones, S., Reeder, R.C., 2012. Implications of cover crops for soil quality and geodiversity in a humid-temperate region in the midwestern USA. Land Degradation & Development 23, 322–330. https://doi.org/10.1002/ldr.2148.
Steele, M.K., Coale, F.J., Hill, R.L., 2012. Winter annual cover crop impacts on No-till soil physical properties and organic matter. Soil Science Society of America Journal 76, 2164. https://doi.org/10.2136/sssaj2012.0008.
Stewart, R.D., Jian, J., Gyawali, A.J., Thomason, W.E., Badgley, B.D., Reiter, M.S., Strickland, M.S., 2018. What we talk about when we talk about soil health. Agric. Environ. Lett. 5–9. https://doi.org/10.2134/ael2018.06.0033.
Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., 2013. Climate Change 2013: the Physical Science Basis.
Tellatin, S., Myers, R.L., 2018. Cover crop impacts on US cropland carbon sequestration. Journal of Soil and Water Conservation 73, 117A–121A. https://doi.org/10.2489/jswc.73.5.117A.
Tesfahunegn, G.B., Tamene, L., Vlek, P.L.G., Mekonnen, K., 2016. Assessing farmers' Knowledge of weed species , crop type and soil management practices in relation to soil quality status in mai-negus catchment, northern Ethiopia. Land Degradation & Development 27, 120–133.
Thenkabail, P.S., Mariotto, I., Gumma, M.K., Middleton, E.M., Landis, D.R., Huemmrich, K.F., 2013. Selection of hyperspectral narrowbands (HNBs) and composition of hyperspectral twoband vegetation indices (HVIs) for biophysical characterization and discrimination of crop types using field reflectance and Hyperion/EO-1 data. IEEE J. Selected Topics Appl. Earth Observ. Remote Sens. 6, 427–439.
Tian, D., Xiang, Y., Wang, B., Li, M., Liu, Y., Wang, J., Li, Z., Niu, S., 2018. Cropland abandonment enhances soil inorganic nitrogen retention and carbon stock in China: a meta-analysis. Land Degradation & Development. https://doi.org/10.1002/ldr.3137.
Tonitto, C., David, M.B., Drinkwater, L.E., 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: a meta-analysis of crop yield and N dynamics. Agriculture, Ecosystems & Environment 112, 58–72. https://doi.org/10.1016/j.agee.2005.07.003.
Tremblay, N., Bouroubi, Y.M., Bélec, C., Mullen, R.W., Kitchen, N.R., Thomason, W.E., Ebelhar, S., Mengel, D.B., Raun, W.R., Francis, D.D., Vories, E.D., Ortiz-Monasterio, I., 2012. Corn response to nitrogen is influenced by soil texture and weather. Agronomy Journal 104, 1658–1671. https://doi.org/10.2134/agronj2012.0184.
Utomo, M., Frye, W.W., Blevins, R.L., 1990. Sustaining soil nitrogen for corn using hairy vetch cover crop. Agronomy Journal 82, 979. https://doi.org/10.2134/agronj1990.00021962008200050028x.
Van Eerd, L.L., Congreves, K.A., Hayes, A., Verhallen, A., Hooker, D.C., 2014. Long-term tillage and crop rotation effects on soil quality, organic carbon, and total nitrogen. Canadian Journal of Soil Science 94, 303–315. https://doi.org/10.4141/cjss2013-093.
Wiesmeier, M., Hübner, R., Barthold, F., Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., von Lützow, M., Kögel-Knabner, I., 2013a. Amount, distribution and driving factors of soil organic carbon and nitrogen in cropland and grassland soils of southeast Germany (Bavaria). Agriculture, Ecosystems & Environment 176, 39–52. https://doi.org/10.1016/j.agee.2013.05.012.
Wiesmeier, M., Prietzel, J., Barthold, F., Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., von Lützow, M., Kögel-Knabner, I., 2013b. Storage and drivers of organic carbon in forest soils of southeast Germany (Bavaria) - implications for carbon sequestration. Forest Ecology and Management 295, 162–172. https://doi.org/10.1016/j.foreco.2013.01.025.
Zhou, X.V., Larson, J.A., Boyer, C.N., Roberts, R.K., Tyler, D.D., 2017. Tillage and cover crop impacts on economics of cotton production in Tennessee. Agronomy Journal 109, 2087–2096. https://doi.org/10.2134/agronj2016.12.0733.
Zhou, G., Xu, S., Ciais, P., Manzoni, S., Fang, J., Yu, G., Tang, X., Zhou, P., Wang, W., Yan, J., Wang, G., Ma, K., Li, S., Du, S., Han, S., Ma, Y., Zhang, D., Liu, J., Liu, S., Chu, G., Zhang, Q., Li, Y., Huang, W., Ren, H., Lu, X., Chen, X., 2019. Climate and litter C/N ratio constrain soil organic carbon accumulation. Nat. Sci. Rev. https://doi.org/10.1093/nsr/nwz045.
Zhu, B., Yi, L., xia, Xu, shui, H., Guo, L., mei, Hu, gao, Y., Zeng, Z. hai, Chen, F., Liu, Z. yong, 2016. Non-leguminous winter cover crop and nitrogen rate in relation to double rice grain yield and nitrogen uptake in Dongting Lake Plain, Hunan Province, China. J. Integr. Agric. 15, 2507–2514. https://doi.org/10.1016/S2095-3119(16)61331-X.
Zinn, Y.L., Lal, R., Resck, D.V.S., 2005. Texture and organic carbon relations described by a profile pedotransfer function for Brazilian Cerrado soils. Geoderma 127, 168–173. https://doi.org/10.1016/j.geoderma.2005.02.010.

Update

Soil Biology and Biochemistry
Volume 161, Issue , October 2021, Page

DOI: https://doi.org/10.1016/j.soilbio.2021.108389

Soil Biology and Biochemistry 161 (2021) 108389

Contents lists available at ScienceDirect

Soil Biology and Biochemistry

journal homepage: www.elsevier.com/locate/soilbio

Corrigendum to "A meta-analysis of global cropland soil carbon change from cover cropping" [Soil Biol. Biochem. 143 (2020) 107735]

Jinshi Jian[a,b], Xuan Du[c], Mark S. Reiter[a,d], Ryan D. Stewart[a,*]

[a] *School of Plant and Environmental Sciences, Virginia Tech, Blacksburg, VA, USA*

[b] *Pacific Northwest National Laboratory-University of Maryland Joint Global Change Research Institute, 5825 University Research Court, Suite, 3500, College Park, MD, USA*

[c] *Department of Hydraulic Engineering, Yangling Vocational & Technical College, Yang Ling, Shaanxi, China*

[d] *Eastern Shore Agricultural Research and Extension Center, Virginia Tech, Painter, VA, USA*

The authors regret that Figure A5b (Supplementary Data) included an error. The originally published figure was labelled as presenting the relationship between bulk density and soil organic carbon concentration (%), but the figure actually showed bulk density versus soil organic carbon stocks (Mg/ha). Figure A5b has been revised to present bulk density versus soil organic carbon concentrations (%), as this relationship was the basis for the analyses presented in the manuscript. The correct data have an *adj*R^2 value of 0.20, whereas *adj*$R^2 = 0.47$ was reported in the original text. All other analyses remain unchanged.

The authors would like to apologise for any inconvenience caused.

DOI of original article: https://doi.org/10.1016/j.soilbio.2020.107735.

* Corresponding author.

E-mail address: ryan.stewart@vt.edu (R.D. Stewart).

https://doi.org/10.1016/j.soilbio.2021.108389

Available online 12 August 2021

0038-0717/

Exhibit G

Agroforest Syst (2012) 86:105–111
DOI 10.1007/s10457-012-9573-x

Agroforestry for biomass production and carbon sequestration: an overview

Shibu Jose · Sougata Bardhan

Received: 10 September 2012 / Accepted: 26 September 2012 / Published online: 4 October 2012

Abstract Ever since the Kyoto Protocol, agroforestry has gained increased attention as a strategy to sequester carbon (C) and mitigate global climate change. Agroforestry has been recognized as having the greatest potential for C sequestration of all the land uses analyzed in the Land-Use, Land-Use Change and Forestry report of the IPCC; however, our understanding of C sequestration in specific agroforestry practices from around the world is rudimentary at best. Similarly, while agroforestry is well recognized as a land use practice capable of producing biomass for biopower and biofuels, very little information is available on this topic. This thematic issue is an attempt to bring together a collection of articles on C sequestration and biomass for energy, two topics that are inextricably interlinked and of great importance to the agroforestry community the world over. These papers not only address the aboveground C sequestration, but also the belowground C and the role of decomposition and nutrient cycling in determining the size of soil C pool using specific case studies. In addition to providing allometric methods for quantifying biomass production, the biological and economic realities of producing biomass in agroforestry practices are also discussed.

S. Jose (✉) · S. Bardhan
The Center for Agroforestry, School of Natural Resources, University of Missouri, 203 Anheuser Busch Natural Resources Building, Columbia, MO 65211, USA
e-mail: joses@missouri.edu

Keywords Soil carbon · Coffee agroforestry · Cacao agroforestry · Bioenergy · Biofuels · Allometric equations · Biomass crops

Introduction

Global climate change and energy security are two key issues that are at the forefront of environmental discussions the world over. Although they bring up unique challenges, global warming and energy security are inextricably interlinked. Increasing concentration of atmospheric carbon dioxide (CO_2) is considered the predominant cause of global climatic change. It is believed that agricultural and forestry practices can partially mitigate increasing CO_2 concentration by sequestering carbon (C). Similarly, alternative agricultural practices where biomass crops are cultivated can impact CO_2 levels not only by sequestering C, but also by replacing fossil fuel with the biomass produced. Agroforestry, like many other land use systems, offers great potential for sequestering C and producing biomass for biofuels.

Ever since the Kyoto Protocol, agroforestry has gained increased attention as a strategy to sequester C from both developed and developing nations. The available estimates of C stored in agroforestry range from 0.29 to 15.21 Mg C/ha/year above ground, and 30–300 Mg C/ha up to 1 m depth in the soil (Nair et al. 2010). Since the industrial revolution, atmospheric CO_2 has increased by more than 40 %, from 280 ppm in 1,750 to about 392 ppm in 2012 and at the current rate it is expected to surpass 400 ppm by 2015 (Hutchinson et al. 2007; Tans 2012). In the three years from 2010 to 2012, CO_2 emissions increased at an alarming rate of 2 ppm/year or 4.4 Pg C/year. While agroforestry has been recognized as having the greatest potential for C sequestration (Fig. 1) of all the land uses analyzed in the Land-Use, Land-Use Change and Forestry report of the IPCC (2000), our understanding of C sequestration in specific agroforestry practices from around the world is rudimentary at best.

The incorporation of trees or shrubs on farms or pastures can increase the amount of C sequestered compared to a monoculture field of crop plants or pasture (Sharrow and Ismail 2004; Kirby and Potvin 2007). In addition to the significant amount of C stored in aboveground biomass, agroforestry systems can also store C belowground. While most studies report aboveground C sequestration, belowground C and soil C are often not reported from agroforestry systems. The soil C pool, comprising about 2,500 Gt, is one of the largest C pools and is larger than the atmospheric pool (760 Gt) (Lal 2004). The extent of soil C is dependent on a delicate balance between litter and rhizodeposition and the release of C due to decomposition and mineralization. Several other factors such as quality of C input, climate, soil physical and chemical properties further determine the rate of decomposition and thus stabilization of soil organic C in a particular ecosystem. Since modernization of agriculture in the 19th-century, soil carbon pool has gradually depleted because of several factors such as deforestation, intensive cropping and biomass removal, soil erosion, and unsustainable agricultural practices. Most of the decline in soil organic matter has been observed in regions under intensive crop production such as continuous row cropping or monocropping. Depletion of soil C has been documented to result in decreased productivity, poor soil physical and chemical properties, and negative secondary environmental impacts. It has been well documented that conversion of degraded agricultural soils into agroforestry systems can rebuild soil productivity.

One of the commodities agroforestry is well suited to producing is biomass for biopower and biofuels (Jose et al. 2012). Heavy reliance on foreign based fossil fuels has sparked an interest in domestic renewable energy sources in many countries. For example, in 2003 the Biomass Research and Development Technical Committee (BRDTC), established by US Congress in 2000, envisioned a goal of a 30 % replacement of US petroleum consumption with biofuels by 2030 (DOE; US Department of Energy 2003). Currently, petroleum products supply about 36 % of US energy consumption, while biomass and biofuels provide 4.3 % of total US energy consumption (EIA; Energy Information Administration 2011).

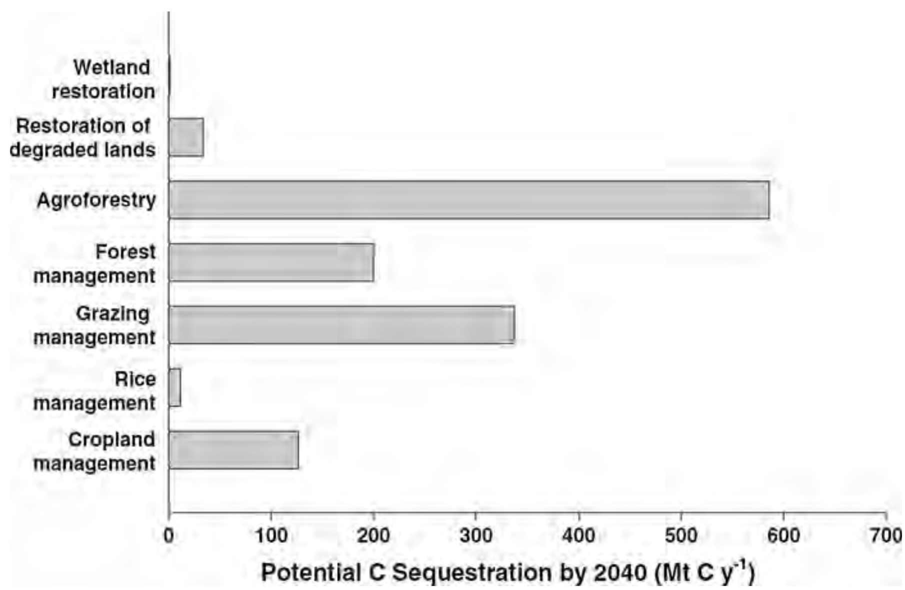


Fig. 1 Carbon sequestration potential of different land use systems by 2040 (adapted from IPCC 2000). Agroforestry offers the greatest potential because of the large extent of area (630×10^6 ha) available worldwide for agroforestry adoption

While a study conducted by the US Department of Energy concluded that achieving this goal is possible, the report stated significant expansion of perennial biofuel crop production would be necessary (DOE 2011). Although agroforestry offers a solution to avoid the food versus fuel debate by combining food production and biomass for energy on the same piece of land (Henderson and Jose 2010; Holzmueller and Jose 2012), very little information is available on this topic.

If agroforestry is to be used in C sequestration schemes such as the clean development mechanisms (CDM), better information is required about above and belowground C stocks, soil C, and areas under agroforestry practices. Although there have been a number of publications recently and in the past about the C sequestration potential of agroforestry systems, the information is widely dispersed (but, see Kumar and Nair 2011). The objective of this thematic issue is to compile several original research articles from North America, South America, and Africa that investigate C sequestration and biomass production potential of specific agroforestry practices.

Quantifying carbon sequestration

The first two papers examined C sequestration in silvopastoral systems. Dube et al. investigated the carbon (C) sequestration potentials of three predominant ecosystems in Patagonia in Chile: *Pinus ponderosa*-based silvopastoral systems (SPS), pine plantations (PPP) and natural pasture (PST). Silvopastoral systems are highly efficient in increasing productivity for both plants and animals as mutually optimum conditions for growth and development are created in a properly managed silvopastoral system. Plants gain benefits through nutrient cycling by addition of manure in the system and partial shade from the canopy while animals enjoy ideal temperature and humidity under the tree canopy. In their study, Dube et al. observed higher aboveground tree C, belowground tree C, and soil organic C stock in the silvopastoral system compared to the other systems. Silvopastoral systems also had more favorable air temperature and soil moisture parameters.

Ermson et al. conducted a similar study in the southeastern US where they explored the effect of grazing and forage enhancement on total soil C (TSC), soil nitrogen(N), and phosphorus (P) dynamics in a goat (*Capra aegagrus hircus*)—loblolly pine (*Pinus taeda* L.) silvopasture system on a Kipling silt loam soil (fine, smectitic, thermic, Typic Paleudalfs) in Alabama from 2006 to 2010. In this study however, silvopasture plots were characterized by low initial pH, low TSC, and the soils were deficient in N and P. Four years after tree thinning and 3 years of grazing in June 2010, the silvopasture treatment still exhibited low soil pH (<6) and TSC levels of less than 20 g/kg. The authors speculated that in the long-term, grazing without additional soil management practices may still improve soil fertility through nutrient recycling and C sequestration and thereby making the goat-loblolly silvopasture system both environmentally and economically sustainable.

The next two papers investigated C sequestration in coffee agroforests, one in Guatemala and another in Costa Rica. Schmitt-Harsh observed that coffee agroforests in Guatemala stored somewhere between 74.0 and 259.0 Mg C/ha with a mean of 127.6 Mg C/ha. The average carbon stocks of coffee agroforests were significantly lower than estimated for the mixed dry forests (198.7 Mg C/ha); however, individual tree and soil C pools were not significantly different suggesting that shade trees played an important role in facilitating C sequestration and soil conservation in these systems. This research demonstrates the importance of conservation-based production systems such as coffee agroforests in sequestering C alongside natural forest systems.

Hãger attempted to unravel the relationship between species composition, diversity, and C storage in coffee agroforests of Costa Rica. Total C stocks were 43 % higher on organic farms than on conventional farms ($P < 0.05$) and although vegetation structure was different, there was no difference in species diversity between organic and conventional farms. Combined effect of farm type, species richness, species composition and slope explained 83 % of the variation in total C storage across all farms. Organic coffee agroforestry farms may contribute to GHG mitigation and biodiversity conservation in a synergistic manner which has implications for the effective allocation of resources for conservation and climate change mitigation strategies in the agricultural sector.

There are three papers included in this thematic issue that provide unique perspectives on soil C sequestration and its interrelationship with organic matter decomposition and nutrient cycling. The first paper by Kim examined how an intercropping system with a nitrogen (N)-fixing tree (*Gliricidia*) and maize could help mitigate climate change through enhanced soil C sequestration in sub-Saharan Africa while dealing with GHG emissions from the soil. Using data from Makumba et al. (2007), the author estimated that 67.4 % of the sequestered soil C (76 Mg C/ha in 0–2 m soil depth) was lost from the system as CO_2 during the first 7 years of intercropping. An annual net gain 3.5 Mg C/ha/year was estimated from soil C sequestered and lost. The author further observed that if N_2O emission was reduced as well, the overall mitigation benefit achieved from the intercropping system would be larger. These results suggest that field measurements and modeling of CO_2, N_2O and CH_4 emissions should be taken into account while estimating C sequestration in agroforestry systems.

Gaiser et al. tested Leucaena (*Leucaena leucocephala* (L), *Senna siamea* (S) and maize (M) residue addition on soil organic matter accumulation under sub-humid tropical conditions in Benin, West Africa. On an *Imperata cylindrica* (I) dominated grass fallow, a total amount of 30 Mg/ha dry matter was applied within 18 months. Changes in the light and heavy soil organic C fraction (LF and HF) and in the total soil organic C content (LF + HF) in the topsoil were observed. All organic materials increased the proportion of the LF fraction in the soil significantly. The increase in HF was 39–51 % of the increase in total organic C, depending on the source of the organic material. The potential of the tested organic materials to increase total soil organic C content (including all soil organic C fractions) was in the order L > S > M > I, whereas the order of the HF fraction was L = S > I > M. Cation exchange capacity of the newly formed heavy soil organic C was highest with L and lowest with M. Ranking of the transformation efficiency of applied plant residues into the heavy soil organic C fraction was I > L = S > M. Transformation efficiency of the residues could neither be explained by lignin nor lignin/N ratio, but rather by extractable polyphenols. The results show that accumulation of the HF fraction in tropical soils is feasible through the application of large quantities of plant residues, but depends strongly on the quality of the organic matter added.

Zaia et al. evaluated the impact of plant litter deposition in cacao agroforestry systems on soil C, N, P and microbial biomass in Bahia, Brazil. They studied five cacao agroforestry systems of different ages under two different soils (Oxisol and Inceptisol). Overall, the average stocks of organic C, total N and total organic P for 0–50 cm soil depth were 89072, 8838 and 790 kg/ha, respectively. At this soil depth the average stock of labile organic P was 55.5 kg/ha. Microbial biomass was mostly dominated in the 0–10 cm soil depth, with a mean of microbial biomass C of 286 kg/ha, microbial biomass N of 168 kg/ha and mineralizable N of 79 kg/ha. The dynamics of organic P in these cacao agroforestry systems were not directly associated with organic C dynamics in soils, in contrast to the N dynamics.

Ecosystem models that can estimate plant and soil C stocks can be an invaluable tool for estimating C sequestration potential of agroforestry systems at larger scales. The CO_2FIX model has been used to estimate the dynamics of C stocks and flows for a variety of ecosystems around the world (Schelhaas et al. 2004). However, this model has not been tested using empirical data from agroforestry systems. Kaonga et al. tested the validity of the CO_2FIX model in predicting the aboveground and soil C stock using empirical data from 7-year-old *Leucaena* woodlots at Msekera, Zambia. They also assessed the impact of converting a degraded agricultural land to woodlots on C stocks. Measured above and belowground tree C stocks and increment of aboveground biomass differed significantly among different species. Measured stem and total aboveground tree C stocks in the *Leucaena* woodlots ranged from 17.1 to 29.2 and from 24.5 to 55.9 Mg/ha, respectively. Measured soil organic carbon (SOC) stocks at 0–200 cm depth in *Leucaena* stands ranged from 106.9 (*L. diversifolia*) to 186.0 Mg/ha (*L. leucocephala*). Although, modeled stem and branch C stocks closely matched measured stocks, the soil module of CO_2FIX could not predict the soil C accurately. The authors concluded that inadequate long-term empirical data on climate, litter quality, litter quantity, and tree growth, and the transient nature of SOC stocks that were disturbed in recent decades were most likely reasons for the inconclusive results from the model.

Udawatta and Jose synthesized the available information to estimate C sequestration under agroforestry systems in the US. They estimated that 530 Tg/year could be sequestered by four major agroforestry practices which could help offset current US emission rate of

1,600 Tg C/year from burning fossil fuel (coal, oil, and gas) by 33 %. These authors estimated C sequestration potential for silvopasture, alley cropping, and windbreaks in the US as 464, 52.4, and 8.6 Tg C/year, respectively. According to them, riparian buffers could sequester an additional 4.7 Tg C/year while protecting water quality. While acknowledging the need for accurate area estimates under different agroforestry practices in the US, they also emphasized the need for long-term data, standardized protocols for C quantification and monitoring, predictive models to understand long-term C sequestration, and site-specific agroforestry design criteria to optimize C sequestration.

The paper by Nair elaborated on the need for rigorous and consistent procedures to measure the extent of C sequestration in agroforestry systems. The author accurately pointed out that the current methods of estimating C varied widely and the estimations were based on several assumptions. According to him, large-scale global models based on such measurements and estimations were more likely to result in serious under- or overestimations of C in agroforestry practices. The author reveals several erroneous assumptions, operational inadequacies and inaccuracies commonly found in the current literature. He provides several practical recommendations for researchers that include using accurate description of the methods and procedures among others. This would help other researchers to examine the datasets and incorporate them in larger databases and help agroforestry earn its deserving place in mainstream efforts.

Estimating biomass production

Allometric equations are commonly used in estimating biomass production by trees in agroforestry systems. However, these equations are most often derived from forest grown trees that are different in their growth form from those open-grown trees in agroforestry configurations. This can introduce errors in estimating not only biomass production potential, but C sequestration as well. It is imperative that species specific allometric equations for different agroforestry practices must be developed in order to overcome this serious weakness in agroforestry research. There are two papers in this thematic issue that provide allometric equations for estimating aboveground biomass for trees in agroforestry.

Tamang et al. conducted their study in Florida, USA, to develop biomass equations for cadaghi (*Corymbia torelliana*) trees in various aged windbreaks. Trees were destructively sampled based on diameter at breast height (DBH) and crown biomass was estimated using randomized branch sampling (RBS) while trunk biomass was measured by taking disks every 1.5 m along the stem. Separate nonlinear equations were developed for crown, trunk and whole tree biomass estimation using DBH and height as predictors. The study found that DBH alone was sufficient to predict aboveground biomass while the inclusion of height provided more accurate results. Using their equation the authors recorded a total biomass per 100 m windbreak length to be between 166 and 26,605 kg. They concluded that fast-growing cadaghi could provide landowners higher returns from biomass or carbon trade to offset the cost of land occupied by the windbreaks.

Kuyah et al., on the other hand, developed new allometric equations using remotely sensed crown area and/or tree height as predictor of aboveground biomass. These equations corresponded well with the data obtained from destructive sampling with about 85 % of the observed variation in aboveground biomass explained by crown area. Addition of height and wood density as second predictor variables improved model fit by 6 and 2 % and lowered the relative error by 7 and 2 %, respectively. Total estimated aboveground biomass carbon was measured at 20.8 t C/ha, which was about 19 % more than the amount estimated using DBH as predictor. These results confirm that the new allometric equations using crown area could be a better predictor of aboveground biomass and can be used as an important tool for predicting carbon stock in such systems.

The last two papers explore biomass production potential of two temperate alley cropping systems, one from Canada and another from the US. As pointed out by Holzmueller and Jose (2012), alley cropping is one of the most suitable agroforestry practices for growing biomass for biopower and biofuels. Cardinael et al. examined short rotation willow production in the alleys of 21-year-old trees on marginal land in Guelph, Canada. As a control, the same willow clones were established on an adjacent piece of land without established trees (conventional willow system). They quantified carbon pools, fine root and leaf biomass inputs, and clone yields in both the intercropping and conventional monocropping systems. Willow biomass yield was significantly higher in the agroforestry field (4.86 odt ha^{-1}/year) compared to the conventional field (3.02 odt ha^{-1}/year). Clonal differences in biomass were also apparent with clones SV1 and SX67 with the highest yields and 9,882–41 with the lowest. Willow fine root biomass in the top 20 cm of soil was significantly higher in the intercropping system (3,062 kg/ha) than in the conventional system (2,536 kg/ha). Soil organic carbon was also significantly higher in the agroforestry field (1.94 %) than in the conventional field (1.82 %).

Susaeta et al. assessed the economics of loblolly pine (*Pinus taeda* L.)—switchgrass (*Panicum virgatum*) alley cropping in the southern US. Assuming a price range of switchgrass between \$15 and \$50 Mg^{-1} and yield of 12 Mg ha^{-1}/year, loblolly pine monoculture would be the most profitable option for landowners instead of intercropping if the price of switchgrass was below \$30 Mg^{-1}. However, when switchgrass prices were $\geq$\$30 Mg^{-1}, landowners would be financially better off adopting intercropping if competitive interaction between crops were minimal. Various assumptions were used in their analysis ranging from no competition between species for resources and reduced loblolly pine productivity due to competition with switchgrass to reduced productivity of both species due to competition for nutrients, water and light. Findings also suggested that the optimal system would depend on the competitive interactions between switchgrass and loblolly pine crops, and the expected prices for each crop.

Conclusion

Research findings from around the world have clearly demonstrated that agroforestry offers unique opportunities to increase C stocks in the terrestrial biosphere. Agroforestry could play a substantial role in reducing atmospheric concentration of CO_2 by (1) storing C in above and belowground biomass and in soil, and (2) growing biomass for biopower and biofuels and thereby replacing fossil fuel. Agroforestry could also protect existing C stocks if improved fallows and similar agroforestry practices could provide food and fuelwood, thereby reducing the rate of deforestation. Despite widespread recognition of agroforestry's potential for C sequestration and biomass production, our understanding of these topics is limited. There is still a lack of quantitative information from specific systems. This thematic issue is an attempt to bring together several original research articles from North America, South America, and Africa that investigate C sequestration and biomass production potential of specific agroforestry practices. While there are issues related to inconsistencies in methodologies, lack of soil C estimates and GHG emissions from soil, and reliable large-scale C estimates for different agroforestry practices, it is apparent that the research community is aggressively generating the much needed data at different scales. This will definitely help quantify agroforestry's role in C sequestration and biomass production and contribute meaningfully to global climate change mitigation efforts.

References

DOE (2011) US billion-ton update: biomass supply for a bioenergy and bioproducts industry. Perlack RD and Stokes BJ (Leads), ORNL/TM-2011/224. Oak Ridge National Laboratory, Oak Ridge, TN. p 227

DOE (US Department of Energy) (2003) Roadmap for agriculture biomass feedstock supply in the United States. US Department of Energy, DOE/NE-ID-1129

EIA (Energy Information Administration) (2011) Annual energy review 2010. US energy information administration, office of energy statistics, US Department of Energy, Washington, DC. http://www.eia.gov/totalenergy/data/annual/index.cfm, Accessed 20 Sep 2012

Henderson DE, Jose S (2010) Biomass production potential of three short rotation woody crop species under varying nitrogen and water availability. Agrofor Syst 80:259–273

Holzmueller EJ, Jose S (2012) Bioenergy crops in agroforestry systems: potential for the US North central region. Agrofor Syst 85:305–314

Hutchinson JJ, Campbell CA, Desjardins RL (2007) Some perspectives on carbon sequestration in agriculture. Agric For Meteorol 142:288–302

IPCC (2000) Land use, land-use change, and forestry. Cambridge University Press, Cambridge, UK, p 375 A special report of the IPCC

Jose S, Gold MA, Garrett HE (2012) The future of agroforestry in the United States. Pp. 217–246. In: Nair PKR, Garrity D (eds) Agroforestry: The future of global land use. Springer, The Netherlands, p 541

Kirby KR, Potvin C (2007) Variation in carbon storage among tree species: implications for the management of a small-scale carbon sink project. For Ecol Manag 246:208–221

Kumar BM, Nair PKR (2011) Carbon sequestration potential of agroforestry systems: opportunities and challenges. Springer, The Netherlands, p 307

Lal R (2004) Soil carbon sequestration impacts on global climate change and food security. Science 304:1623–1627

Makumba W, Akinnifesi FK, Janssen B, Oenema O (2007) Long-term impact of a gliricidiamaize intercropping system on carbon sequestration in southern Malawi. Agric Ecosyst Environ 11:237–243

Nair PKR, Nair VD, Kumar BM, Showalter JM (2010) Carbon sequestration in agroforestry systems. Adv Agron 108: 237–307

Schelhaas MJ, van Esch PW, Groen TA, de Jong BHJ, Kanninen M, Liski J, Masera O, Mohren GMJ, Nabuurs GJ, Palosuo T, Pedroni L,Vallejo A, Vilén T (2004) CO2FIX V 3.1 – A modelling framework for quantifying carbon sequestration in forest ecosystems, Wageningen, Alterra, Alterra-rapport 1068, 122 p

Sharrow SH, Ismail S (2004) Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. Agrofor Syst 60:123–130

Tans P (2012) Concentrations of CO_2 in the earth's atmosphere (parts per million) derived from in situ air measurements at the Mauna Loa observatory, Hawaii. ftp://ftp.cmdl.noaa.gov/ccg/co2/trends/co2_mm_mlo.txt; Accessed 24 Sep 2012

Exhibit H

MR. YAWEN HUANG (Orcid ID : 0000-0001-8854-6005)

PROF. WEI REN (Orcid ID : 0000-0003-0271-486X)

DR. DAFENG HUI (Orcid ID : 0000-0002-5284-2897)

DR. JIAN YANG (Orcid ID : 0000-0001-9121-3308)

Article type : Primary Research Articles

Xiongxiong Bai (ORCID: 0000-0001-5104-2946)

Article type: Primary Research Articles

Responses of soil carbon sequestration to climate smart agriculture practices: A meta-analysis

Keywords

Soil organic carbon, biochar, cover crop, conservation tillage, climate, meta-analysis

Running title

Climate-smart agriculture and C sequestration

Xiongxiong Bai[1, 2#], Yawen Huang[1#], Wei Ren[1]*, Mark Coyne[1], Pierre-Andre Jacinthe[3], Bo Tao[1], Dafeng Hui[4], Jian Yang[5], and Chris Matocha[1]

[1] Department of Plant and Soil Sciences, University of Kentucky, Lexington, KY 40546, USA.

[2] College of Life Sciences, Henan Normal University, Xinxiang, Henan 453007, China.

[3] Indiana University Purdue University Indianapolis, Indianapolis, IN 46202, USA.

[4] Department of Biological Sciences, Tennessee State University, Nashville, TN 37209, USA.

[5] Department of Forestry and Natural Resources, University of Kentucky, Lexington, KY 40546, USA.

[#] Xiongxiong Bai and Yawen Huang equally contribute to this work.

* Corresponding author: Wei Ren, wei.ren@uky.edu

Abstract

Climate-smart agriculture (CSA) management practices (e.g., conservation tillage, cover crops, and biochar applications) have been widely adopted to enhance soil organic carbon (SOC) sequestration and to reduce greenhouse gas emissions while ensuring crop productivity. However, current measurements regarding the influences of CSA management practices on SOC sequestration diverge widely, making it difficult to derive conclusions about individual and combined CSA management effects and bringing large uncertainties in quantifying the potential of the agricultural sector to mitigate climate change. We conducted a meta-analysis of 3,049 paired measurements from 417 peer-reviewed articles to examine the effects of three common CSA management practices on SOC sequestration as well as the environmental controlling factors. We found that, on average, biochar applications represented the most effective approach for increasing SOC content (39%), followed by cover crops (6%) and conservation tillage (5%). Further analysis suggested that the effects of CSA management practices were more pronounced in areas with relatively warmer climates or lower nitrogen fertilizer inputs. Our meta-analysis demonstrated that, through adopting CSA practices, cropland could be an improved carbon sink. We also highlight the importance of considering local environmental factors (e.g., climate and soil conditions and their combination with other management practices) in identifying appropriate CSA practices for mitigating greenhouse gas emissions while ensuring crop productivity.

1. Introduction

Soil organic carbon (SOC) is a primary indicator of soil health and plays a critical role in food production, greenhouse gas balance, and climate mitigation and adaptation (Lorenz & Lal, 2016). The dynamic of agricultural SOC is regulated by the balance between carbon inputs (e.g., crop residues and organic fertilizers) and outputs (e.g., decomposition and erosion) under long-term constant environment and management conditions. However, this balance has been dramatically altered by climate change, which is expected to enhance SOC decomposition and weaken the capacity of soil to sequester carbon (Wiesmeier *et al*., 2016). Generally, agricultural soils contain considerably less SOC than soils under natural vegetation due to land conversion and cultivation (Hassink, 1997; Poeplau & Don, 2015), with a potential to sequester carbon from the atmosphere through proper management practices (Lal, 2018). Therefore, it is crucial to seek practical approaches to enhance agricultural SOC sequestration without compromising the provision of ecosystem services such as food, fiber or other agricultural products.

Climate-smart agriculture (CSA) has been promoted as a systematic approach for developing agricultural strategies to ensure sustainable food security in the context of climate change (FAO, 2013). One of the major objectives of CSA is to reduce greenhouse gas emissions and enhance soil carbon sequestration and soil health (Campbell *et al*., 2014; Lipper *et al*., 2014). The key for sequestering more carbon in soils lies in increasing carbon inputs and reducing carbon outputs. Frequently recommended approaches for SOC sequestration include adding cover crops into the crop rotation, applying biochar to soils, and minimizing soil tillage (i.e., conservation tillage). In recent decades, these management practices have been applied in major agricultural regions globally, and a large number of observations/measurements have been accumulated (e.g., Chen *et al.*, 2009; Spokas *et al.*, 2009; Clark *et al.*, 2017).

Several mechanisms have been proposed to explain the positive effects of CSA management practices on SOC sequestration. For example, conservation tillage reduces soil disturbance and the soil organic matter decomposition rate (Salinas-Garcia *et al.*, 1997) and promotes fungal and earthworm biomass (Lavelle, 1999; Briones & Schmidt, 2017), thereby improving SOC stabilization (Liang & Balser, 2012). Cover crops provide additional biomass inputs from above- and belowground (Blanco-Canqui *et al.*, 2011), increase carbon and nitrogen inputs, and enhance the biodiversity of agroecosystems (Lal, 2004). Moreover, cover crops can promote soil aggregation and structure (Sainju *et al*., 2003), therefore indirectly reduce carbon loss from soil erosion (De Baets *et al*., 2011). Biochar amendments affect SOC dynamics through two pathways: (1) improving soil aggregation and physical protection of aggregate-associated SOC against microbial attack; (2) increasing the pool of recalcitrant organic substrates resulting in a low SOC decomposition rate and substantial negative priming (Zhang *et al.*, 2012; Du *et al.*, 2017a, Weng *et al.*, 2017).

Although these CSA management practices have been widely used to enhance soil health (e.g., Thomsen & Christensen, 2004; Denef *et al.*, 2007; Fungo *et al.*, 2017; Weng *et al.*, 2017), their effects on SOC sequestration are variable and highly dependent on experiment designs and site-specific conditions such as climate and soil properties (Poeplau & Don, 2015; Abdalla *et al.*, 2016; Liu *et al*., 2016; Paustian *et al.,* 2016). The potential to sequester soil carbon varies greatly among CSA practices, which has not been well addressed. Some studies even suggested negative effects of CSA management practices on SOC (e.g., Tian *et al*., 2005; Liang *et al*., 2007). Also, most prior quantitative research focused on the effects of a single CSA practice on SOC (e.g., Poeplau & Don, 2015; Abdalla *et al*., 2016; Liu *et al*., 2016), very few studies estimated the combined effects of diverse CSA and conventional management practices. Some recent studies reported that a combination of cover crops and conservation tillage could significantly increase SOC compared to a single management practice (Blanco-Canqui *et al*., 2013; Ashworth *et al*., 2014; Higashi *et al*., 2014; Duval *et al*., 2016). For example, Sainju *et al.* (2006) suggested that soil carbon sequestration may increase 0.267 Mg C ha^{-1} yr^{-1} under a combination of no-till and cover crop practices, where the latter was a mixed culture of hairy vetch (*Vicia villosa*) and rye (*Secale cereale*); in contrast, a carbon loss of 0.967 Mg C ha^{-1} yr^{-1} occurred when only no-till was used. Agegnehu *et al.* (2016) reported that 1.58% and 0.25% more SOC were sequestered in the mid-season and end-season, respectively, under conservation tillage when biochar was also applied. These findings highlight the importance of quantitatively evaluating the combined effects of multiple CSA management practices (including the combination of CSA and conventional management practices) on SOC sequestration under different climate and soil conditions.

This study aims to fill the above-mentioned knowledge gap through a meta-analysis to simultaneously examine the effects of three widely used CSA management practices (i.e., conservation tillage [no-till, NT; and reduced tillage, RT], cover crops, and biochar) on SOC sequestration (Fig. 1). Our scientific objectives were to: (1) evaluate and compare the effects of conservation tillage, cover crops, and biochar use on SOC; (2) examine how environmental factors (e.g., soil properties and climate) and other agronomic practices (e.g., nitrogen fertilization, residue management, irrigation, and crop rotation) influence SOC in these CSA management environments.

[Insert Figure 1]

2. Materials and methodology

2.1. Data collection

We extracted data from 417 peer-reviewed articles (297 for conservation tillage, 64 for cover crops, and 56 for biochar) published from 1990 to May 2017 (Data S1). Among all publications, 113 for conservation tillage, 32 for cover crops, and 7 for biochar were conducted in the U.S. All articles were identified from the Web of Science. The search keywords were "soil organic carbon" and "tillage" for conservation tillage treatments; "soil organic carbon" and "cover crop" for cover crop treatments; and "soil organic carbon" and "biochar" for biochar treatments. All selected studies meet the following inclusion criteria: (1) SOC was measured in field experiments (to estimate the potential of biochar to increase soil carbon, we also included soil incubation and pot experiments with regard to biochar use); (2) observations were conducted on croplands excluding orchards and pastures; (3) ancillary information was provided, such as experiment duration, replication, and sampling depth; and (4) other agronomic management practices were included besides the three target management practices in this study. We considered conventional tillage as the control for NT and RT. Experiments that eliminated any tillage operation were grouped into the NT category, and experiments using tillage with lower frequency or shallower till-depth or less soil disturbance in comparison to the paired conventional tillage (e.g., moldboard plow and chisel plow) were grouped into the RT category. Likewise, "no cover crop" and "no biochar" were treated as control experiments relative to cover crop and biochar treatments, respectively. We only considered studies that viewed cover crops as treatments and fallow (or weeds) as controls.

Soil organic carbon data were either derived from tables or extracted from figures using the GetData Graph Digitizer software v2.26 (http://getdata-graph-digitizer.com/download.php). Other related information from the selected studies was also recorded, including location (i.e., longitude and latitude), experiment duration, climate (mean annual air temperature and precipitation), soil properties (texture, depth, and pH), and other agronomic practices (crop residues, nitrogen fertilization, irrigation, and crop rotation). The study durations were grouped into three categories: short (≤5 years), medium (6-20 years), and long term (>20 years). Climate was grouped according to the aridity index published by UNEP (1997) as either arid (≤ 0.65) or humid (> 0.65). Study sites were grouped into cool (temperate and Mediterranean climates) and warm zones (semitropical and tropical climates) (Shi *et al.*, 2010). Soil texture was grouped as silt loam, sandy loam, clay and clay loam, loam, silty clay and silty clay loam, and loamy sand according to the USDA soil texture triangle. Soil depth was grouped as 0-10 cm, 10-20 cm, 20-50 cm, and 50-100 cm. Soil pH was grouped as acidic (< 6.6), neutral (6.6-7.3), and alkaline (> 7.3). Crop residue management was grouped as "residue returned" and "residue removed." We only included those studies that used the same residue management in the control and treatment groups. Similarly, nitrogen fertilization was grouped into no addition, low (1-100 kg N ha^{-1}), medium (101-200), and high levels (> 200). Irrigation management was grouped as irrigated or rainfed. Crop sequence was grouped as rotational or continuous crops (including crop-fallow systems). We also estimated the response of SOC in the whole-soil profiles (from the soil surface to 120 cm, with an interval of 10 cm) to CSA management practices.

The standard deviation (SD) of selected variables, an important input variable to the meta-analysis, was computed as SD = SE$\times\sqrt{n}$, where SE is the standard error and n is the number of observational replications. If the results of a study were reported without SD or SE, SD was calculated based on the average coefficient of variation for the known data. Publication bias was analyzed by the method of fail-safe number, which suggests that the meta-analysis can be considered robust if the fail-safe number is larger than 5*k+10 (where k is the number of observed studies) (Rothstein *et al.*, 2006).

2.2. Meta-analysis

A random-effect model of meta-analysis was used to explore environmental and management variables that might explain the response of SOC to CSA management practices. The data analysis was performed in R (R Development Core Team 2009). The response ratio (RR) was defined as the ratio between the outcome of CSA management practices and that of the control group. The logarithm of RR ($\ln RR$) was calculated as the effect size of each observation (Hedges *et al.*, 1999, Equation (1)):

$$\ln RR = \ln\,(\overline{X_t}/\overline{X_c}) = \ln \overline{X_t} - \ln \overline{X_c} \qquad (1)$$

where $\overline{X_t}$ and $\overline{X_c}$ are SOC values in the treatment and control groups, respectively. The variance (v) of $\ln RR$ was computed as:

$$\nu = \frac{S_t^2}{n_t \overline{X_t^2}} + \frac{S_c^2}{n_c \overline{X_c^2}} \quad (2)$$

where S_t and S_c are the standard deviations of the treatment and control groups, respectively, while n_t and n_c are the sample sizes of the treatment and control, respectively.

The weighting factor (w), as the inverse of the variance, was computed for each observation to obtain a final weighting factor (w'), which was then used to calculate the mean effect size (RR_{++}). The equations were:

$$w = 1 / \nu \quad (3)$$

$$w' = w / n \quad (4)$$

$$RR_{++} = \frac{\sum_i \ln RR'_i}{\sum_i w'_i} \quad (5)$$

where $\ln RR' = w' \ln RR$ is the weighted effect size, n is the total number of observations per study, and i is the ith observation.

The 95% confidence intervals (CI) of $\ln RR_{++}$ were computed to determine statistical significance. The comparison between treatment and control was considered significant if the 95% CIs did not overlap zero (vertical lines in the graphs). The percent change was transformed [($e^{RR_{++}}$-1) ×100%] to explain the response of the estimated CSA management practices.

3. Results

3.1 SOC responses to conservation tillage, cover crops, and biochar

Biochar applications enhanced SOC storage by 39% (28% in the field and 57% in incubation and pot experiments, Fig. S1), representing the most effective practice, followed by cover crops (6%) and conservation tillage (5%) (Fig. 2). Cover crop species had a pronounced positive effect on SOC sequestration (Fig. S1), ranging from 4% for non-leguminous cover crops to 9% for leguminous cover crops. When investigating different types of conservation tillage, NT and RT had similar effects on SOC (approximately 8% increase). All results were statistically significant (Fig. 2). Theoretically, the combination of CSA management practices may result in greater or lesser effects on soil sequestration compared to single CSA management practice. However, if synergistic effects were the prevalent interactions, this combination might potentially enhance carbon accumulation (e.g., over 50% increase in SOC), which is subject to further investigation in field experiments. Across the whole dataset we compiled, the SOC varied widely in each CSA treatment (Fig. S2). We calculated the distribution of the data points (the ratio of SOC of each treatment to that of the corresponding control, i.e., NT/RT vs. conventional tillage, cover crops vs. no cover crop, and biochar use vs. non-biochar; Fig. S2). Most of the studies used in this meta-analysis reported positive responses of SOC to NT, RT, cover crops, and biochar treatment (60%, 65%, 68%, and 91%, respectively). The SOC change rates were 0.38±0.71 Mg ha^{-1} yr^{-1} (n=56) and -0.29±0.79 Mg ha^{-1} yr^{-1} (n=30) in NT and RT systems, respectively (Fig. S3). We did not calculate SOC sequestration rates for other treatments (i.e., cover crops and biochar) due to the lack of some ancillary information (e.g., bulk density).

[Insert Figure 2]

3.2 Effects of CSA management practices in different climate zones

Overall, CSA management practices sequestered more SOC in arid areas than in humid areas (Fig. 3a). Biochar and cover crops increased 12% (38% vs. 26%) and 3% (9% vs. 6%) more SOC in arid areas, respectively, compared to humid areas. In comparison, the NT-induced SOC uptake was slightly higher in arid areas than that in humid areas (9% and 8%, respectively). However, the RT-induced SOC increment in arid areas was two times greater than that in humid areas. Our further analysis suggested that CSA management practices significantly increased SOC in both cool and warm climate zones with diverse responses (Fig. 3b). For example, in warm areas, biochar applications only increased SOC by half of the enhancement observed in cool areas. Cover crops increased SOC by 15% in warm areas, three times larger than that in cool areas. In warm areas, NT increased SOC by 15% compared to 8% in cool areas. Reduced tillage increased SOC by 7% and 6% in warm and cool areas, respectively.

[Insert Figure 3]

3.3 Effects of CSA management practices with different soil properties

The effects of CSA management practices on SOC were strongly influenced by soil texture (Fig. 4). Biochar applications increased SOC by 63, 62%, and 52% in silty clay and silty clay loam soils, loam soils, and loamy sand soils, respectively. While relatively lower soil carbon uptakes under biochar applications were found in clay loam and clay soils (32%), silt loam soils (35%), and sandy loam soils (34%). Cover crops increased SOC by 4%, 6%, 7%, and 6% in clay loam and clay soils, silt loam soils, loam soils, and sandy loam soils, respectively. No-till increased SOC by 16% in silty clay and silty clay loam soils, compared to 12% in sandy loam soils and 7% in loamy sand soils. Reduced tillage increased SOC by 21%, 7%, and 15% in silty clay and silty clay loam soils, loam soils, and loamy sand soils, respectively. Overall, cover crops sequestered more carbon in coarse-textured soils than in fine-textured soils. In contrast, NT and RT increased SOC more in fine-textured soils than in coarse-textured soils. No obvious relationship was found between biochar use and soil textures.

[Insert Figure 4]

The positive effects of CSA management practices on SOC decreased with soil depth (Fig. 5). Biochar significantly increased SOC by 41% and 14% in the 0-10 cm and 0-30 cm soil layers, respectively (Table S1). Cover crops significantly increased SOC by 9%, 3%, and 9% in the 0-10 cm, 10-20 cm, and 20-50 cm depth ranges, respectively. Further analysis showed that cover crops could increase SOC (5%) in the entire 0-70 cm soil profile (Table S1). Both NT and RT could significantly increase SOC most at 0-10 cm depth (22% and 17%, respectively). Although reduced SOC was observed in the 10-20 cm and 20-50 cm soil layers (-4% and -10%, respectively), NT could still enhance SOC sequestration in the entire soil profile up to 120 cm (Table S1). In comparison, RT could increase SOC in the 0-70 cm soil profile (Table S1) although decreased soil carbon (not statistically significant) was observed in the 10-50 cm soil layer (Fig. 5).

[Insert Figure 5]

All CSA management practices except RT positively influenced the SOC pool regardless of soil pH. The management-induced SOC uptake was generally higher in alkaline soils than in acid soils (Fig. 6). Biochar use increased SOC by 65%, 35%, and 28% in alkaline, neutral, and acid soils, respectively. Cover crops increased SOC by 15% in neutral soils, followed by alkaline (9%) and acid soils (6%). No-till increased SOC by 6% in acid soils and 13% in alkaline soils. The SOC increased by RT was greater in alkaline soils (9%) than acid soils (6%), but RT had no significant influence on SOC in neutral soils.

[Insert Figure 6]

3.4 Combined effects of experiment duration and other agronomic practices

The CSA management practices are generally applied together with other agronomic practices such as residue return, nitrogen fertilizer use, and irrigation. These agronomic practices may interact with the CSA management practices with positive or negative effects on the capacity of soils to sequester carbon. In this study, we considered experiment duration and four other agronomic practices, including residue return, nitrogen fertilization, irrigation, and crop sequence, to quantify these effects.

Our results demonstrated that the influences of three CSA management practices on SOC varied with experiment duration. Biochar amendments significantly increased SOC by 45% and 36% in short-term and medium-term experiments, respectively. Cover crops significantly increased SOC by 5%, 11%, and 20% in the short-term, medium-term, and long-term experiments, respectively (Fig. 7). No-till significantly increased SOC by 13% in the long-term experiments, followed by medium-term (7%) and short-term (6%). Reduced tillage increased SOC by 12% in long-term studies, followed by medium-term (9%) and short-term experiments (3%). The average durations differed in each group (Table S2), which may influence the effect of CSA management practices on SOC. When excluding short and medium experiment durations (≤ 20 years) and shallow sampling (< 20 cm), RT significantly increased SOC by 14%, while NT had no significant effect on SOC (Fig. S4).

[Insert Figure 7]

When crop residues were returned, conservation tillage and cover crops significantly increased SOC: 9% for NT, 6% for cover crops, and 5% for RT (Fig. 8). However, if crop residues were removed, neither cover crops nor RT had a significant effect on SOC, although there was a significant increase in SOC under NT (5%).

[Insert Figure 8]

Our results suggested that nitrogen fertilizer use could alter the magnitude of soil carbon uptake induced by CSA management practices. Biochar boosted the most SOC among CSA management practices regardless of nitrogen fertilizer levels, with the strongest effects under the low-level nitrogen inputs, followed by the high-level (38%), medium-level (29%), and no nitrogen fertilizer use (27%) (Fig. 9). Cover crops increased SOC by 6% under both low-level and medium-level nitrogen inputs, slightly higher than that under the high-level nitrogen fertilizer use (3%). No-till tended to sequester more soil carbon when nitrogen fertilizer input was relatively lower (11%, 8%, and 6% for low-level, medium-level, and high-level nitrogen fertilization, respectively). While RT increased SOC by 13% at the medium-level nitrogen fertilizer rate, approximately two times larger than those under the low-level and high-level nitrogen fertilizer use (Fig. 9).

[Insert Figure 9]

When investigating the irrigation effects, our results suggested that biochar markedly stimulated SOC increases in irrigated croplands (49%), three times higher than those under rainfed condition. Similarly, NT increased SOC by 15% in irrigated croplands, twice as much soil carbon as that in rainfed croplands. Cover crops increased SOC by 7% and 4% in irrigated and rainfed croplands, respectively. In contrast, the RT-induced SOC increase was 16% under the rainfed condition, 5% higher than that in irrigated croplands (Fig. 10a).

The CSA management practices significantly promoted SOC uptakes in both rotational and continuous cropping systems (Fig. 10b). Specifically, biochar amendments enhanced SOC by 52% in rotational cropping systems, much higher than that in the continuous cropping system (31%). While SOC uptakes induced by NT and RT showed no obvious differences in the rotational and continuous cropping systems (9% and 8% vs. 8% and 7%). Cover crops increased SOC by 4% in rotational cropping systems, lower than that in continuous cropping systems (8%).

[Insert Figure 10]

3.5 Combinations of CSA management practices

Our results demonstrated that combining different CSA management practices might significantly enhance SOC sequestration. In warm regions, SOC increased by 13% with the combination of conservation tillage and cover crops (Fig. 11). In loamy sand and sandy clay loam soils, associated SOC uptakes increased to 31% and 21%, respectively. A similar effect was also observed in medium-term experiments. However, in clay soils, the combination of cover crops and conservation tillage significantly decreased SOC by 19%.

[Insert Figure 11]

4. Discussion

4.1 Effects of CSA management practices on SOC

Common approaches for enhancing SOC focus on increasing carbon inputs, decreasing losses, or simultaneously affecting both inputs and losses. All CSA management practices discussed here, i.e., biochar, cover crops, and conservation tillage, increase soil carbon sequestration to different extents. For example, SOC enhancement by biochar applications can reach up to 40% (Liu *et al.*, 2016), while conservation tillage and cover crops increase SOC by only 3-10% (Luo *et al.*, 2010; Abdalla *et al.*, 2016; Du *et al.*, 2017b; Zhao *et al.*, 2017) and ~10% (Aguilera *et al.*, 2013), respectively. Our results agree with these earlier findings: biochar use increased SOC by 39%, followed by cover crops (6%) and conservation tillage (5%). The discrepancies among various CSA management practices in enhancing SOC fundamentally lie in their functional mechanisms. Biochar addition, with a low turnover rate, contributes directly to soil carbon storage and indirectly decreases native SOC decomposition rates by negative priming (Wang *et al.*, 2016). Cover crops are green manure that increases carbon inputs to the soil and subsequent SOC (Poeplau & Don, 2015). Conservation tillage practices may not necessarily add carbon; their contribution is primarily accomplished by protecting SOC from decomposition and erosion (Six *et al.*, 2000; Lal, 2005). Additionally, all three CSA management practices can potentially improve soil properties, thereby stimulating more carbon inputs from residue return and rhizodeposition due to promoted plant growth, and reducing carbon losses via decreasing leaching and erosion. However, the effectiveness of these practices on SOC sequestration and the mechanisms involved vary with environmental factors and other agronomic practices.

4.2 Environmental control in CSA management practices

Environmental factors such as climate and soil properties may influence carbon inputs to the soil and affect the processes that regulate carbon loss, considering that all CSA practices are implemented in site-specific climate and soil conditions. The effects of CSA management practices on SOC could be biased by environmental factors.

4.2.1 Climate variability

Climate is one of the major driving forces that regulate SOC distribution. On average, SOC accumulation is greater than decomposition in wet areas than in dry and warm regions (Jobbágy & Jackson, 2000). Soil carbon is positively related to precipitation and negatively correlated with temperature (Rusco *et al.*, 2001), with the former correlation tending to be stronger (Martin *et al.*, 2011; Meersmans *et al.*, 2011). High precipitation is usually associated with abundant growth and high rates of carbon inputs to soils (Luo *et al.*, 2017), while low temperatures may remarkably reduce microbial activity, resulting in low rates of organic matter decomposition and measurable amounts of SOC accumulation (Castro *et al.*, 1995; Garcia *et al.*, 2018). Biochar applications result in greater SOC accumulation in arid/cool areas than in humid/warm environments (Fig. 3), probably due to the porous structure and the capacity of biochar to promote greater soil water retention (Karhu *et al.*, 2011; Abel *et al.*, 2013). It is not clear why biochar has a greater impact on SOC accrual in cool regions. A possible explanation is that high soil temperatures may promote biochar decomposition and oxidation (Cheng *et al.*, 2008).

Cover crops and NT increased SOC with no significant difference between aridity conditions (Table 1), although they performed better at storing SOC in arid areas (Fig. 3a). This result suggests that arid-region soils have a high potential to store carbon when using proper management practices (Tondoh *et al.*, 2016). In addition, cover crops and NT can enhance carbon sequestration more in warm areas than in cool areas. Temperature could affect the establishment and growth of cover crops (Akemo *et al.*, 2000). In warm areas, cover crops may develop well and potentially capture more carbon dioxide (CO_2) from the atmosphere, thus providing more carbon inputs into soils after they die (e.g., Bayer *et al.*, 2009).

Tillage results in the breakdown of macroaggregates and the release of aggregate-protected SOC (Six *et al.*, 2000; Mikha & Rice, 2004). Tillage-induced SOC decomposition usually proceeds at higher rates in warm than in cool areas. Implementing NT, with minimal soil disturbance, protects SOC from decomposition. As a result, SOC increases can be more significant in warm conditions considering the relatively higher baseline of the decomposition rate compared to that in cool areas.

[Insert Table 1]

4.2.2 Soil properties

Soil organic carbon is strongly correlated with clay content, with an increasing trend toward more SOC in fine-textured soils (Stronkhorst & Venter, 2008; Meersmans *et al.*, 2012). The SOC mineralization rate probably diminishes as clay concentrations increase (Sainju *et al.*, 2002). Clay minerals can stabilize SOC against microbial attack through absorption of organic molecules (Ladd *et al.*, 1996). By binding organic matter, clay particles help form and stabilize soil aggregates, imposing a physical barrier between decomposer microflora and organic substrates and limiting water and oxygen available for decomposition (Dominy *et al.*, 2002).

Biochar use and cover crops promote carbon sequestration for all soil texture types. Such an enhancement of SOC does not vary significantly with soil texture (Table 1). The ability of conservation tillage to enhance SOC, however, differs with soil texture (Fig. 4). Conservation tillage merely reduces soil disturbance and normally does not add extra materials to soils. It can be inferred that the effect of conservation tillage on SOC is more texture-dependent than the other two management practices. Biochar is a carbon-rich material with a charged surface, organic functional groups, and a porous structure, which can potentially increase soil aggregation and cation exchange capacity (Jien & Wang, 2013). Similarly, cover crops directly provide carbon inputs to soils, and their root development and rhizodeposition can also benefit soil structure. These benefits are embedded in the source of biochar and cover crops *per se*. Thus, the effectiveness of biochar and cover crops in increasing SOC may depend on their properties other than soil texture.

Soil depth may potentially influence the effects of the CSA practices on SOC (Baker *et al.*, 2007). The CSA practices were most beneficial to SOC accumulation in surface soils. For example, NT increased SOC by 7% in the 0-3 cm soil layer (Abdalla *et al.*, 2016) and by 3% at the 40 cm depth (Luo *et al.*, 2010). Our findings suggested that CSA practices can enhance SOC sequestration in the entire soil profile, although the positive effects vary with soil depths (Table S1). Conventional tillage breaks soil aggregates and increases aeration and thus enhances soil organic matter mineralization (Cambardella & Elliott, 1993). Conventional tillage also incorporates residues into deeper soil layers, resulting in a more uniform distribution of SOC (albeit at lower concentrations) in the soil profile (Sainju *et al.*, 2006; Plaza-Bonilla *et al.*, 2010). In contrast, conservation tillage keeps residues at the soil surface and reduces their degree of incorporation into soil (Franzluebbers *et al.*, 1995). Nevertheless, positive effects of NT on SOC have been found in a deep soil profile (0-60 cm, Liu *et al.*, 2014). As noted, in the 10-50 cm soil layer, the effect of cover crops on SOC was found to be the greatest among all the CSA management practices we discussed (Fig. 5). This is perhaps because much of the crop and cover crop root growth occurs in the surface soil (e.g., Box & Ramsuer, 1993; Sainju *et al.*, 1998) and the generally greater contribution of roots to SOC than aboveground biomass (Balesdent & Balabane, 1996; Allmaras *et al.*, 2004).

Soil pH is recognized as a dominant factor governing the soil organic matter turnover rate, although its mode of impact is still unclear (Van Bergen *et al.*, 1998). Soil pH affects selective presentation or metabolic modification of specific components (e.g., lignin-cellulose, lipids) during decomposition (Kemmitt *et al.*, 2006) and therefore abiotic factors (e.g., carbon and nutrient availability) and biotic factors (e.g., the composition of the microbial community). Also, soil pH can change the decomposition rate of crop residues and SOC via its effect on SOC solubility and indirectly by altering microbial growth, activity, and community structure (Pietri & Brookes, 2009; Wang *et al.*, 2017). The levels of soluble organic carbon may increase with increasing acidity (Willett *et al.*, 2004; Kemmitt *et al.*, 2006). Motavalli *et al.* (1995) suggested that increased soil acidity would cause greater soil organic matter accumulation due to reduced microbial mineralization; however, this was challenged by Kemmitt *et al.* (2006) who found no significant trend in SOC in response to pH changes. In this study, most CSA management practices resulted in greater increases in SOC in neutral or alkaline soils compared to acid soils.

4.3 CSA and other agronomic practices

Crop residues provide substantial amounts of organic matter and may influence the effect of CSA practices on SOC. Residue retention changes the formation of soil macroaggregates (Benbi & Senapati, 2010), promoting SOC preservation and accumulation (Six *et al.*, 2002). Residue cover protects the soil surface from direct impact by raindrops (Blanco-Canqui *et al.*, 2014). In addition, crop residues provide organic substrates to soil microorganisms that can produce binding agents and promote soil aggregation (Guggenberger *et al.*, 1999). Conversely, residue removal reduces carbon input to the soil system and ultimately decreases SOC storage (Manna *et al.*, 2005; Koga & Tsuji, 2009). This suggests that the amount of carbon inputs predominantly controls changes in SOC stocks (Virto *et al.*, 2012). For the conditions of cover crops and NT, enhancing SOC was significantly greater with residue return than with residue removal. Our study suggests that changes in SOC did not differ with residue management in RT (Table 1), although a slightly greater increase in SOC occurred with residue retention than with residue removal (Fig. 8). This unexpected result is likely due to the limited number of observations with residue removal. Another possible reason is that the interaction between residue management and soil type may lead to various responses in SOC stocks. For example, residue removal increased SOC by 3.6% while residue retention had no effect on SOC in clay and clay loam soils. The decomposition of crop residues involves complex processes, which are controlled by multiple biogeochemical and biophysical conditions.

Nitrogen fertilization noticeably increases SOC stock but with diminishing returns. For example, Blanco-Canqui *et al.* (2014) indicate that nitrogen fertilizer increases SOC when the nitrogen fertilization rate is below 80 kg N ha^{-1}, above which it reduces aggregation and then decreases SOC stocks. Nitrogen fertilization can stimulate biological activity by altering carbon/nitrogen ratios, thereby promoting soil respiration and decreasing SOC content (Mulvaney *et al.*, 2009); however, excessive nitrogen addition may reduce soil fungi populations, inhibit soil enzyme activity, and decrease CO_2 emissions (Wilson & Al Kazi, 2008). These findings suggest that nitrogen fertilization enhances the positive effect of CSA management practices on SOC, likely through increased plant biomass production (Gregorich *et al.*, 1996). However, nitrogen addition complicates the effects of biochar on SOC (Fig. 9). Nitrogen fertilizer may affect biochar stability and the response of native SOC decomposition to biochar addition (Jiang *et al.*, 2016). Positive (Bebber *et al.*, 2011; Jiang *et al.*, 2014) and negative (Pregitzer *et al.*, 2008) effects of nitrogen on SOC mineralization rates have been reported. These contrasting effects could be an alleviation of microbial nitrogen limitations (Jiang *et al.*, 2016) and changes in the microbial decomposer community toward more efficient carbon-users (Janssens *et al.*, 2010). A possible explanation of the various responses of nitrogen rate in biochar-modified soils is that either inadequate or excessive nitrogen addition may inhibit microbial activity to some extent, whereas medium-level nitrogen fertilization rates benefit microbes the most, which needs to be confirmed in future research.

Aridity can limit plant growth and crop residue return and ultimately compromise SOC accumulation (Moreno *et al.*, 2006). Jien and Wang (2013) suggest that CSA management practices can potentially enhance soil water retention by improving soil porosity and erosion control. Irrigation ensures sufficient water for plant growth, resulting in more biomass production than in rainfed conditions (Shipitalo *et al.*, 1990; Chan, 2004; Capowiez *et al.*, 2009; Swanepoel *et al.*, 2016). The crop root density is much higher in irrigated conditions compared to rainfed conditions (Jobbágy & Jackson, 2000), leading to higher organic matter input. Thus, CSA management practices in combination with irrigation could further increase SOC content.

Rotational cropping potentially provides high carbon input to soils. Compared to continuous cropping systems, crops in rotational cropping systems have a greater belowground allocation of biomass (Van Eerd *et al.*, 2014), resulting in more inputs of crop residue to the soil system. Enhancing rotation complexity can benefit carbon sequestration (West & Post, 2002). The present analysis suggests that all CSA practices can prominently increase SOC sequestration regardless of the crop rotation system. Biochar addition increased SOC more in rotational cropping systems than in continuous cropping systems, while cover crops increased SOC more in continuous systems (Fig. 10). This is likely because cover crops increased the diversity of the original continuous systems, resulting in larger percentage changes in SOC content compared to rotational systems. Cover crop species introduce large uncertainties because the quantity and quality of cover crop residues may vary greatly with species. Residues with a high carbon/nitrogen ratio probably increase the amount of SOC (Duong *et al.*, 2009). The growth period of legume cover crops may be longer in continuous than in rotational cropping systems, thus providing more organic matter and nitrogen input to the soil. Ultimately, these processes would increase SOC stocks.

The effect size of combined cover crops and conservation tillage was generally less than 11% (the sum of the effect size of cover crops and conservation tillage). However, in sandy clay loam and loamy sand soils, the sum of the effect size was 21% and 31%, respectively. Coarse-textured soils are not carbon-saturated and have great potential for carbon uptake. Cultivated land tends to suffer from SOC degradation, and SOC accumulation could quickly increase upon initiating farming practices due to high carbon inputs to the soil system (Vieira *et al.*, 2009). For example, in sandy loam soils, Higashi *et al.* (2014) showed that SOC increased by 22% with a combination of cover crops and NT. These results may be attributed to the stability of soil water-stable aggregates when cover crops are grown in sandy clay loam soils (McVay *et al.*, 1989), given that aggregate stability has been linked to protection of SOC from mineralization (Unger, 1997). The combination of cover crops and conservation tillage significantly decreased SOC in clay soils. The reason for this unexpected result may be due to the limited number of study sites where this combination of treatments was evaluated (few data points in our meta-analysis) but also to the diverse methods (e.g., burning) by which the cover crop biomass was managed (Tian *et al.*, 2005).

4.4 Uncertainty analysis and prospects

Our meta-analysis, based on 3,049-paired comparisons from 417 peer-reviewed articles, quantitatively analyzed SOC changes as influenced by major CSA management practices and associated environmental factors and other agronomic practices. The publication bias analysis suggested that most results in this study are robust (Table S3). The accuracy and robustness of metadata analysis depend highly on both the data quality and quantity. A detailed statement of the experimental conditions will provide more information for in-depth analysis. Future CSA research also requires standardized field management, for example, the definitions and names of different conservation tillage methods should be uniform across studies to facilitate classification research.

To the best of our knowledge, this study made the first attempt to examine synergistic effects when two or more CSA management practices are used together. Although our results present the positive effects of CSA management on soil carbon storage, especially when multiple management practices are adopted collectively, each practice may have constraints regarding enhancing soil carbon sequestration. The SOC benefit of CSA management practices strongly depends on environmental factors and other agronomic practices. Therefore, the choice of proper practices is potentially highly region-specific. Our results imply that CSA may have great potential for climate change mitigation as the combination of conservation tillage, cover crops, and biochar can theoretically enhance SOC by 50%. However, field experiments are still needed to support this claim. In addition, some CSA management practices may promote nitrous oxide or methane emissions (e.g., Six *et al.*, 2004; Spokas & Reicosky, 2009; Kessel *et al.*, 2013; Huang *et al.,* 2018), which, to some extent, would offset their benefit on climate change mitigation. Therefore, evaluating the CSA effects should also include non-CO_2 greenhouse gases such as nitrous oxide and methane. We call for field experiments that can fully examine key indicators (such as soil carbon and greenhouse gases) in response to single and combined CSA management practices.

Additionally, incorporating cover crops into current cropping systems could potentially alter conventional rotations. For example, cover crops in herbaceous crop rotations can substitute bare fallows or commercial crops. We only considered studies that treated cover crops as treatments and fallow (or weeds) as controls in this study. In comparison to bare fallows, cover crops can enhance soil health and quality (Jarecki & Lal, 2003). The benefits of cover crops include uptakes and stores of soil nutrients between seasons when they are susceptible to leaching (Doran & Smith, 1987). However, the substitution of commercial crops could reduce the productivity of the system, which has climatic implications related to the opportunity cost of the extra land required (e.g., Balmford *et al.*, 2018; Searchinger *et al.*, 2018). Thus, future studies should further address these potential side effects caused by land use change.

Materials producing biochar may have other uses or fates, and the biochar-making processes may produce CO_2 (e.g., Llorach-Massana *et al.*, 2017), although biochar addition is an effective way to sequester SOC. These uncertainties, to some extent, can offset the benefits of biochar for climate change mitigation through SOC sequestration (Powlson *et al.*, 2008). The carbon footprint of biochar production depends on production technology and the types of feedstocks (Meyer *et al.*, 2017). Mukherjee and Lal (2014) found that "carbon dioxide emissions from biochar-amended soils have been enhanced up to 61% compared with unamended soils." However, with a low carbon footprint, each ton of biochar could sequester 21 to 155 kg of equivalent CO_2 (Llorach-Massana *et al.*, 2017). Matovic (2011) also suggested that 4.8 Gt C yr^{-1} would be sequestered if 10% of the world's net primary production were converted into biochar, "at 50% yield and 30% energy from volatiles." To fully understand the net impacts of biochar on climate mitigation, future studies should stress the carbon footprint in the lifecycle of biochar.

It is essential to realistically examine the effects of CSA management practices on SOC and greenhouse gases at multiple scales from plot and field levels to regional and global scales. Therefore, future CSA research is expected to include varied climate and geographic conditions, address more biogeochemical and hydrological processes, and apply diverse methods such as the data-model fusion approach. For example, modeling studies have attempted to investigate regional cropland SOC dynamics as influenced by multiple global environmental changes while considering more traditional and less CSA practices (e.g., Molina *et al.,* 2017; Nash *et al.,* 2018; Ren *et al.,* 2012, 2018). In the future, ecosystem models need to be improved to incorporate multiple common CSA management practices. Additional model evaluations are needed to quantify the potential of cropland carbon sequestration by adopting multiple CSA practices at broad scales as new data become available from suggested field experiments and observations.

Acknowledgements

This work was supported by the National Institute of Food and Agriculture, U.S. Department of Agriculture (NIFA-USDA Hatch project 2352437000) and the National Science Foundation under Cooperative Agreement No.1355438. Bai acknowledges the support of the China Scholarship Council. We thank Elisa D'Angelo for the comments and suggestions on the manuscript. We also thank three anonymous reviewers and editor for comments that improved this manuscript. The authors declare no conflict of interest.

References

Abdalla, K., Chivenge, P., Ciais, P., & Chaplot, V. (2016). No-tillage lessens soil CO_2 emissions the most under arid and sandy soil conditions: results from a meta-analysis. *Biogeosciences*, 13(12), 3619-3633. doi: 10.5194/bg-13-3619-2016

Abel, S., Peters, A., Trinks, S., Schonsky, H., Facklam, M., & Wessolek, G. (2013). Impact of biochar and hydrochar addition on water retention and water repellency of sandy soil. *Geoderma*, 202-203(2), 183-191.

Agegnehu, G., Bass, A. M., Nelson, P. N., & Bird, M. I. (2016). Benefits of biochar, compost and biochar-compost for soil quality, maize yield and greenhouse gas emissions in a tropical agricultural soil. *Science of The Total Environment*, 543, 295-306. doi: 10.1016/j.scitotenv.2015.11.054

Aguilera, E., Lassaletta, L., Gattinger, A., & Gimeno, B. S. (2013). Managing soil carbon for climate change mitigation and adaptation in Mediterranean cropping systems: a meta-analysis. *Agriculture Ecosystems & Environment*, 168, 25-36. doi: 10.1016/j.agee.2013.02.003

Akemo, M. C., Regnier, E. E., & Bennett, M. A. (2000). Weed suppression in spring-sown rye (*Secale cereale*)–pea (*Pisum sativum*) cover crop mixes. *Weed Technology*, 14(3), 545-549.

Allmaras, R. R., Linden, D. R., & Clapp, C. (2004). Corn-residue transformations into root and soil carbon as related to nitrogen, tillage, and stover management. *Soil Science Society of America Journal*, 68(4), 1366-1375.

Ashworth, A. J., Allen, F. L., Wight, J. P., Saxton, A. M., & Tyler, D. D. (2014). Long-term Soil organic carbon changes as affected by crop rotation and bio-covers in no-till crop systems. In A. E. Hartemink & K. McSweeney (Eds.), *Soil Carbon* (pp. 271-279).

Baker, J. M., Ochsner, T. E., Venterea, R. T., & Griffis, T. J. (2007). Tillage and soil carbon sequestration—What do we really know? *Agriculture Ecosystems & Environment*, 118(1), 1-5.

Balesdent, J., & Balabane, M. (1996). Major contribution of roots to soil carbon storage inferred from maize cultivated soils. *Soil Biology & Biochemistry*, 28(9), 1261-1263.

Balmford, A., Amano, T., Bartlett, H., Chadwick, D., Collins, A., Edwards, D., . . . Smith, P. (2018). The environmental costs and benefits of high-yield farming. *Nature Sustainability*, 1(9), 477.

Bayer, C., Dieckow, J., Amado, T. J. C., Eltz, F. L. F., & Vieira, F. C. B. (2009). Cover crop effects increasing carbon storage in a subtropical no-till sandy Acrisol. *Communications in Soil Science and Plant Analysis*, 40(9-10), 1499-1511.

Bebber, D. P., Watkinson, S. C., Boddy, L., & Darrah, P. R. (2011). Simulated nitrogen deposition affects wood decomposition by cord-forming fungi. *Oecologia*, 167(4), 1177-1184.

Benbi, D. K., & Senapati, N. (2010). Soil aggregation and carbon and nitrogen stabilization in relation to residue and manure application in rice–wheat systems in northwest India. *Nutrient Cycling in Agroecosystems*, 87(2), 233-247.

Blanco-Canqui, H., Ferguson, R. B., Shapiro, C. A., Drijber, R. A., & Walters, D. T. (2014). Does inorganic nitrogen fertilization improve soil aggregation? Insights from two long-term tillage experiments. *Journal of Environmental Quality*, 43(3), 995-1003. doi: 10.2134/jeq2013.10.0431

Blanco-Canqui, H., Holman, J. D., Schlegel, A. J., Tatarko, J., & Shaver, T. M. (2013). Replacing fallow with cover crops in a semiarid soil: effects on soil properties. *Soil Science Society of America Journal*, 77(3), 1026-1034. doi: 10.2136/sssaj2013.01.0006

Blanco-Canqui, H., Mikha, M. M., Presley, D. R., & Claassen, M. M. (2011). Addition of cover crops enhances no-till potential for improving soil physical properties. *Soil Science Society of America Journal*, 75(4), 1471-1482. doi: 10.2136/sssaj2010.0430

Box, J., & Ramsuer, E. (1993). Minirhizotron wheat root data: comparisons to soil core root data. *Agronomy Journal*, 85(5), 1058-1060.

Briones, M. J. I., & Schmidt, O. (2017). Conventional tillage decreases the abundance and biomass of earthworms and alters their community structure in a global meta-analysis. *Global Change Biology*, 23(10), 4396-4419. doi: 10.1111/gcb.13744

Cambardella, C., & Elliott, E. (1993). Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. *Soil Science Society of America Journal*, 57(4), 1071-1076.

Campbell, B. M., Thornton, P., Zougmoré, R., Asten, P. V., & Lipper, L. (2014). Sustainable intensification: what is its role in climate smart agriculture? *Current Opinion in Environmental Sustainability*, 8(8), 39-43.

Capowiez, Y., Cadoux, S., Bouchant, P., Ruy, S., Roger-Estrade, J., Richard, G., & Boizard, H. (2009). The effect of tillage type and cropping system on earthworm communities, macroporosity and water infiltration. *Soil & Tillage Research*, 105(2), 209-216.

Castro, M. S., Steudler, P. A., Melillo, J. M., Aber, J. D., & Bowden, R. D. (1995). Factors controlling atmospheric methane consumption by temperate forest soils. *Global Biogeochemical Cycles*, 9(1), 1-10.

Chan, K. Y. (2004). Impact of tillage practices and burrows of a native Australian anecic earthworm on soil hydrology. *Applied Soil Ecology*, 27(1), 89-96.

Chen, H. Q., Marhan, S., Billen, N., & Stahr, K. (2009). Soil organic-carbon and total nitrogen stocks as affected by different land uses in Baden-Wurttemberg (southwest Germany). *Journal of Plant Nutrition and Soil Science*, 172(1), 32-42. doi: 10.1002/jpln.200700116

Cheng, C. H., Lehmann, J., Thies, J. E., & Burton, S. D. (2008). Stability of black carbon in soils across a climatic gradient. *Journal of Geophysical Research Biogeosciences*, 113(G2), 50-55.

Clark, K. M., Boardman, D. L., Staples, J. S., Easterby, S., Reinbott, T. M., Kremer, R. J., . . . Veum, K. S. (2017). Crop yield and soil organic carbon in conventional and no-till organic systems on a claypan soil. *Agronomy Journal*, 109(2), 588-599. doi: 10.2134/agronj2016.06.0367

De Baets, S., Poesen, J., Meersmans, J., & Serlet, L. (2011). Cover crops and their erosion-reducing effects during concentrated flow erosion. *Catena*, 85(3), 237-244.

Denef, K., Zotarelli, L., Boddey, R. M., & Six, J. (2007). Microaggregate-associated carbon as a diagnostic fraction for management-induced changes in soil organic carbon in two Oxisols. *Soil Biology & Biochemistry*, 39(5), 1165-1172. doi: 10.1016/j.soilbio.2006.12.024

Dominy, C., Haynes, R., & Van Antwerpen, R. (2002). Loss of soil organic matter and related soil properties under long-term sugarcane production on two contrasting soils. *Biology and Fertility of Soils*, 36(5), 350-356.

Doran, J., & Smith, M. (1987). Organic matter management and utilization of soil and fertilizer nutrients. *Soil fertility and organic matter as critical components of production systems*, 53-72.

Du, Z. L., Angers, D. A., Ren, T. S., Zhang, Q. Z., & Li, G. C. (2017b). The effect of no-till on organic C storage in Chinese soils should not be overemphasized: a meta-analysis. *Agriculture Ecosystems & Environment*, 236, 1-11. doi: 10.1016/j.agee.2016.11.007

Du, Z. L., Zhao, J. K., Wang, Y. D., & Zhang, Q. Z. (2017a). Biochar addition drives soil aggregation and carbon sequestration in aggregate fractions from an intensive agricultural system. *Journal of Soils and Sediments*, 17(3), 581-589. doi: 10.1007/s11368-015-1349-2

Duong, T., Baumann, K., & Marschner, P. (2009). Frequent addition of wheat straw residues to soil enhances carbon mineralization rate. *Soil Biology & Biochemistry*, 41(7), 1475-1482.

Duval, M. E., Galantini, J. A., Capurro, J. E., & Martinez, J. M. (2016). Winter cover crops in soybean monoculture: effects on soil organic carbon and its fractions. *Soil & Tillage Research*, 161, 95-105. doi: 10.1016/j.still.2016.04.006

FAO. (2013). Climate-smart Agriculture: Sourcebook. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Franzluebbers, A., Hons, F., & Zuberer, D. (1995). Tillage and crop effects on seasonal soil carbon and nitrogen dynamics. *Soil Science Society of America Journal*, 59(6), 1618-1624.

Fungo, B., Lehmann, J., Kalbitz, K., Thiongo, M., Okeyo, I., Tenywa, M., & Neufeldt, H. (2017). Aggregate size distribution in a biochar-amended tropical Ultisol under conventional hand-hoe tillage. *Soil & Tillage Research*, 165, 190-197. doi: 10.1016/j.still.2016.08.012

Garcia, C., Nannipieri, P., & Hernandez, T. (2018). Chapter 9 - The future of soil carbon. In C. Garcia, P. Nannipieri & T. Hernandez (Eds.), *The Future of Soil Carbon* (pp. 239-267): Academic Press.

Gregorich, E., Liang, B., Ellert, B., & Drury, C. (1996). Fertilization effects on soil organic matter turnover and corn residue C storage. *Soil Science Society of America Journal*, 60(2), 472-476.

Guggenberger, G., Frey, S. D., Six, J., Paustian, K., & Elliott, E. T. (1999). Bacterial and fungal cell-wall residues in conventional and no-tillage agroecosystems. *Soil Science Society of America Journal*, 63(5), 1188-1198.

Hassink, J. (1997). The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant and Soil*, 191(1), 77-87.

Hedges, L. V., Gurevitch, J., & Curtis, P. S. (1999). The meta-analysis of response ratios in experimental ecology. *Ecology*, 80(4), 1150-1156.

Higashi, T., Yunghui, M., Komatsuzaki, M., Miura, S., Hirata, T., Araki, H., . . . Ohta, H. (2014). Tillage and cover crop species affect soil organic carbon in Andosol, Kanto, Japan. *Soil & Tillage Research*, 138, 64-72. doi: 10.1016/j.still.2013.12.010

Huang, Y., Ren, W., Wang, L., Hui, D., Grove, J. H., Yang, X., ... & Goff, B. (2018). Greenhouse gas emissions and crop yield in no-tillage systems: A meta-analysis. *Agriculture, Ecosystems & Environment*, 268, 144-153.

Janssens, I., Dieleman, W., Luyssaert, S., Subke, J. A., Reichstein, M., Ceulemans, R., . . . Matteucci, G. (2010). Reduction of forest soil respiration in response to nitrogen deposition. *Nature Geoscience*, 3(5), 315.

Jarecki, M. K., & Lal, R. (2003). Crop management for soil carbon sequestration. *Critical Reviews in Plant Sciences*, 22(6), 471-502. doi: 10.1080/07352680390253179

Jiang, X., Cao, L., & Zhang, R. (2014). Changes of labile and recalcitrant carbon pools under nitrogen addition in a city lawn soil. *Journal of Soils and Sediments*, 14(3), 515-524.

Jiang, X., Haddix, M. L., & Cotrufo, M. F. (2016). Interactions between biochar and soil organic carbon decomposition: effects of nitrogen and low molecular weight carbon compound addition. *Soil Biology and Biochemistry*, 100, 92-101.

Jien, S. H., & Wang, C. S. (2013). Effects of biochar on soil properties and erosion potential in a highly weathered soil. *Catena*, 110(110), 225-233.

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

Karhu, K., Mattila, T., Bergström, I., & Regina, K. (2011). Biochar addition to agricultural soil increased CH_4 uptake and water holding capacity - Results from a short-term pilot field study. *Agriculture Ecosystems & Environment*, 140(1-2), 309-313.

Kemmitt, S. J., Wright, D., Goulding, K. W., & Jones, D. L. (2006). pH regulation of carbon and nitrogen dynamics in two agricultural soils. *Soil Biology and Biochemistry*, 38(5), 898-911.

Kessel, C., Venterea, R., Six, J., Adviento-Borbe, M. A., Linquist, B., & Groenigen, K. J. (2013). Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: a meta-analysis. *Global Change Biology*, 19(1), 33-44.

Koga, N., & Tsuji, H. (2009). Effects of reduced tillage, crop residue management and manure application practices on crop yields and soil carbon sequestration on an Andisol in northern Japan. *Soil Science & Plant Nutrition*, 55(4), 546-557.

Ladd, J. N., Foster, R. C., Nannipieri, P., & Oades, J. (1996). Soil structure and biological activity. *Soil Biochemistry*, 9, 23-78.

Lal, R. (2004). Soil carbon sequestration to mitigate climate change. *Geoderma*, 123(1), 1-22.

Lal, R. (2005). Soil erosion and carbon dynamics. *Soil & Tillage Research*, 81(2), 137-142.

Lal, R. (2018). Digging deeper: A holistic perspective of factors affecting soil organic carbon sequestration in agroecosystems. *Global Change Biology*, 24(8):3285-3301. doi: 10.1111/gcb.14054

Lavelle, P., Brussaard, L., & Hendrix, P. (1999). *Earthworm Management in Tropical Agroecosystems*. Biddles Ud, Guildford and King's Lynn, UK.

Liang, A. Z., Zhang, X. P., Fang, H. J., Yang, X. M., & Drury, C. F. (2007). Short-term effects of tillage practices on organic carbon in clay loam soil of northeast China. *Pedosphere*, 17(5), 619-623. doi: 10.1016/s1002-0160(07)60073-3

Liang, C., & Balser, T. C. (2012). Warming and nitrogen deposition lessen microbial residue contribution to soil carbon pool. *Nature Communications*, 3, 1222. doi: 10.1038/ncomms2224

Lipper, L., Thornton, P., Campbell, B. M., Baedeker, T., Braimoh, A., Bwalya, M., . . . Henry, K. (2014). Climate-smart agriculture for food security. *Nature Climate Change*, 4(12), 1068–1072.

Liu, E. K., Teclemariam, S. G., Yan, C. R., Yu, J. M., Gu, R. S., Liu, S., . . . Liu, Q. (2014). Long-term effects of no-tillage management practice on soil organic carbon and its fractions in the northern China. *Geoderma*, 213, 379-384. doi: 10.1016/j.geoderma.2013.08.021

Liu, S. W., Zhang, Y. J., Zong, Y. J., Hu, Z. Q., Wu, S., Zhou, J., . . . Zou, J. W. (2016). Response of soil carbon dioxide fluxes, soil organic carbon and microbial biomass carbon to biochar amendment: a meta-analysis. *Global Change Biology Bioenergy*, 8(2), 392-406. doi: 10.1111/gcbb.12265

Llorach-Massana, P., Lopez-Capel, E., Peña, J., Rieradevall, J., Montero, J. I., & Puy, N. (2017). Technical feasibility and carbon footprint of biochar co-production with tomato plant residue. *Waste Management*, 67, 121-130.

Lorenz, K., & Lal, R. (2016). Soil organic carbon: an appropriate indicator to monitor trends of land and soil degradation within the SDG framework. Dessau-Roßlau, Germany.

Luo, Z., Feng, W., Luo, Y., Baldock, J., & Wang, E. (2017). Soil organic carbon dynamics jointly controlled by climate, carbon inputs, soil properties and soil carbon fractions. *Global Change Biology*, 23(10), 4430-4439. doi: 10.1111/gcb.13767

Luo, Z. K., Wang, E. L., & Sun, O. J. (2010). Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. *Agriculture Ecosystems & Environment*, 139(1-2), 224-231. doi: 10.1016/j.agee.2010.08.006

Manna, M. C., Swarup, A., Wanjari, R. H., Ravankar, H. N., Mishra, B., Saha, M. N., . . . Sarap, P. A. (2005). Long-term effect of fertilizer and manure application on soil organic carbon storage, soil quality and yield sustainability under sub-humid and semi-arid tropical India. *Field Crops Research*, 93(2-3), 264-280. doi: 10.1016/j.fcr.2004.10.006

Martin, M. P., Wattenbach, M., Smith, P., & Meersmans, J. (2011). Spatial distribution of soil organic carbon stocks in France. *Biogeosciences*, 8(5), 1053-1065.

Matovic, D. (2011). Biochar as a viable carbon sequestration option: Global and Canadian perspective. *Energy*, 36(4).

McVay, K., Radcliffe, D., & Hargrove, W. (1989). Winter legume effects on soil properties and nitrogen fertilizer requirements. *Soil Science Society of America Journal*, 53(6), 1856-1862.

Meersmans, J., Martin, M. P., De Ridder, F., Lacarce, E., Wetterlind, J., De Baets, S., . . . Bispo, A. (2012). A novel soil organic C model using climate, soil type and management data at the national scale in France. *Agronomy for Sustainable Development*, 32(4), 873-888.

Meersmans, J., Van, W. B., Goidts, E., Van, M. M., De, B. S., & De, R. F. (2011). Spatial analysis of soil organic carbon evolution in Belgian croplands and grasslands, 1960-2006. *Global Change Biology*, 17(1), 466-479.

Meyer, S., Genesio, L., Vogel, I., Schmidt, H.-P., Soja, G., Someus, E., . . . Glaser, B. (2017). Biochar standardization and legislation harmonization. *Journal of Environmental Engineering and Landscape Management*, 25(2), 175-191.

Mikha, M. M., & Rice, C. W. (2004). Tillage and manure effects on soil and aggregate-associated carbon and nitrogen. *Soil Science Society of America Journal*, 68(3), 809-816.

Molina, L. G., Moreno Pérez, E. D. C., & Pérez, A. B. (2017). Simulation of soil organic carbon changes in Vertisols under conservation tillage using the RothC model. *Scientia Agricola*, 74(3), 235-241.

Moreno, F., Murillo, J., Pelegrín, F., & Girón, I. (2006). Long-term impact of conservation tillage on stratification ratio of soil organic carbon and loss of total and active $CaCO_3$. *Soil & Tillage Research*, 85(1), 86-93.

Motavalli, P., Palm, C., Parton, W., Elliott, E., & Frey, S. (1995). Soil pH and organic C dynamics in tropical forest soils: evidence from laboratory and simulation studies. *Soil Biology and Biochemistry*, 27(12), 1589-1599.

Mukherjee, A., & Lal, R. (2014). The biochar dilemma. *Soil Research*, 52(3), 217-230.

Mulvaney, R., Khan, S., & Ellsworth, T. (2009). Synthetic nitrogen fertilizers deplete soil nitrogen: a global dilemma for sustainable cereal production. *Journal of Environmental Quality*, 38(6), 2295-2314.

Nash, P. R., Gollany, H. T., Liebig, M. A., Halvorson, J. J., Archer, D. W., & Tanaka, D. L. (2018). Simulated soil organic carbon responses to crop rotation, tillage, and climate change in North Dakota. *Journal of Environmental Quality*, 47(4), 654-662.

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G. P., & Smith, P. (2016). Climate-smart soils. *Nature*, 532(7597), 49-57.

Pietri, J. A., & Brookes, P. (2009). Substrate inputs and pH as factors controlling microbial biomass, activity and community structure in an arable soil. *Soil Biology and Biochemistry*, 41(7), 1396-1405.

Plaza-Bonilla, D., Cantero-Martinez, C., & Alvaro-Fuentes, J. (2010). Tillage effects on soil aggregation and soil organic carbon profile distribution under Mediterranean semi-arid conditions. *Soil Use and Management*, 26(4), 465-474. doi: 10.1111/j.1475-2743.2010.00298.x

Poeplau, C., & Don, A. (2015). Carbon sequestration in agricultural soils via cultivation of cover crops—a meta-analysis. *Agriculture, Ecosystems & Environment*, 200, 33-41. doi: 10.1016/j.agee.2014.10.024

Powlson, D. S., Riche, A. B., Coleman, K., Glendining, M. J., & Whitmore, A. P. (2008). Carbon sequestration in European soils through straw incorporation: limitations and alternatives. *Waste Management*, 28(4), 741-746.

Pregitzer, K. S., Burton, A. J., Zak, D. R., & Talhelm, A. F. (2008). Simulated chronic nitrogen deposition increases carbon storage in Northern Temperate forests. *Global Change Biology*, 14(1), 142-153.

Ren, W., Tian, H., Tao, B., Huang, Y., & Pan, S. (2012). China's crop productivity and soil carbon storage as influenced by multifactor global change. *Global Change Biology*, 18(9), 2945-2957.

Ren W. (2018) Towards an Integrated Agroecosystem Modeling Approach for Climate-Smart Agricultural Management. In: Wendroth, O., R.J. Lascano, and L. Ma (Eds.) Bridging among disciplines by synthesizing soil and plant processes. *Advances in Agricultural Systems Modeling*, Volume 8: 127-144, ASA, CSSA, SSSA, Madison, WI 53711-5801, USA.

Rothstein, H. R., Sutton, A. J., & Borenstein, M. (Eds.) (2006). *Publication Bias in Meta-Analysis: Prevention, Assessment and Adjustments*. John Wiley & Sons.

Rusco, E., Jones, R., & Bidoglio, G. (2001). Organic matter in the soils of Europe: Present status and future trends. European Soil Bureau, Soil and Waste Unit, Institute for Environment and Sustainability, JRC Ispra Institute, Joint Research Centre European Commission, Italy.

Sainju, U., Singh, B., & Whitehead, W. (2002). Long-term effects of tillage, cover crops, and nitrogen fertilization on organic carbon and nitrogen concentrations in sandy loam soils in Georgia, USA. *Soil & Tillage Research*, 63(3), 167-179.

Sainju, U., Whitehead, W., & Singh, B. (2003). Cover crops and nitrogen fertilization effects on soil aggregation and carbon and nitrogen pools. *Canadian Journal of Soil Science*, 83(2), 155-165.

Sainju, U. M., Singh, B. P., & Whitehead, W. F. (1998). Cover crop root distribution and its effects on soil nitrogen cycling. *Agronomy Journal*, 90(4), 511-518.

Sainju, U. M., Singh, B. P., Whitehead, W. F., & Wang, S. (2006). Carbon supply and storage in tilled and nontilled soils as influenced by cover crops and nitrogen fertilization. *Journal of Environmental Quality*, 35(4), 1507-1517. doi: 10.2134/jeq2005.0189

Salinas-Garcia, J., Hons, F., & Matocha, J. (1997). Long-term effects of tillage and fertilization on soil organic matter dynamics. *Soil Science Society of America Journal*, 61(1), 152-159.

Searchinger, T. D., Wirsenius, S., Beringer, T., & Dumas, P. (2018). Assessing the efficiency of changes in land use for mitigating climate change. *Nature*, 564(7735), 249.

Shi, J., Luo, Y. Q., Fan, Z., & Ping, H. (2010). The relationship between invasive alien species and main climatic zones. *Biodiversity & Conservation*, 19(9), 2485-2500.

Shipitalo, M. J., Edwards, W. M., Dick, W. A., & Owens, L. B. (1990). Initial storm effects on macropore transport of surface-applied chemicals in no-till soil. *Soil Science Society of America Journal*, 54(6), 1530-1536.

Six, J., Conant, R., Paul, E. A., & Paustian, K. (2002). Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. *Plant and Soil*, 241(2), 155-176.

Six, J., Elliott, E. T., & Paustian, K. (2000). Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology & Biochemistry*, 32(14), 2099-2103.

Six, J., Ogle, S. M., Conant, R. T., Mosier, A. R., & Paustian, K. (2004). The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. *Global Change Biology*, 10(2), 155-160.

Spokas, K., Koskinen, W., Baker, J., & Reicosky, D. (2009). Impacts of woodchip biochar additions on greenhouse gas production and sorption/degradation of two herbicides in a Minnesota soil. *Chemosphere*, 77(4), 574-581.

Spokas, K. A., & Reicosky, D. C. (2009). Impacts of sixteen different biochars on soil greenhouse gas production. *Annals of Environmental Science*, 3, 179-193.

Stronkhorst, L., & Venter, A. (2008). Investigating the soil organic carbon status in South African soils and the relationship between soil organic carbon and other soil chemical properties. ARC-ISCW Report No. Agricultural Research Council, Pretoria.

Swanepoel, C. M., Laan, M. V. D., Weepener, H. L., Preez, C. C. D., & Annandale, J. G. (2016). Review and meta-analysis of organic matter in cultivated soils in southern Africa. *Nutrient Cycling in Agroecosystems*, 104(2), 107-123.

Thomsen, I. K., & Christensen, B. T. (2004). Yields of wheat and soil carbon and nitrogen contents following long-term incorporation of barley straw and ryegrass catch crops. *Soil Use and Management*, 20(4), 432-438.

Tian, G., Kang, B. T., Kolawole, G. O., Idinoba, P., & Salako, F. K. (2005). Long-term effects of fallow systems and lengths on crop production and soil fertility maintenance in West Africa. *Nutrient Cycling in Agroecosystems*, 71(2), 139-150.

Tondoh, J. E., Ouédraogo, I., Bayala, J., Tamene, L., Sila, A., Vågen, T. G., & Kalinganiré, A. (2016). Soil organic carbon stocks in semi-arid West African drylands: implications for climate change adaptation and mitigation. *Soil,* Copernicus GmbH, 1-41 p.

UNEP. (1997). World Atlas of Desertification, 2nd edn (eds Middleton N, Thomas D), Edward Arnold, London.

Unger, P. W. (1997). Management-induced aggregation and organic carbon concentrations in the surface layer of a Torrertic Paleustoll. *Soil & Tillage Research*, 42(3), 185-208.

Van Bergen, P. F., Nott, C. J., Bull, I. D., Poulton, P. R., & Evershed, R. P. (1998). Organic geochemical studies of soils from the Rothamsted Classical Experiments—IV. Preliminary results from a study of the effect of soil pH on organic matter decay. *Organic Geochemistry*, 29(5–7), 1779-1795.

Van Eerd, L. L., Congreves, k. A., Hayes, A., Verhallen, A., & Hooker, D. C. (2014). Long-term tillage and crop rotation effects on soil quality, organic carbon, and total nitrogen. *Canadian Journal of Soil Science*, 94(3), 303-315.

Vieira, F. C. B., Bayer, C., Zanatta, J. A., Mielniczuk, J., & Six, J. (2009). Building up organic matter in a subtropical Paleudult under legume cover-crop-based rotations. *Soil Science Society of America Journal*, 73(5), 1699-1706. doi: 10.2136/sssaj2008.0241

Virto, I., Barré, P., Burlot, A., & Chenu, C. (2012). Carbon input differences as the main factor explaining the variability in soil organic C storage in no-tilled compared to inversion tilled agrosystems. *Biogeochemistry*, 108(1-3), 17-26.

Wang, J., Xiong, Z., & Kuzyakov, Y. (2016). Biochar stability in soil: meta-analysis of decomposition and priming effects. *Global Change Biology Bioenergy*, 8(3), 512-523.

Wang, X., Butterly, C. R., Baldock, J. A., & Tang, C. (2017). Long-term stabilization of crop residues and soil organic carbon affected by residue quality and initial soil pH. *Science of The Total Environment*, 587, 502-509.

Weng, Z., Van Zwieten, L., Singh, B. P., Tavakkoli, E., Joseph, S., Macdonald, L. M., . . . Cowie, A. (2017). Biochar built soil carbon over a decade by stabilizing rhizodeposits. *Nature Climate Change*, 7(5), 371-376. doi: 10.1038/nclimate3276

West, T. O., & Post, W. M. (2002). Soil organic carbon sequestration rates by tillage and crop rotation. *Soil Science Society of America Journal*, 66(6), 1930-1946.

Wiesmeier, M., Poeplau, C., Sierra, C. A., Maier, H., Frühauf, C., Hübner, R., . . . Hangen, E. (2016). Projected loss of soil organic carbon in temperate agricultural soils in the 21st century: effects of climate change and carbon input trends. *Scientific Reports*, 6, 32525.

Willett, V. B., Reynolds, B. A., Stevens, P. A., Ormerod, S. J., & Jones, D. L. (2004). Dissolved organic nitrogen regulation in freshwaters. *Journal of Environmental Quality*, 33(1), 201-209.

Wilson, H. M., & Alkaisi, M. M. (2008). Crop rotation and nitrogen fertilization effect on soil CO_2 emissions in central Iowa. *Applied Soil Ecology*, 39(3), 264-270.

Zhang, A., Bian, R., Pan, G., Cui, L., Hussain, Q., Li, L., . . . Han, X. (2012). Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: a field study of 2 consecutive rice growing cycles. *Field Crops Research*, 127, 153-160.

Zhao, X., Liu, S. L., Pu, C., Zhang, X. Q., Xue, J. F., Ren, Y. X., . . . Zhang, H. L. (2017). Crop yields under no-till farming in China: A meta-analysis. *European Journal of Agronomy*, 84, 67-75. doi: 10.1016/j.eja.2016.11.009

Table 1. Between-group variability (Q_M) of the variables controlling the effects of climate-smart agriculture management practices on soil organic carbon.

Variables	No-till		Reduced tillage		Cover crop		Biochar	
	df	Q_M	df	Q_M	df	Q_M	df	Q_M
Duration	2	12.14**	2	13.69**	2	26.19***	1	0.04
Aridity index	1	0.13	1	10.99***	1	0.04	1	5.73*
Mean annual air temperature	1	16.32***	1	0.47	1	55.99***	1	6.48*
Soil texture	5	20.98***	5	32.15***	4	3.58	5	9.65
Soil depth	3	210.69***	3	73.38***	2	17.38***	-	-
Soil pH	2	9.8**	2	3.52	2	9.05*	2	28.64***
Residue	1	6.56*	1	0.04	1	4.07*	-	-
Nitrogen fertilization	3	7.62	3	11.43*	2	0.89	2	7.22*
Irrigation	1	9.61**	1	0.92	1	0.16	1	1.7
Crop rotation	1	1.72	1	0.26	1	19.43***	1	4.53*

Statistical significance of Q_M: * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

Figure captions

Figure 1. Relationship between climate-smart management practices and soil processes. "+" means a positive feedback or promotion effect; "-" means a negative feedback or inhibition function; and "?" means the effect is unclear. Blue, black, and red show the effect of cover crops, conservation tillage, and biochar on the soil environment, processes, and pools, respectively. SOC: soil organic carbon.

Figure 2. Comparison of climate-smart management vs. their controls for the entire dataset. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 3. Comparison of climate-smart management vs. their controls for subcategories of climate zone (a: the climate zones were divided by aridity index; b: the climate zones were

divided by mean annual air temperature). The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 4. Comparison of climate-smart management vs. their controls for subcategories of soil textures. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 5. Comparison of climate-smart management vs. their controls for subcategories of soil depth. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage. The average depths of each categorical group were presented in supplementary files (Table S4-S7).

Figure 6. Comparison of climate-smart management vs. their controls for subcategories of soil pH. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 7. Comparison of climate-smart management vs. their controls for subcategories of experiment duration. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

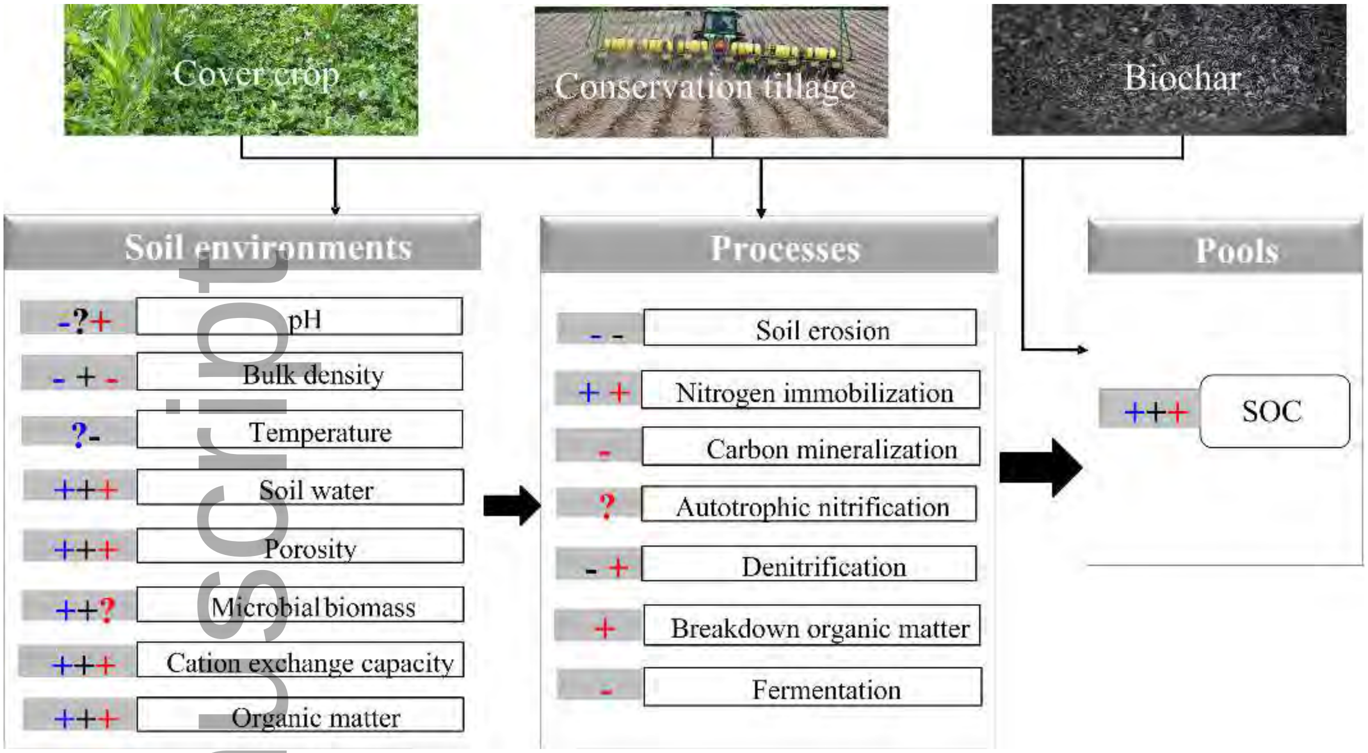
Figure 8. Comparison of climate-smart management vs. their controls for subcategories of crop residues. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

Figure 9. Comparison of climate-smart management vs. their controls for subcategories of nitrogen fertilizer use. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. Low, medium, and high levels of nitrogen fertilizer use represent 1-100, 101-200, and >200 kg N ha^{-1}, respectively. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

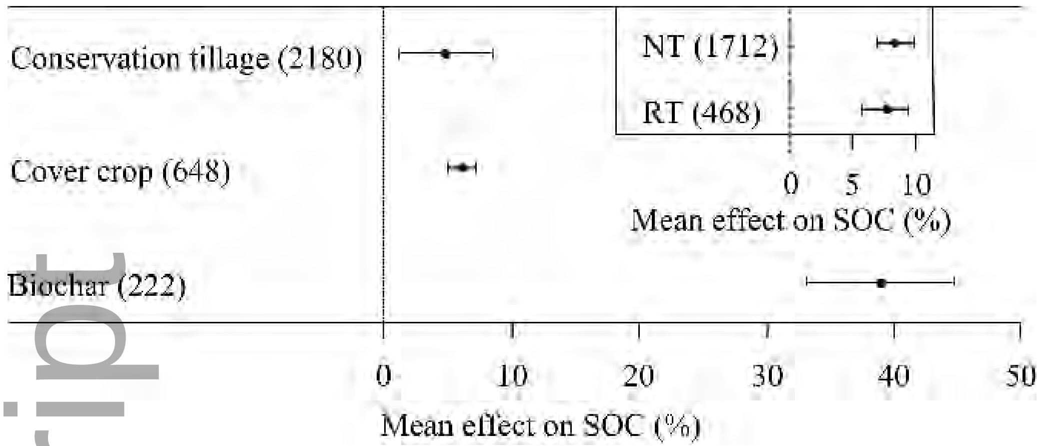
Figure 10. Comparison of climate-smart management vs. their controls for subcategories of water management (a) and cropping systems (b). The number in parentheses represents the

number of observations. Error bars represent 95% confidence intervals. SOC: soil organic carbon; NT: no-till; RT: reduced tillage.

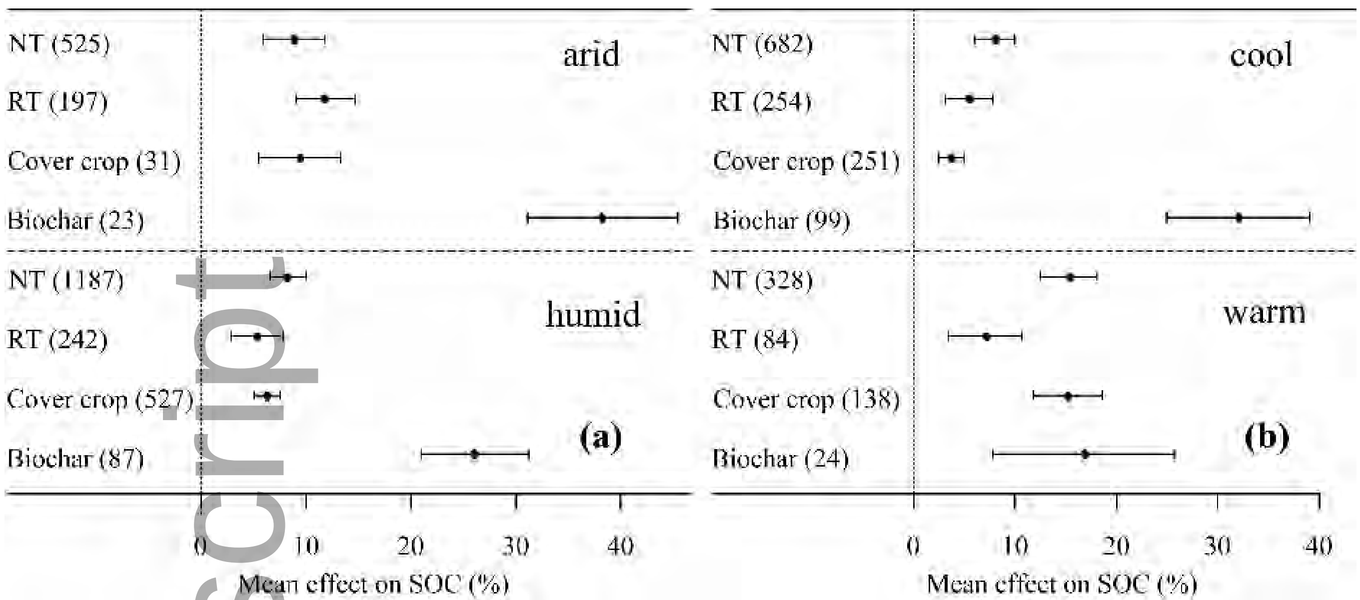
Figure 11. The effect size of combined conservation tillage and cover crops for different subcategories. The number in parentheses represents the number of observations. Error bars represent 95% confidence intervals. The vertical solid line represents 11%, which is the theoretical sum of the effect sizes of conservation tillage and cover crops. SOC: soil organic carbon.



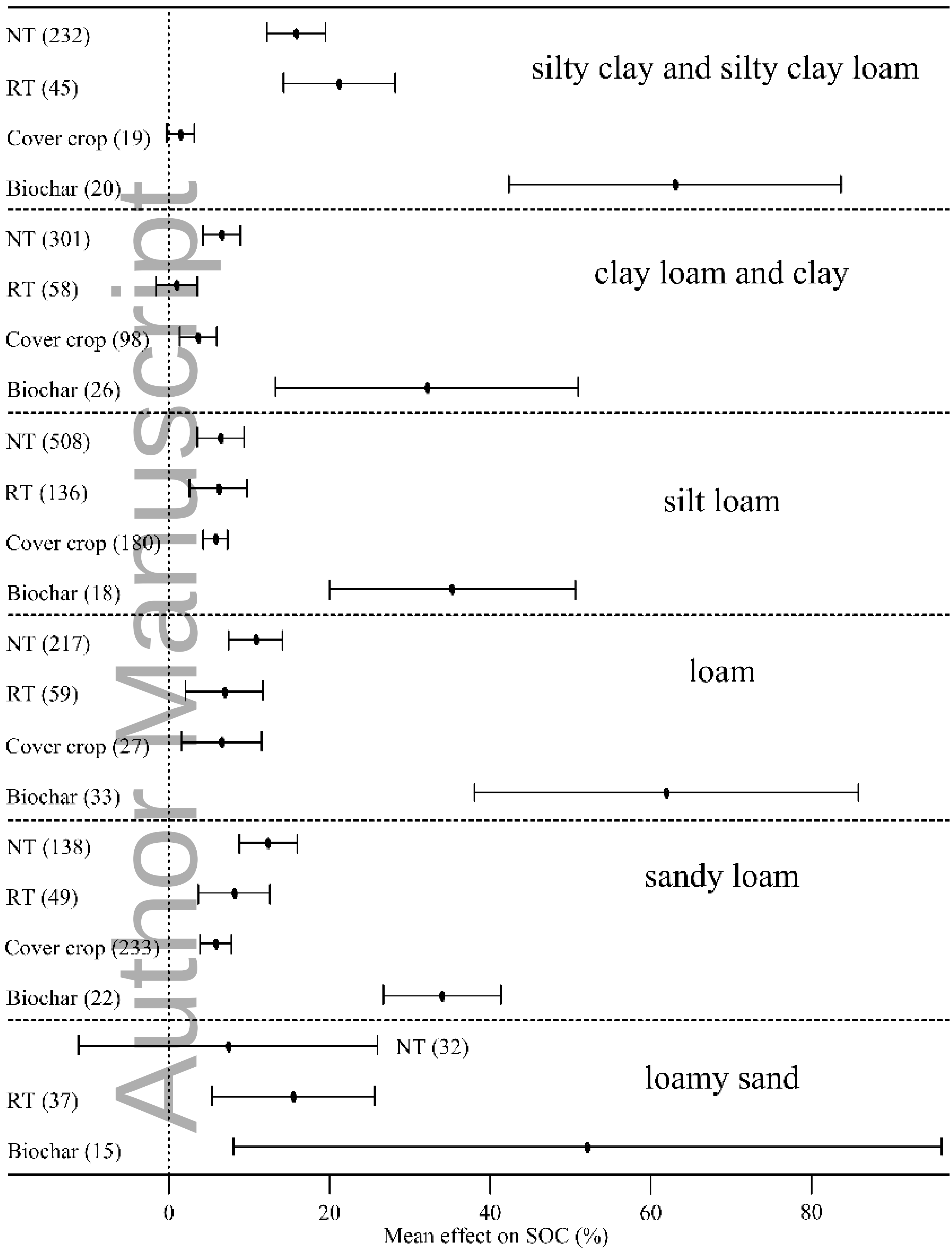
gcb_14658_f1.tif



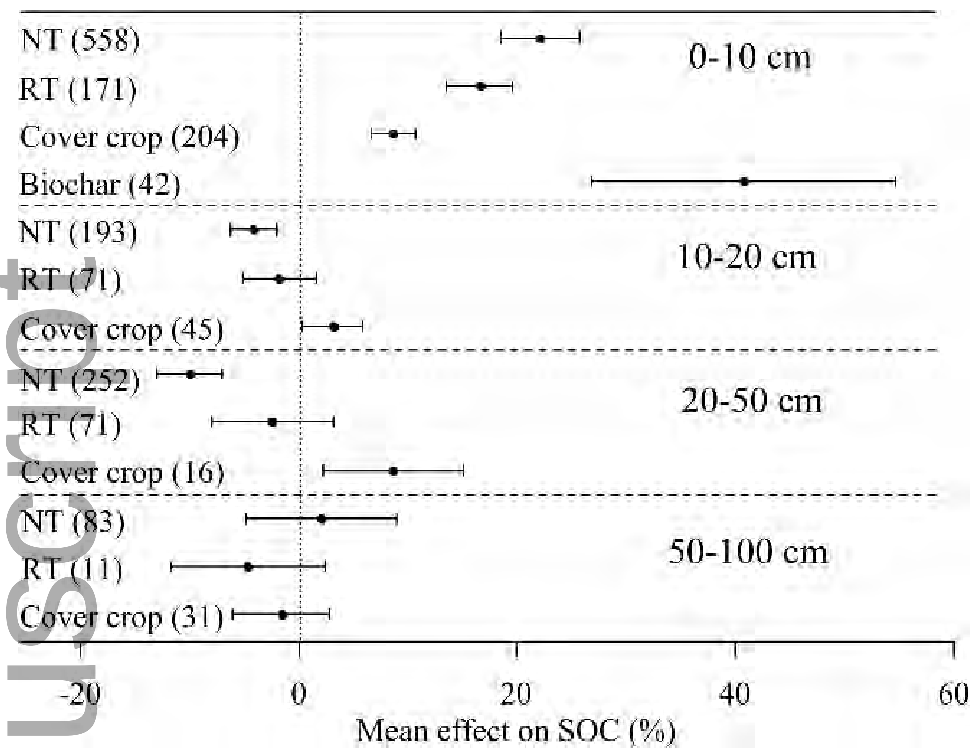
gcb_14658_f2.tif



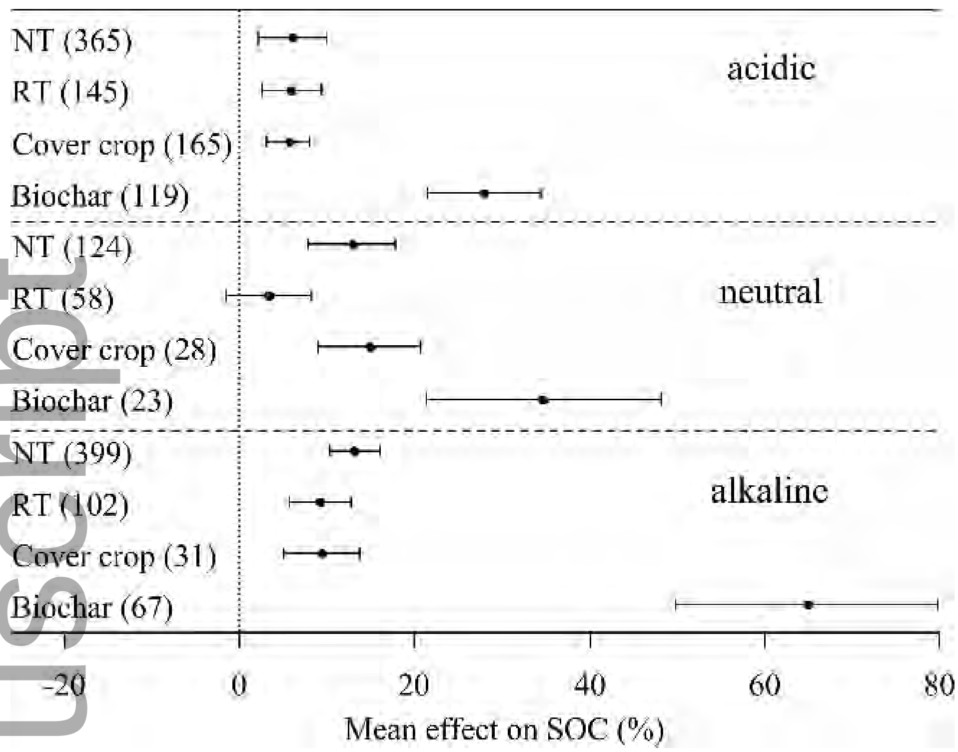
gcb_14658_f3.tif



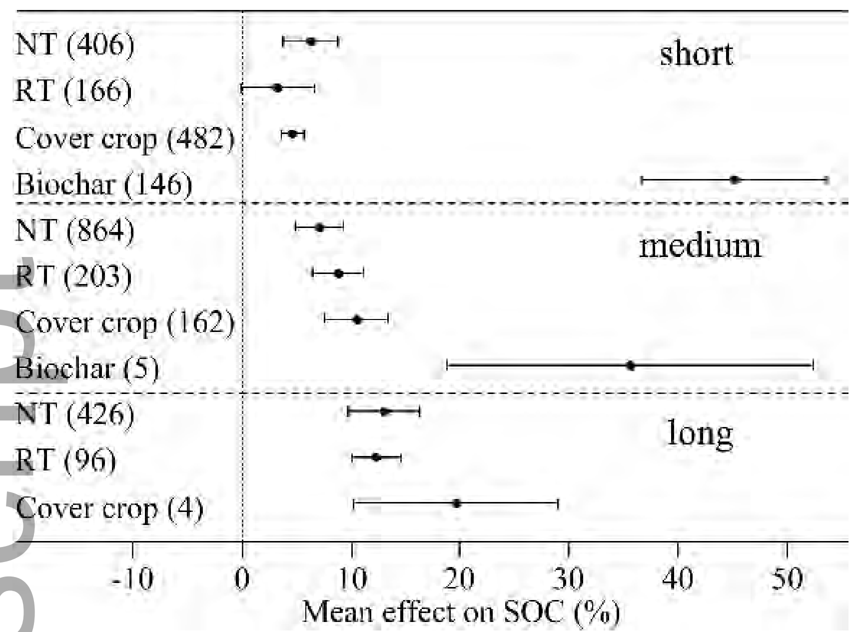
gcb_14658_f4.tif



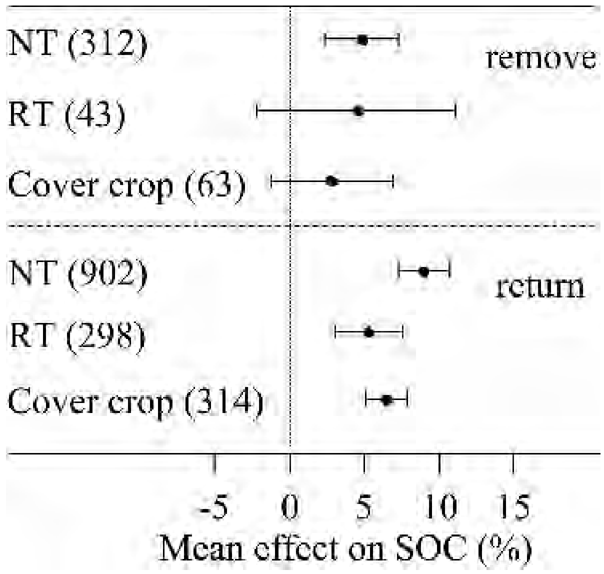
gcb_14658_f5.tif



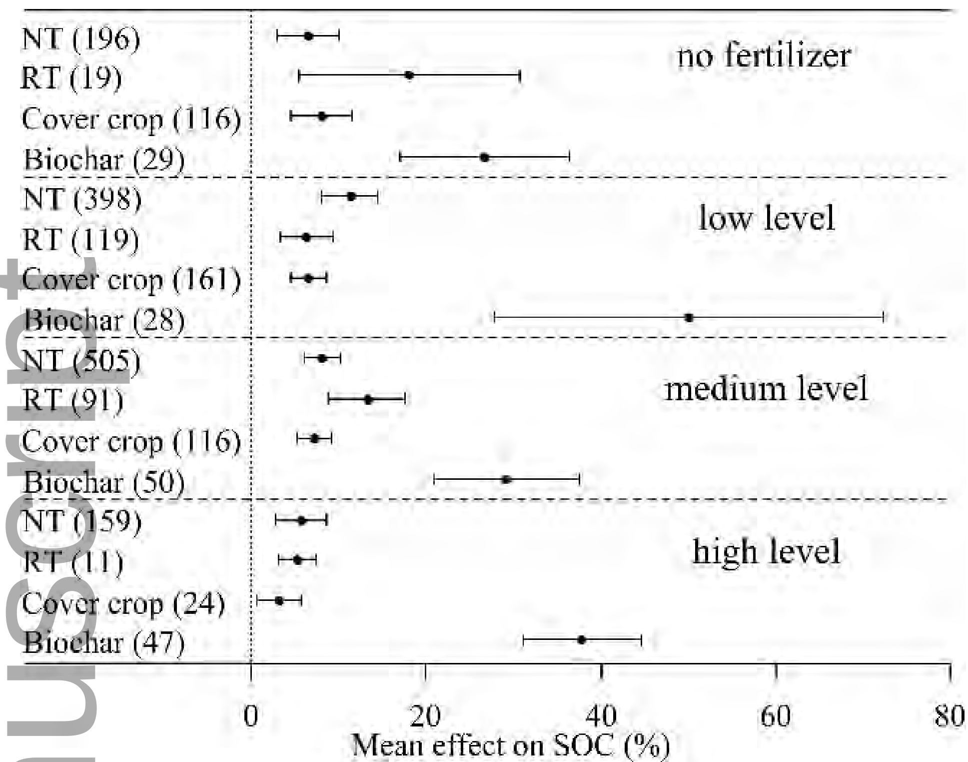
gcb_14658_f6.tif



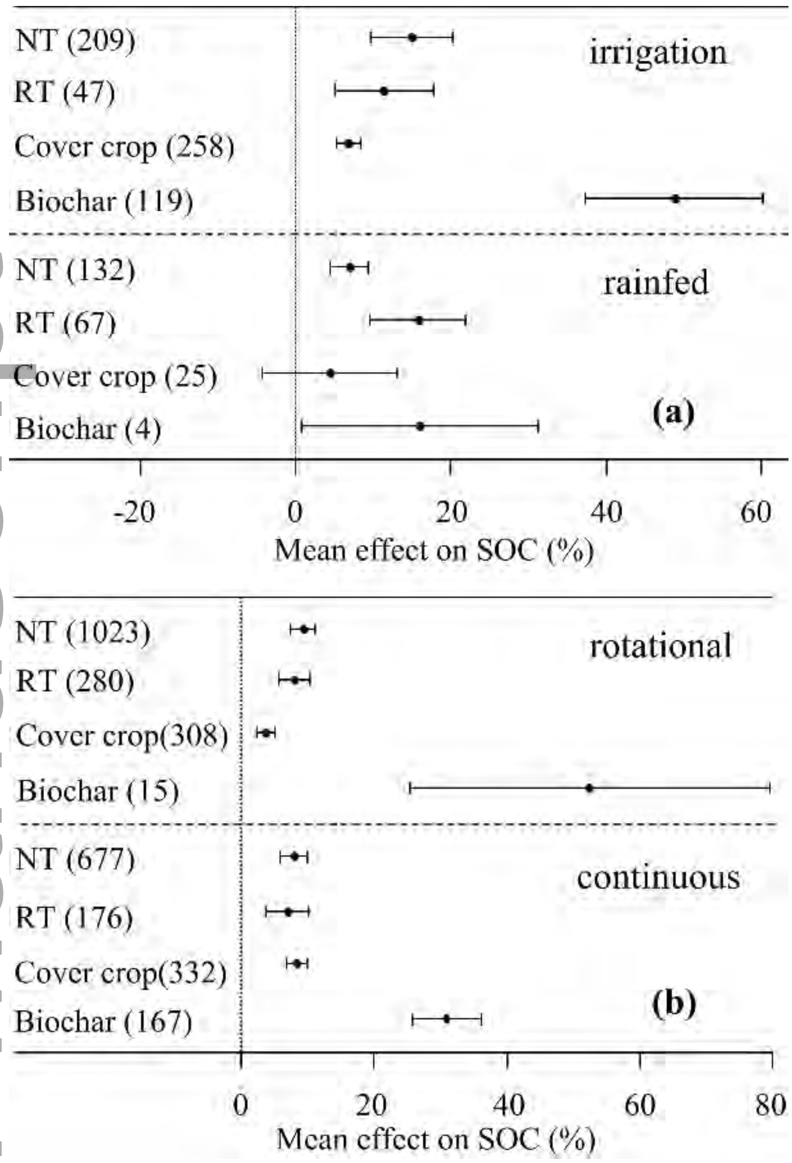
gcb_14658_f7.tif



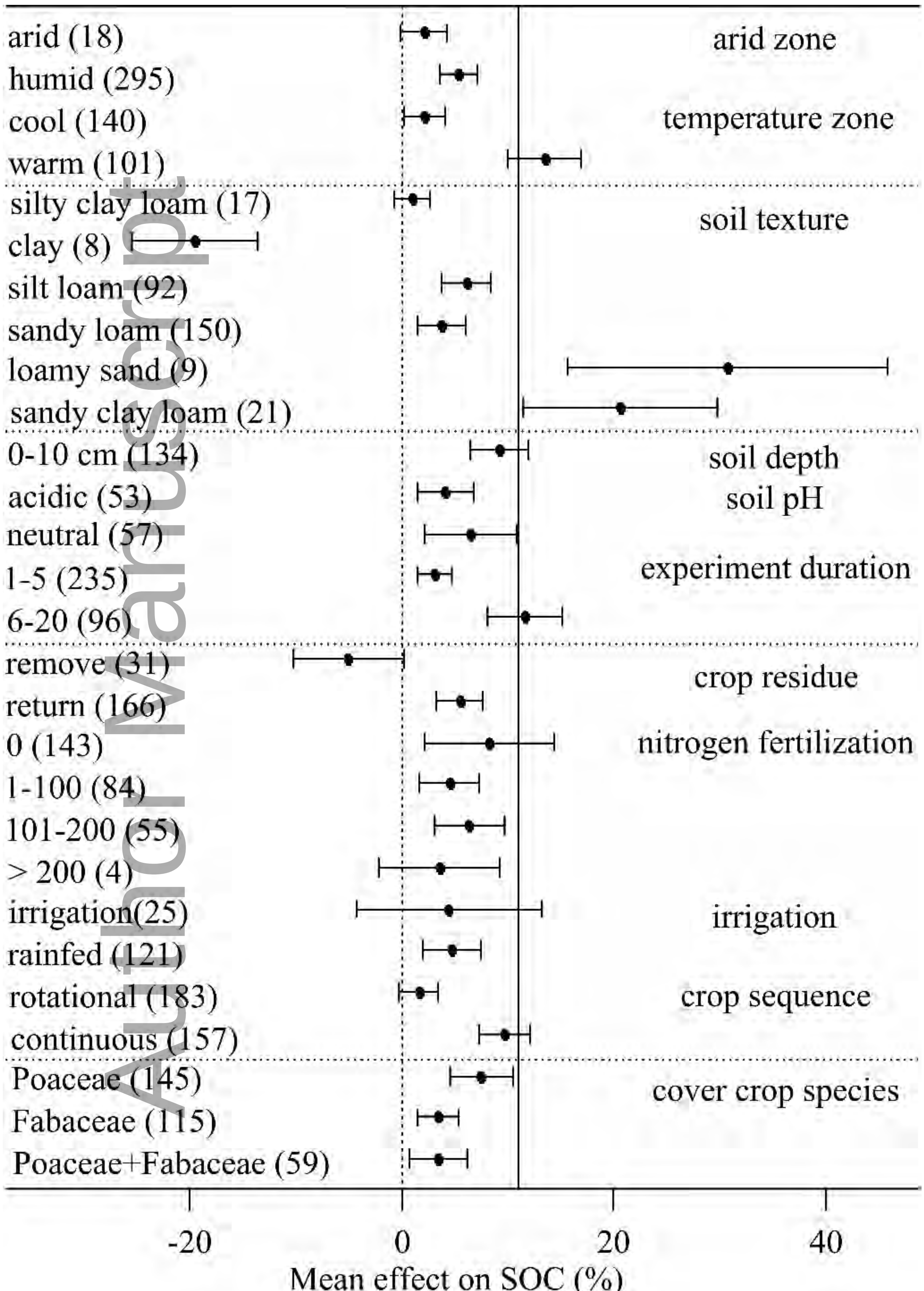
gcb_14658_f8.tif



gcb_14658_f9.tif



gcb_14658_f10.tif



gcb_14658_f11.tif

Exhibit I

Nutrient Cycling in Agroecosystems **60:** 1–8, 2001.

Nitrous oxide emissions from an irrigated soil as affected by fertilizer and straw management

X. Hao*, C. Chang, J.M. Carefoot, H.H. Janzen & B.H. Ellert
Lethbridge Research Centre, Agriculture and Agri-Food Canada, PO Box 3000, Lethbridge, AB Canada T1J 4B1
(**Corresponding author; e-mail: haoxy@em.agr.ca)*

Key words: greenhouse gas, N_2O flux, straw and fertilizer management, tillage

Abstract

Nitrous oxide (N_2O) emission from farmland is a concern for both environmental quality and agricultural productivity. Field experiments were conducted in 1996–1997 to assess soil N_2O emissions as affected by timing of N fertilizer application and straw/tillage practices for crop production under irrigation in southern Alberta. The crops were soft wheat (*Triticum aestivum* L.) in 1996 and canola (*Brassica napus* L.) in 1997. Nitrous oxide flux from soil was measured using a vented chamber technique and calculated from the increase in concentration with time. Nitrous oxide fluxes for all treatments varied greatly during the year, with the greatest fluxes occurring in association with freeze-thaw events during March and April. Emissions were greater when N fertilizer (100 kg N ha^{-1}) was applied in the fall compared to spring application. Straw removal at harvest in the fall increased N_2O emissions when N fertilizer was applied in the fall, but decreased emissions when no fertilizer was applied. Fall plowing also increased N_2O emissions compared to spring plowing or direct seeding. The study showed that N_2O emissions may be minimized by applying N fertilizer in spring, retaining straw, and incorporating it in spring. The estimates of regional N_2O emissions based on a fixed proportion of applied N may be tenuous since N_2O emission varied widely depending on straw and fertilizer management practices.

Introduction

Agricultural soils are recognized as an important source of atmospheric nitrous oxide (N_2O), a gas contributing to the 'enhanced greenhouse effect' (Mosier and Schimel, 1991). Nitrous oxide also participates in reactions which destroy stratospheric ozone, resulting in a higher UV-B intensity (Cicerone, 1987). In addition to these environmental concerns, N_2O emission also affects crop production as it represents a loss of plant-available N which reduces fertilizer efficiency (Eichner, 1990).

Nitrous oxide is produced from plant-available soil N during nitrification and denitrification (Sahrawat and Keeney, 1986). The emission of N_2O from soil follows an irregular pattern during the year, often depending on agricultural practices, soil properties, and climatic conditions (Jarvis et al., 1991; Ramos, 1996; Henault et al., 1998; Lemke et al., 1998a).

Application of fertilizer N could potentially increase N_2O emissions by supplying more plant-available N for N_2O production (Loro et al., 1997; Mulvaney et al., 1997). However, emission of N_2O from N-fertilized cropland varies considerably, ranging from 0.001% to 6.84% of applied N (Eichner, 1990). Past research has concentrated primarily on the effects of various chemical forms of N fertilizer on N_2O emissions (e.g., Clayton et al., 1997; Mulvaney et al., 1997). Other studies considered how fertilizer rate, placement and application method affected N_2O emissions (Henault et al., 1998; Skiba et al., 1993; Mulvaney et al., 1997).

This study considers N_2O emissions from irrigated soils in a semi-arid region. Under irrigation, crop yields must be higher than dry land farming to cover the added costs. These yields are achieved through a combination of higher moisture levels and increased fertilization, both of which could affect N_2O emis-

Table 1. Treatments used in 1996 – 1997 study

Treatment	N Fertilizer		
# Nrate&season_straw_tillage	Rate	Time	Straw-Tillage
	(kg N ha^{-1})		
1 Fert0_NS_Fall	0		No straw[b]-Fall plow[c]
2 Fert0_NS_DS	0		No straw- Direct seed
3 Fert0_S_Fall	0		Straw-Fall plow
4 Fert100f_NS_Fall	100	Fall[a]	No straw-Fall plow
5 Fert100f_NS_DS	100	Fall	No straw-Direct seed
6 Fert100f_NS_Spring	100	Fall	No straw-Spring plow[c]
7 Fert100f_S_Fall	100	Fall	Straw-Fall plow
8 Fert100f_S_Spring	100	Fall	Straw-Spring plow
9 Fert100s_NS_Fall	100	Spring[a]	No straw-Fall plow
10 Fert100s_S_Fall	100	Spring	Straw-Fall plow

[a] N fertilizer applied on 30 October 1996 and 5 May 1997, for fall and spring applications, respectively.
[b] Straw was baled and removed from the field on 15 October 1996.
[c] Field plowed on 31 October 1996 and 12 May 1997 for fall and spring plow, respectively.

sions. Higher yields under irrigation mean more crop residue is produced. Both the amount of residue and its management (fall or spring incorporation or direct seeding) could also affect N_2O emissions. Often management involves straw removal, and in some areas removal has increased to satisfy the demand for livestock bedding and raw material to produce fiber and energy. Although straw removal may adversely affect the quality and long-term sustainability of agricultural land (Dormaar and Carefoot, 1998), the effect of straw management on N_2O emissions from irrigated land has not been studied. Furthermore, recent trends toward increased use of conservation tillage may also have implications for N_2O emissions. For example, higher N_2O emissions have been reported with zero-till when compared to conventional tillage (Aulakh et al., 1984; MacKenzie et al., 1997; Palma et al., 1997).

The objectives of this study were to investigate the effects of the timing of N fertilizer application (either spring or fall), straw removal, and tillage practices (fall plow, spring plow or direct seed) on N_2O emission under irrigated conditions. Such information will be imperative to develop management practices that minimize N losses, including N_2O emissions, from agricultural land.

Materials and methods

Nitrous oxide emissions were measured in a long-term irrigated residue management experiment initiated in 1986 on a Dark Brown Chernozemic soil (Typic Haploboroll) at Lethbridge Alberta (49^{0}42' N, 112^{0}48' W) (Carefoot et al., 1994). The cropping sequence from 1986 to 1996 was wheat–wheat–oats, including soft white spring wheat (*Triticum aestivum* L.) and spring seeded oats (*Avena sativa* L.). To help manage an infestation of wild oats, canola (*Brassica napus* L.) was used instead of oats in 1997.

In the fall of 1986, a factorial experiment was set up with five straw-tillage treatments (straw-fall plow, straw-spring plow, no straw-fall plow, no straw-spring plow or no straw-direct seed), four N fertilizer application rates (0, 50, 100 or 200 kg N ha^{-1}) and two N fertilizer application times (fall or spring) using a randomized complete block design with four replications. ‘No straw’ indicates removal by baling of threshed straw but not the standing stubble (typically 15 to 17 cm tall) remaining after grain harvest. Nitrogen fertilizer in the form of NH_4NO_3 was broadcast onto the soil surface. Soil properties have been described in detail by Carefoot et al. (1994) and by Dormaar and Carefoot (1998).

The effects of fertilizer timing and tillage-straw management on N_2O emission were studied for selected treatments in 1996–1997 (Table 1). Soft wheat (cv. AC Reed) was harvested on 12 October 1996

Table 2. Average N_2O emission, water-filled porosity and temperature during 1996 – 1997

Treatment[a]	Daily flux[b]	Water-filled porosity[b]	Soil temperature[b]
	g N ha^{-1} d^{-1}	m^3 m^{-3}	°C
1 Fert0_NS_Fall	1.18b[c]	0.353d	5.73ab
2 Fert0_NS_DS	1.60b	0.494a	5.09ab
3 Fert0_S_Fall	5.23ab	0.348d	5.90a
4 Fert100f_NS-Fall	15.64a	0.337d	5.10ab
5 Fert100f_NS_DS	5.74ab	0.486ab	4.56ab
6 Fert100f_NS_Spring	4.60b	0.441bc	4.69ab
7 Fert100f_S_Fall	8.55ab	0.333d	4.97ab
8 Fert100f_S_Spring	2.50b	0.427c	5.57ab
9 Fert100s_NS_Fall	9.47ab	0.345d	5.55ab
10 Fert100s_S_Fall	4.34b	0.351d	4.28b

[a] Treatments are defined in Table 1.
[b] Average over the entire experimental period.
[c] Means followed by different letters indicate significant differences at the 0.05 probability level, according to the Tukey test (SAS Institute, 1990).

and straw was baled for the required treatments (treatments 1, 2, 4, 5, 6, and 9 in Table 1). Fertilizer rates of 0 and 100 kg N ha^{-1} were selected. The 100 kg N ha^{-1} represents the current recommended N fertilizer rate for irrigated fields in southern Alberta (McKenzie and Kryzanowski, 1993). For the fertilized treatments, NH_4NO_3 was broadcast on 30 October 1996 for fall-applied treatments or on 5 May 1997 for spring-applied treatments.

The direct-seeded treatments were not tilled. The other treatments were tilled with a moldboard plow (to 20 cm depth) followed by cultivation with a disc, either on 31 October 1996 for the fall tillage treatments or on 12 May 1997 for the spring tillage treatments. Canola (cv. Tobin) was seeded on 21 May, harvested on 7 August and the residue was baled on 12 August 1997. Nitrous oxide fluxes were measured from 6 November 1996 to 9 September 1997.

Nitrous oxide fluxes from soil were measured on four replicate plots for each treatment, using a vented chamber (Hutchinson and Mosier, 1981) modified to permit separation of the chamber cover from the base (Chang et al., 1998). One base collar was installed in each plot where it remained for the entire experimental period, except during tillage, seeding or harvesting operations. The volume of the chamber was 1604 cm^3 (9.2 cm high by 14.9 cm in diam.) and its cross-sectional area was 174 cm^2. Fluxes were measured at weekly intervals (between 0700 and 0830 h) by attaching the chamber covers to the base collars and collecting air samples from the chamber head space. For each chamber, 5 ml of headspace air was drawn through a septum into 10 ml polypropylene syringes at 0, 10, 20, 30, and 60 min after the soil was covered. Immediately after air sampling, the syringe needle was stuck into a rubber stopper to prevent gas exchange. On the same day, air samples were analyzed for N_2O using a gas chromatograph (Varian 3600, Varian Instruments, Walnut Creek, CA) equipped with an electron capture detector. During winter months, snow was brushed off the soil surface before the chamber cover was attached to the base.

For each chamber, the flux was calculated by fitting a second order polynomial equation (SAS Institute, 1990) to the five successive N_2O concentrations versus time. Fluxes estimated from single interval measurements may not be accurate (Anthony et al., 1995), because the pattern of N_2O accumulation during 60 min is usually curvilinear (Hutchinson and Mosier, 1981; Chang et al., 1998). This likely reflects the change in concentration gradient as N_2O accumulates in the chamber headspace. The flux of N_2O was calculated by taking derivatives of the second order polynomials and converting them into g ha^{-1} d^{-1}. For each plot, cumulative N_2O emission was calculated by summing the products of the measurement interval (i.e., days between measurements) and the mean flux for that interval (i.e., arithmetic mean of fluxes measured at the start and the end of the interval). Soil temperatures (2.5 cm depth), estimated from thermocouples (Digital Omega HH – 25C, Omega Technology, Stamford, CT), were recorded when the air samples were collected from the chambers. When the soil was not frozen, soil moisture content in the 0-15 cm layer was determined by time domain reflectometry using three-wire probes similar to the design of Zegelin et al. (1989) and a cable tester (1502C, Tektronix, Beaverton OR). The ratio of water- to air-filled porosity was calculated based on the volumetric moisture content and bulk density (Table 2). Since bulk density (BD) data were not collected during 1996-1997 cropping season, BD data collected in spring 1995 (Domaar and Carefoot, 1998) were used in the calculation. Climatic data were obtained from the Lethbridge Research Centre meteorological station located less than 300 m away from the experimental plots.

ANOVA was conducted on the arithmetic mean daily N_2O fluxes for the entire 1996–1997 experimental period (SAS Institute, 1990). Since the ANOVA was significant at a probability level of 0.05, the multi-range Tukey test was performed to assess treatment differences. The contrast method (via. the GLM procedure, SAS Institute, 1990) was also used to evaluate the influence of key management variables.

Results and discussion

Environmental conditions

During the study period (November 1996 through September 1997), the total precipitation (393 mm) was slightly above the long-term average (379 mm) for the 11-month period. Most precipitation occurred in May (96 mm) and June (101 mm) 1997 (Figure 1a). Air temperatures were highly variable during winter and spring when warm 'Chinook' winds blow in southern Alberta (Grace and Hobbs, 1986). For example, between 19 and 22 December 1996, mean daily air temperature changed from 1.2 to -27.2 ^{0}C (Figure 1a).

The average soil temperature (2.5 cm depth) for all treatments followed similar patterns to air temperature, but the variations were much smaller (Figure 1b). Among the different treatments, water-filled porosity was affected only by the tillage operations (Figure 1c and Tables 2 and 3). Fall plow had the lowest while direct seeding had the highest water-filled porosity throughout the experimental period.

Temporal trends

The average N_2O daily flux for unfertilized and 100 kg ha^{-1} N fertilized treatments fluctuated greatly, ranging from as low as -5 (uptake by soil) to 18 for the unfertilized and 63 g N ha^{-1} d^{-1} for the N fertilized treatment during the year (Figure 1d). Maximum daily fluxes occurred in early spring (February to early April) as observed by other researchers in western Canada (Nyborg et al., 1997; Chang et al., 1998; Lemke et al., 1998b). The N_2O fluxes increased whenever the soil temperature increased (Figure 1b and 1d), perhaps associated with freeze-thaw events

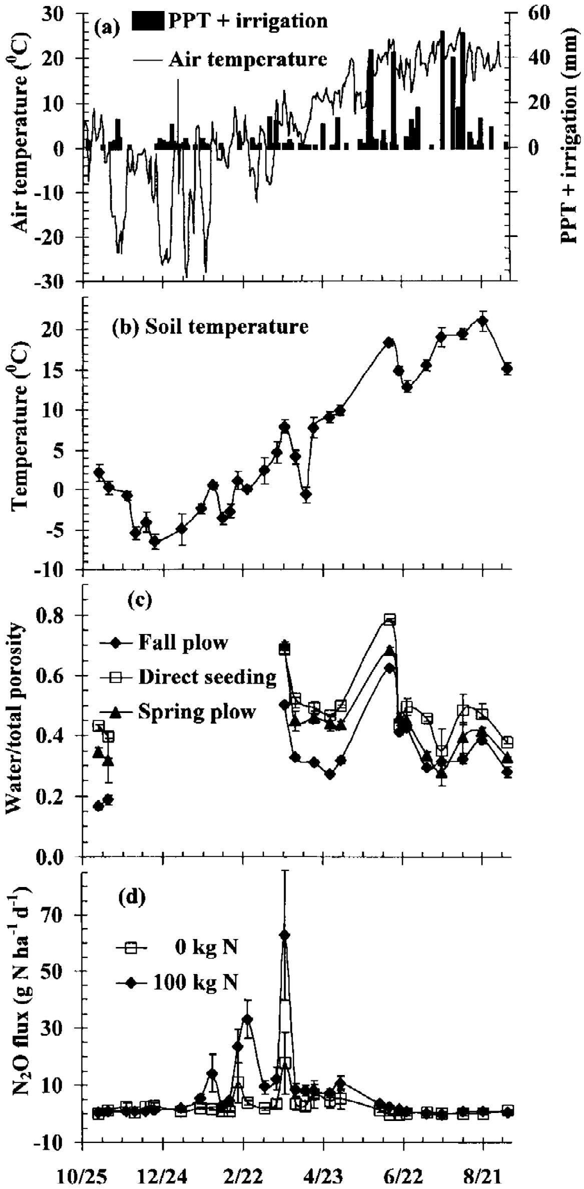


Figure 1. Mean N_2O fluxes in relation to environmental conditions during 1996 – 1997. (a) Daily air temperature and precipitation + irrigation, (b) Soil temperature at 2.5 cm, (c) water-filled porosity in the 0–15 cm soil layer and (d) mean daily N_2O flux patterns for unfertilized and fertilized treatments.

Table 3. Effect of management practices on N_2O flux, soil moisture and surface temperature

Effect	Contrast	Mean difference[b]		
		N_2O flux	Water-filled porosity	Soil temperature
		(g $ha^{-1}d^{-1}$)	m^3 m^{-3}	°C
N fertilizer				
Zero vs. 100 kg in fall	1+2+3 vs. 4+5+6+7+8+9+10[a]	−4.47*	−0.0136	0.630**
100 kg in fall vs. spring	4+7 vs. 9+10	4.77*	0.0098	0.116
Straw vs. no straw				
Zero N fertilizer	3 vs. 1	4.14	0.0051	0.172
100 kg N fertilizer	7+8+10 vs. 4+5+6+9	−3.56§	−0.0312**	−0.003
Plowing time				
Fall vs. spring/DS	4+7 vs. 5+6+8	8.66**	−0.1135**	0.279
DS vs. spring	5 vs. 6	2.24	0.0452**	−0.132

[a] Treatment numbers are defined in Table 1.
[b] Average over the entire experimental period.
§, *, ** Significant at 0.1, 0.05 and 0.01 probability levels, respectively.

during this time of year (Chen et al., 1995). The daily N_2O flux was very low from late July to the end of September. This flux decline was probably related to the decrease in available N content caused by crop uptake and leaching over the growing season (Xu et al., 1998). Relatively high water-filled porosity in late June (due to above normal precipitation) and in late July and early August (due to irrigation on 17 July, 25 July, and 1 August 1997) failed to produce appreciable N_2O fluxes.

Influence of fertilizer timing

Nitrous oxide emission was affected by the timing of N fertilizer application (Figure 2). Using contrast comparison, fall fertilizer application produced significantly greater N_2O emissions than spring fertilizer application (Table 3). Fall application provided a longer period with available N and moisture conditions favorable for denitrification and N_2O production during the freeze-thaw cycles in the early spring. Carefoot et al. (1994) and Nyborg et al. (1990) attributed the low recovery of fall-applied N fertilizer to denitrification during winter and spring. The N_2O fluxes from spring fertilized plots were lower than fall fertilized plots because the fertilizer was applied in early May after the maximum N_2O flux had occurred. Previous research seldom considered fertilizer timing (fall vs. spring application), but this study clearly demonstrates a significant influence on N_2O emissions. Despite the reduced frequency of N_2O flux measurements during late May to early June, previously published work suggests that large fluxes do not typically occur during this time of year in this region (Chang et al., 1998; Lemke et al., 1998a). While N_2O emissions from spring fertilized plots were greater than emissions from unfertilized plots (Figure 2a and 2c), the differences were not statistically significant.

Effect of straw removal

The effect of straw removal on N_2O emission depended on fertilizer treatment (Figures 2 and 3). When no fertilizer was applied and plots were plowed in the fall, N_2O emission from the ‘straw-removed’ treatment was about 25% that of the treatment where straw was incorporated into the soil (Figure 3a and 3c). The lower N_2O emission associated with straw removal, also observed by Zhengping et al. (1991), could be linked to several factors. Annual straw removal eventually depletes soil organic C and N (Dormaar and Carefoot, 1998), and decreases the amount of N potentially available for nitrification and denitrification. The availability of organic C as an electron donor for denitrification also influences N_2O production in soil (Sahrawat and Keeney, 1986). On one hand, the annual addition of fresh crop residue to soil stimulates denitrification (Burford and Bremner, 1975; Groffman, 1985) by providing readily available C for denitrifying bacteria (with adequate NO_3^- and moisture). Decomposition of crop residue consumes oxygen, which also stimulates denitrification and N_2O emission. On the other hand, addition of fresh wheat residue with a high C:N ratio could cause N immobilization and reduce

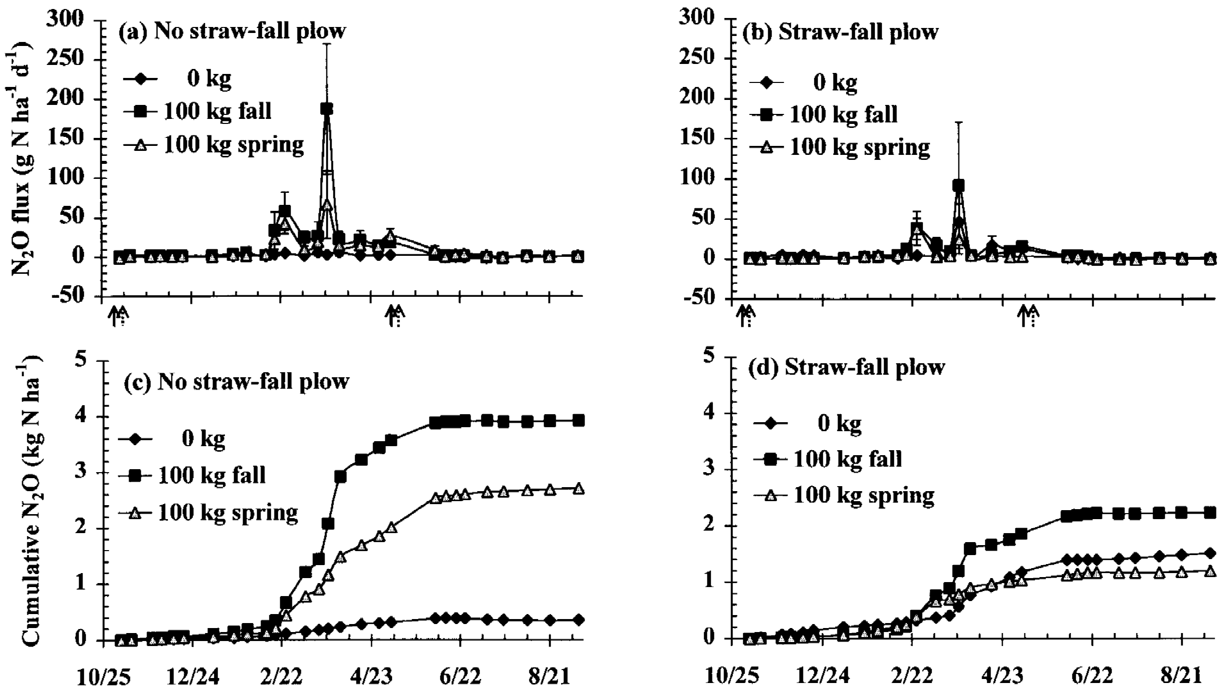


Figure 2. Effect of N fertilizer application timing during 1996–1997 on mean daily N_2O flux (a and b) and on cumulative N_2O emissions (c and d) measured on the fall plow treatments where straw was removed (a and c) and not removed (b and d). (Solid arrows point to the dates of fertilizer application and dotted arrows point to the dates of tillage operation.)

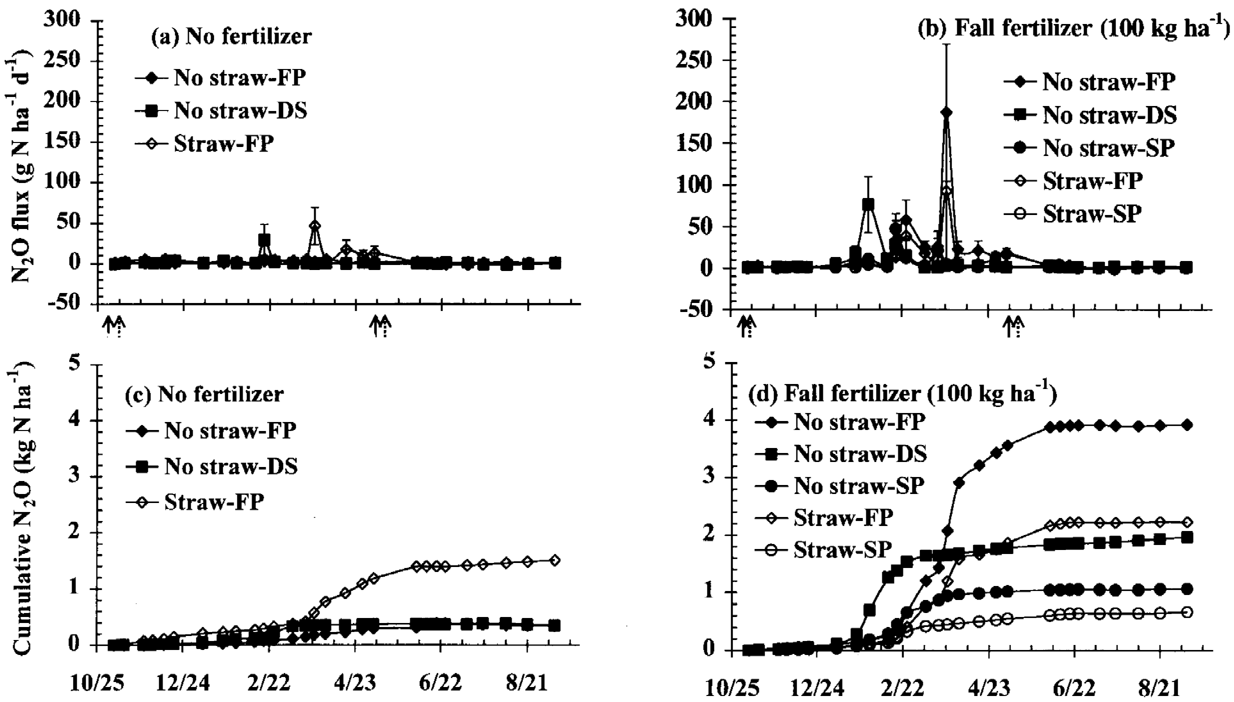


Figure 3. Effect of straw-tillage practices during 1996–1997 on mean daily N_2O flux (a and b) and cumulative N_2O emissions (c and d) measured on treatments with zero (a and c) and 100 kg N fertilizer applied in the fall (b and d). (Solid arrows point to the dates of fertilizer application and dotted arrows point to the dates of tillage operation.)

the N available for denitrification and nitrification, thus reducing N_2O production in the short term. The amount of N_2O produced reflects the combined effect of all these processes.

When 100 kg N ha^{-1} fertilizer was applied, however, the treatments with straw incorporated into the soil for both fall and spring tillage had about half the N_2O emission of those plots where straw was removed (Figures 2 and 3 and Table 3). In contrast to unfertilized treatments, straw incorporation caused much lower N_2O emissions compared to straw removed treatments. If a fixed proportion of straw N is converted to N_2O after incorporation into the soil (IPCC: Intergovernmental Panel on Climate Change, 1997), then more N_2O should be emitted from treatments with straw retained (especially at higher rates of N fertilization where straw N inputs are more appreciable). Since this was not observed, even in an experiment which had been in place for over 10 years, it appears that the proportion of straw N converted to N_2O is not the same as, and may be considerably less than, the proportion of fertilizer N converted to N_2O.

Effect of tillage

The effects of tillage were only studied for plots where fertilizer was applied in the fall. Spring plow produced the lowest average daily N_2O flux followed by direct seeding (but the difference was not significant). However, fall plow did produce significantly higher N_2O emissions (Figure 3 and Table 3) than the other two tillage treatments.

Several factors contribute to the higher N_2O emission from fall plow. First, tilling the soil in the fall incorporates the applied N fertilizer with the topsoil and thereby increases the amount of N available in soil for nitrification and denitrification in the winter and early spring. Fall plow also stimulates N mineralization, which further increases the amount of N available in soil as reported earlier for this field by Carefoot and Janzen (1997). Second, tilling the soil in the fall incorporates straw residue into soil and its decomposition consumes oxygen and makes the soil more anaerobic. Although the water-filled porosity was lower for fall plow, localized anaerobic sites could occur due to the decomposition of fresh straw residue. This is possible even for plots with straw removed, since baling does not remove all residue (especially standing stubble, leaves and chaff). Third, soil temperature was also higher for fall plow than spring plow or direct seeding treatments over the winter and spring months. Higher temperature increases denitrification and nitrification because biological activity is temperature-dependent.

The slightly higher N_2O emission from direct seeding over spring plowing probably reflects the greater water-filled porosity in these soils during spring. Although some researchers found that direct seeding produced higher N_2O emissions than conventional tillage (Aulakh et al., 1984; Hilton et al., 1994; MacKenzie et al., 1997; Palma et al., 1997), their N_2O emission data were collected during the growing season. The timing of conventional tillage operations in these experiments was not studied. Our study suggests that the timing of tillage operations (fall or spring) may have a greater influence on N_2O emissions than the tillage method (e.g., spring soil disturbance by direct seeding or pre-seeding tillage).

Implications

Nitrogen fertilizer is essential to grain production throughout the Canadian Prairies, especially on irrigated cropland. Many farmers in this area apply N fertilizer in the fall after harvest, when there is less demand on labor and machinery and better fertilizer prices. But this study shows that fall N fertilization can lead to higher N_2O emissions compared to those from spring fertilization or from unfertilized soil. Thus, N fertilizer should be applied in the spring.

The removal of straw as a raw material for production of fiber and energy reduces the amount of organic matter and nutrients returned back to soil. The results of this study provide an additional argument for limiting this practice. Removal of straw from fertilized soil increased N_2O emissions.

Tillage also had an impact on N_2O emission. Tillage effects are not only related to the method of tillage, but also when the tillage operation was conducted. Spring plow resulted in the lowest N_2O emission among the three tillage practices studied.

The N_2O emission from agricultural land has been estimated based upon the amount of N added to the soil without giving consideration to management practices (IPCC: Intergovernmental Panel on Climate Change, 1997). The IPCC assumes 1.25% of applied N (from fertilizer application or straw N addition) is lost as N_2O, implying a linear relationship between N_2O loss and the amount of N returned to the soil. As demonstrated in this study, N_2O emissions from agricultural land also depend on the timing of fertilizer application (fall vs. spring) and the tillage/straw management practices. Thus estimates of N_2O emissions based on the ‘fraction of applied N’ approach may be tenuous.

Acknowledgments

The authors gratefully acknowledge the dedicated technical assistance of Greg Travis, Elaine Nakonechny, Tony Curtis and Brett Hill in carrying out the field and laboratory work for this study. We also thank the AARI for their financial support.

References

Anthony WH, Hutchinson GL & Livingston GP (1995) Chamber measurement of soil-plant-atmosphere gas exchange: linear vs diffusion-based flux models. Soil Sci Soc Am J 59: 1308–1310

Aulakh MS, Rennie DA & Paul EA (1984) Gaseous nitrogen losses from soils under zero-till as compared with conventional-till management systems. J Environ Qual 13: 130–136

Burford JR & Bremner JM (1975) Relationship between the denitrification capacities of soils and total water soluble and readily decomposable soil organic matter. Soil Biol Biochem 7: 389–394

Carefoot JM, Janzen HH & Lindwall CW (1994) Crop residue management for irrigated cereals on the semi-arid Canadian prairies. Soil Tillage Res 32: 1–20

Carefoot JM & Janzen HH (1997) Effect of straw management, tillage timing and timing of fertilizer nitrogen application on the crop utilization of fertilizer and soil nitrogen in an irrigated cereal rotation. Soil Tillage Res 44: 195–210

Chang C, Cho cm & Janzen HH (1998) Nitrous oxide emission from long-term manured soils. Soil Sci Soc Am J 62: 677–682

Chen Y, Tessier S, MacKenzie AF & LaverdiPre MR (1995) Nitrous oxide emission from an agricultural soil subject to different freeze-thaw cycles. Agric Ecosys Environ 55: 123–128

Cicerone RJ (1987) Changes in stratospheric ozone. Science 237: 35–42

Clayton H, McTaggart IP, Parker J & Swan L (1997) Nitrous oxide emissions from fertilised grassland: A 2-year study of the effects of N fertilizer form and environmental conditions. Biol Fertil Soils 25: 252–260

Dormaar JF & Carefoot JM (1998) Effect of straw management and nitrogen fertilizer on selected soil properties as potential soil quality indicators of an irrigated Dark Brown Chernozemic soil. Can J Soil Sci 78: 511–517

Eichner MJ (1990) Nitrous oxide emissions from fertilized soils: Summary of available data. J Environ Qual 19: 272–280

Grace B & Hobbs EH (1986) The climate of the Lethbridge agricultural area: 1902-1985. LRS Mimeo Rep 3, Agric Can Res Stn, Lethbridge, AB Canada

Groffman PM (1985) Nitrification and denitrification in conventional and no-tillage soils. Soil Sci Soc Am J 49: 329–334

Heaney DJ, Nyborg m, Solberg ED, Malhi SS & Ashworth J (1992) Over winter nitrate loss and denitrification potential of cultivated soils in Alberta. Soil Biol Biochem 24: 877–884

Henault C, Devis X, Page S, Justes E, Reau R & Germon JC (1998) Nitrous oxide emission under different soil and land management conditions. Biol Fertil Soils 26: 199–207

Hilton BR, Fixen PE & Woodard HJ (1994) Effects of tillage, nitrogen placement, and wheel compaction on denitrification rates in the corn cycle of a corn-oats rotation. J Plant Nutr 17: 1341–1357

Hutchinson GL & Mosier AR (1981) Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Sci Soc Am J 45: 311–316

Intergovernmental Panel on Climate Change (1997) Revised 1996 IPCC guidelines for national greenhouse gas inventories: Reference manual. Chapter 4 Agriculture. Organization for Economic Corporation and Development (OECD) and the International Energy Agency (IEA) Paris

Jarvis SC, Barraclough D, Williams J & Rook AJ (1991) Patterns of denitrification loss from grazed grassland: Effect of N fertilizer inputs at different sites. Plant Sci 131: 77–88

Lemke RL, Izaurralde RC, Malhi SS, Arshad MA & Nyborg m (1998a) Nitrous oxide emissions from agricultural soils of the Boreal and Parkland regions of Alberta. Soil Sci Soc Am J 62: 1096–1102

Lemke RL, Izaurralde RC & Nyborg m (1998b) Seasonal distribution of nitrous oxide emission from soils in the Parkland region. Soil Sci Soc Am J 62: 1320–1326

Loro PJ, Bergstrom DW & Beauchamp EG (1997) Intensity and duration of denitrification following application of manure and fertilizer to soil. J Environ Qual 26: 706–713

McKenzie RH, & Kryzanowski L K (1993) Fertilizing irrigated grain and oilseed crops. Agdex 100/541-1 Alberta Agriculture Edmonton, Alberta, Canada

MacKenzie AF, Fan MX & Cadrin F (1997) Nitrous oxide emission as affected by tillage, corn-soybean-alfalfa rotations and nitrogen fertilization. Can J Soil Sci 77: 145–152

Mosier AR & Schimel DS (1991) Influence of agricultural nitrogen on atmospheric methane and nitrous oxide. Chem Ind 23: 874–877

Mulvaney RL Khan SA & Mulvaney CS (1997) Nitrogen fertilizers promote denitrification. Biol Fertil Soils 24: 211–220

Nyborg M, Malhi SS & Solberg ED (1990) Effect of date of application on the fate of ^{15}N-labelled urea and potassium nitrate. Can J Soil Sci 70: 21–31

Nyborg m, Ladlaw JW, Solberg ED & Malhi SS (1997) Denitrification and nitrous oxide emissions from a Black Chernozemic soil during spring thaw in Alberta. Can J Soil Sci 77: 153–160

Palma RM, Rimolo m, Saubidet MI & Conti ME (1997) Influence of tillage systems on denitrification in maize-cropped soils. Biol Fertil Soils 25: 142–146

Ramos C (1996) Effect of agricultural practices on the nitrogen losses to the environment. Fertil Res 43: 183–189

Sahrawat KL & Keeney DR (1986) Nitrous oxide emission from soils. Adv Soil Sci 4: 103–148

SAS Institute (1990) SAS/STAT User’s guide. 4th edn SAS Institute Inc, Cary, NC

Skiba U, Smith KA & Fowler D (1993) Nitrification and denitrification as sources of nitric oxide and nitrous oxide in a sandy soil. Soil Biol Biochem 25: 1527–1536

Thornton FC & Valente RJ (1996) Soil emissions of nitric oxide and nitrous oxide from no-till corn. Soil Sci Soc Am J 60: 1127–1133

Xu C, Shaffer MJ & Al-Kaisi m (1998) Simulating the impact of management practices on nitrous oxide emissions. Soil Sci Soc Am J 62: 736–742

Zegelin SJ, White I & Jenkins DR (1989) Improved field probes for soil water content and electrical conductivity measurement using time domain reflectometry. Water Resour Res 25: 2367–2374

Zhengping W, Liantie L, van Cleemput O & Baert L (1991) Effect of urease inhibitors on denitrification in soil. Soil Use Man 7: 230–232

Exhibit J

Nutrient Cycling in Agroecosystems **52:** 151–163, 1998.

Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices

C. Wagner-Riddle & G.W. Thurtell
Department of Land Resource Science, University of Guelph, Guelph, Ontario, Canada, N1G 2W1

Key words: flux-gradient method, gaseous losses from manure applications, spring thaw N_2O emissions, nitrogen cycle

Abstract

Highest rates of N_2O emissions from fertilized as well as natural ecosystems have often been measured at spring thaw. But, it is not clear if management practices have an effect on winter and spring thaw emissions, or if measurements conducted over several years would reveal different emission patterns depending on winter conditions. In this study, we present N_2O fluxes obtained using the flux-gradient approach over four winter and spring thaw periods, spanning from 1993 to 1996, at two locations in Ontario, Canada. Several agricultural fields (bare soil, barley, soybean, canola, grass, corn) subjected to various management practices (manure and nitrogen fertilizer addition, alfalfa ploughing, fallowing) were monitored. Nitrous oxide emissions from these fields from January to April over four years ranged between 0 and 4.8 kg N ha^{-1}. These thaw emissions are substantial and should be considered in the nitrous oxide budgets in regions where thaw periods occur. Our study indicates that agricultural management can play a role in mitigating these emissions. Our data show that fallowing, manure application and alfalfa incorporation in the fall lead to high spring emissions, while the presence of plants (as in the case of alfalfa or grass) can result in negligible emissions during thaw. This presents an opportunity for mitigation of N_2O emissions through the use of over-wintering cover crops.

Introduction

Uncertainties in the global nitrous oxide budget (Watson et al., 1990) have not only been related to the lack of N_2O flux measurement programmes, but also to the high spatial and temporal variability of fluxes. Because agriculture is estimated to contribute 80% of the anthropogenic N_2O emissions to the atmosphere (Isermann, 1994), a need for continuous N_2O flux measurements under various agricultural management practices has been identified (Mosier et al., 1996).

Highest rates of N_2O emissions from agricultural and natural ecosystems have often been measured at spring thaw (Goodroad and Keeney, 1984a; Goodroad et al., 1984; Cates and Keeney, 1987; Flessa and Dörsch, 1995; Wagner-Riddle et al., 1997). Flessa and Dörsch (1995) observed frost-induced releases at four agricultural sites, and suggested that this could be a general phenomenon for soils in temperate and boreal climates. No significant difference in the thaw-induced emission from extensively and intensively managed plots was observed during that study. In a study of two maize fields managed at two nitrogen levels, Goodroad et al. (1984) also did not observe a significant effect of management on spring thaw N_2O emission.

Field and laboratory studies have demonstrated that freeze/thaw cycles induce high N_2O production in soils (Goodroad and Keeney, 1984b; Christensen and Tiedje, 1990). With the intention of simulating typical emissions from freeze/thaw cycles for regions of continental climate, Chen et al. (1995) observed increased emissions with increased number of freeze/thaw cycles applied to soil cores. Because field N_2O fluxes have not been monitored over several winter and spring thaws, it is unclear if field conditions would always be conducive to increased emissions with increased number of freeze/thaw cycles.

Among the methods available for N_2O flux monitoring, micrometeorological methods are ideally suited for continuous monitoring, providing spatially integrated fluxes from large areas with minimum disturbance of atmospheric conditions (Denmead and Raupach, 1993). In addition, the ability to provide hourly N_2O fluxes over periods when diurnal soil temperatures cycle between freezing and thawing is important in measurement programmes aimed at quantifying N_2O emissions due to winter and spring thaw.

In this study, we present N_2O fluxes obtained using the flux-gradient method over four winter and spring thaw periods, spanning from 1993 to 1996. Our objective was to quantify N_2O emissions from several agricultural fields, and identify management practices (manure and nitrogen fertilizer addition, alfalfa ploughing, and fallowing) that contribute to increased emissions. The effect of winter conditions (snow cover, freeze/thaw cycles), and soil conditions (nitrate and ammonium concentration, and temperature) on winter and spring thaw N_2O emissions are discussed.

Material and methods

Nitrous oxide fluxes were measured at the Elora Research Station (43°49'N, 80°35'W, Conestogo silt loam), Ontario, Canada, from the end of March 1993, until the end of April 1996. In addition, N_2O fluxes were measured at the Arkell Research Station (43°30'N, 80°15'W, Burford loam), Ontario, Canada, from October 1995 to the end of April 1996. The soil characteristics for Elora were 32% sand, 52% silt, 16% clay, pH (H_2O)=7, 3.7% total organic C, and 0.3% total organic N. At Arkell, the soil was composed of 38% sand, 47% silt, 16% clay, pH (H_2O)=7.4, 2.1% total organic C and 0.2% total organic N.

Four 1-ha plots were monitored starting in 1993 at Elora: Plot (1) a bare soil fallowed for the second consecutive year to which fertilizer had last been added in September 1991, followed by a barley (*Hordeum vulgare* L.) crop fertilized with 75 kg N ha^{-1} of ammonium nitrate during each of May 1994 and May 1995; Plot (2) a bare soil fallowed for the second consecutive year to which liquid dairy cattle manure was applied in August 1992 (60 kg N ha^{-1}) and May 1993 (90 kg N ha^{-1}), followed by a soybean (*Glycine max* L.) crop during each of 1994 and 1995; (3) an alfalfa (*Medicago sativa* L.) plot established in May 1992, cut twice in the summers of 1992 and 1993 and ploughed down in October 1993, followed by a canola (*Brassica napus*) crop fertilized with 100 kg N ha^{-1} of ammonium nitrate during each of May 1994 and May 1995; and (4) an established 20-year old stand of Kentucky bluegrass (*Poa pratensis L.*) fertilized with 50 kg N ha^{-1} in May 1992 and May 1995. Two additional 2-ha plots were monitored at Elora starting in January 1994: Plot (5) a spring-ploughed corn crop receiving 100 Kg N ha^{-1} in ammonium nitrate in May 1994, followed by a no-till corn crop receiving 100 kg N ha^{-1} as anhydrous ammonia in June 1995; Plot (6) a fall-ploughed corn plot receiving 100 Kg N ha^{-1} in ammonium nitrate during each of May 1994 and May 1995. Crop residues were left on the soil surface after harvest, except for the barley plot where part of the straw was removed, and for fall-ploughing of corn and alfalfa. A summary of plots and management treatments is given in Table 1.

At Arkell, we measured N_2O fluxes from four 1-ha plots previously cropped with wheat (*Triticum aestivum* L.), consisting of: Plot (1) wheat stubble; Plot (2) oats no-till planted through wheat stubble; Plot (3) red clover no-till planted through wheat stubble; and Plot (4) control plot (wheat stubble ploughed-down). Plots 1, 2 and 3 received a fall application of liquid swine manure (75 kg N ha^{-1}), while plot 4 did not receive any treatments. Cover crop establishment was poor in plots 2 and 3, so that results from these plots were treated as replicates of plot 1, that is, as wheat stubble plots.

Nitrous oxide fluxes from each plot were calculated using the flux-gradient method:

$$\text{Flux} = -K\frac{\partial C}{\partial z}$$

where K (m^2 s^{-1}) is the eddy diffusivity of N_2O, and $\partial C/\partial z$ is the concentration gradient. Concentration gradients were estimated using a finite concentration difference, ΔC (ng N_2O m^{-3}), occurring over a vertical distance Δz(m). The eddy diffusivity compatible with Δz was estimated for each plot using a wind profile method as described by Wagner-Riddle et al. (1996). Cup anemometers were placed at four heights above each plot, and only hourly wind speeds larger than 1.5 m s^{-1} were considered for the calculation of eddy diffusivity.

The hourly concentration difference (ΔC in parts per trillion) over a height difference Δz (m) was measured using a Tunable Diode Laser Trace Gas Analyzer, TDLTGA (Edwards et al., 1994). Three

Table 1. Summary of plots and management practices monitored for N_2O flux at the locations of Elora and Arkell

Plot	Year	Description	Management practices
Elora			
1	1993	fallow, bare soil	last N fertilization September 1991
	1994, 1995	barley	75 kg N ha^{-1} NH_4NO_3 in May
2	1993	fallow + manure	liquid dairy cattle manure in August 1992 (60 kg N ha^{-1}) and May 1993 (90 kg N ha^{-1})
	1994, 1995	soybeans	-
3	1993	alfalfa	plough-down in September
	1994, 1995	canola	100 kg N ha^{-1} NH_4NO_3 in May
4	1993	grass	50 kg N ha^{-1} NH_4NO_3 in May
	1994	grass	-
	1995	grass	50 kg N ha^{-1} NH_4NO_3 in May
5	1994	spring-ploughed corn	100 kg N ha^{-1} anhydrous ammonia in May
	1995	no-till corn	100 kg N ha^{-1} NH_4NO_3 in May
6	1994, 1995	fall-ploughed corn	100 kg N ha^{-1} NH_4NO_3 in May
Arkell			
1	1995, 1996	wheat stubble	liquid swine manure (75 kg N ha^{-1}) in Oct
2	1995, 1996	oats no-till on wheat stubble	liquid swine manure (75 kg N ha^{-1}) in Oct
3	1995, 1996	red clover no-till on wheat stubble	liquid swine manure (75 kg N ha^{-1}) in Oct
4	1995, 1996	fall-ploughed wheat stubble	-

TDLTGA units were used during this study, one for each of: plots 1 to 4 at Elora, plots 5 and 6 at Elora, and plots 1 to 4 at Arkell. The wavenumber for the N_2O absorption line for both units used at Elora was 2233.333 cm^{-1}, and for the unit used at Arkell was 2236.2235 cm^{-1}. Air was drawn alternately every 5 s from two heights, typically spaced at 0.40 m above the plot surface, and then directed to a centrally located TDLTGA via approximately 70 m of tubing. The height of the lower air intake was 0.25 m for the bare soil plots, and a height within the range of 1.3 to 3 times the crop height for the vegetated plots. A site valve was used to select the plot to be sampled during each consecutive hour. The setup of sample air intakes, tubing, valves and pump are described in detail by Wagner-Riddle et al. (1997). During each measurement hour, an average N_2O concentration difference between intake heights was obtained for the monitored plot. With four 1-ha plots sampled sequentially (plots 1 to 4, at Elora and Arkell), this sampling scheme resulted in six hourly concentration differences for each plot during each measurement day. For the larger 2-ha plots cropped with corn, sequential sampling involved switching between 2 plots (plots 5 and 6 at Elora) resulting in twelve hourly concentration differences for each plot during each measurement day. Only concentration gradients measured when the wind direction at the adjacent weather station allowed for a fetch-to-height ratio of at least 50:1 (horizontal distance to height of measurement ratio) were used in the flux calculations. Due to the variable positioning of the sample intakes in the various plots this criteria resulted in a different number of total hourly or daily flux measurements in each plot.

Hourly N_2O fluxes were calculated using the concentration gradient and the eddy diffusivity as described above. For hourly fluxes, the TDLTGA has a resolution of approximately ± 10 ppt for the N_2O concentration gradient, which combined with an average eddy diffusivity of 0.05 m^2 s^{-1} would result in an error of approximately ± 2 ng m^{-2} s^{-1}. The resolution was further improved by averaging hourly fluxes to obtain daily mean fluxes.

Snow depth accumulated on each plot was measured with a ruler once every week, or as necessary after snowfalls. Hourly soil temperatures at 1, 10 and 20 cm were measured in each plot, except for 1996, using copper-constantan thermocouples encased in epoxy filled 20 cm-long copper tubing following procedure by Berard and Thurtell (1990). Rainfall and air temperatures were measured at the weather station located at the Elora research station. For 1996, hourly soil temperatures at 5, 10 and 20 cm were recorded at a weather station located at approximately 15 km from the Elora site and 2 km from the Arkell site.

Soil samples were collected from the Ap horizon during the fall of each year. Five soil cores (5 cm i.d. by 5 cm) were collected weekly at randomly selected locations at 2.5 cm below the soil surface within each plot. Moisture content was determined gravimetrically with 15 g moist soil. Moist soil (25 g) was extracted with 50 mL 0.5 *M* K_2SO_4 solution. Extract solutions were filtered, then frozen until analyzed for NH_4^+ and (NO_3^- + NO_2^-) (Tel and Heseltine, 1990).

Results and discussion

Several freeze/thaw periods occurred during January to April in 1994 and 1996, while 1995 presented one freeze/thaw cycle in January, followed by an extended cold period and a short thaw in March (Figures 1A, 2A and 3A). In 1993, the measurement period only started at the end of March, being limited to one final thaw period, and therefore, not discussed here in detail. During the freeze periods from 1994 to 1996, when the soil temperature was below 0 °C and the surface was covered by snow, daily N_2O fluxes from all plots, except the ploughed alfalfa, were small ($<$10 ng m^{-2} s^{-1}) (Figures 1, 2 and 3). Similar small fluxes were observed before the January freeze, that is during October to December in 1993, 1994 and 1995 (data not shown).

For the ploughed-down alfalfa, emissions were high throughout the fall of 1993 (data not shown), and the winter of 1994 (Figure 1B), increasing when air temperature increased, even though both air and soil temperatures were still below 0 °C. For example, during the freeze period from day 1 to 49 in 1994, daily fluxes averaged 38.8 ng m^{-2} s^{-1} (Table 2), with notable increases ($>$100 ng m^{-2} s^{-1}) during day 48 and 49, when daily mean air temperature was still averaging below 0 °C. Emissions from ploughed alfalfa continued high during the first thaw period in 1994 (day 50–53), presenting the highest average of all plots (Table 2) for that period. While N_2O emissions from alpine and subalpine snowpacks were less than 1 ng m^{-2} s^{-1} (Sommerfeld et al., 1993), Van Bochove et al. (1996) have estimated winter emissions of the order of 50 ng m^{-2} s^{-1} from a field under 60 cm of snow that had previously been cropped with barley and fertilized with 25 kg N ha^{-1}. Although we observed comparable emissions from the ploughed alfalfa plot, the other fields monitored showed fluxes of only 1 to 10 ng m^{-2} s^{-1}.

For most plots monitored, daily N_2O fluxes averaged less than 20 ng m^{-2} s^{-1} during the first thaw in 1994, 1995 and 1996 (Figures 1–3, Tables 2–4), with the exceptions of ploughed-down alfalfa in 1994, ploughed corn stubble plot in 1994 and 1995, and the ploughed-down wheat stubble and the wheat stubble that had received an application of liquid swine manure in the fall of 1995. The latter emissions at 90 ng m^{-2} s^{-1} for day 17 to 20 in 1996 were comparable to the ploughed alfalfa in 1994 (71.6 ng m^{-2} s^{-1}).

As subsequent freeze/thaw cycles occurred after the first thaw during all years, the soil temperature at 1 and 10 cm depth sequentially decreased and increased (Table 2–4). Average temperatures for the thaw periods did not always increase above 0 °C, but N_2O flux averages clearly increased during these periods. The exception was the over-wintering alfalfa (not ploughed) in 1993 (data not shown) and the grass plot during all years, which did not show a clear N_2O emission increase during any thaw periods.

The timing of emission peaks was coincidental for all plots presenting emission episodes, particularly in 1994 for fallow, manured fallow, ploughed alfalfa, standing and ploughed corn stubble (Figure 1B and C). Note that data are missing for the corn plots for most of March 1994 (days 60–90). The effect of the May 1993 manure addition to the fallow plot was still evident in the spring of 1994 with larger amplitude in the daily fluxes of the manured plot (up to 614 ng m^{-2} s^{-1}) when compared to bare soil (up to 370 ng m^{-2} s^{-1}). An exception to the timing pattern was the earlier decrease in emissions from the ploughed alfalfa and ploughed corn stubble, on day 105 when compared to day 115 for the other plots. This is evident when averages for the final thaw period (day 98–120) are compared among plots (Table 2).

In contrast to 1994, only two freeze/thaw periods were observed in 1995. As well, the final thaw period in 1995 was characterized by an initial sharp increase in temperature followed by a relatively cool period when the soil temperatures did not drop below 0 °C, and rainfall was low. Nitrous oxide emissions then decreased sharply after only 5 days during the final thaw

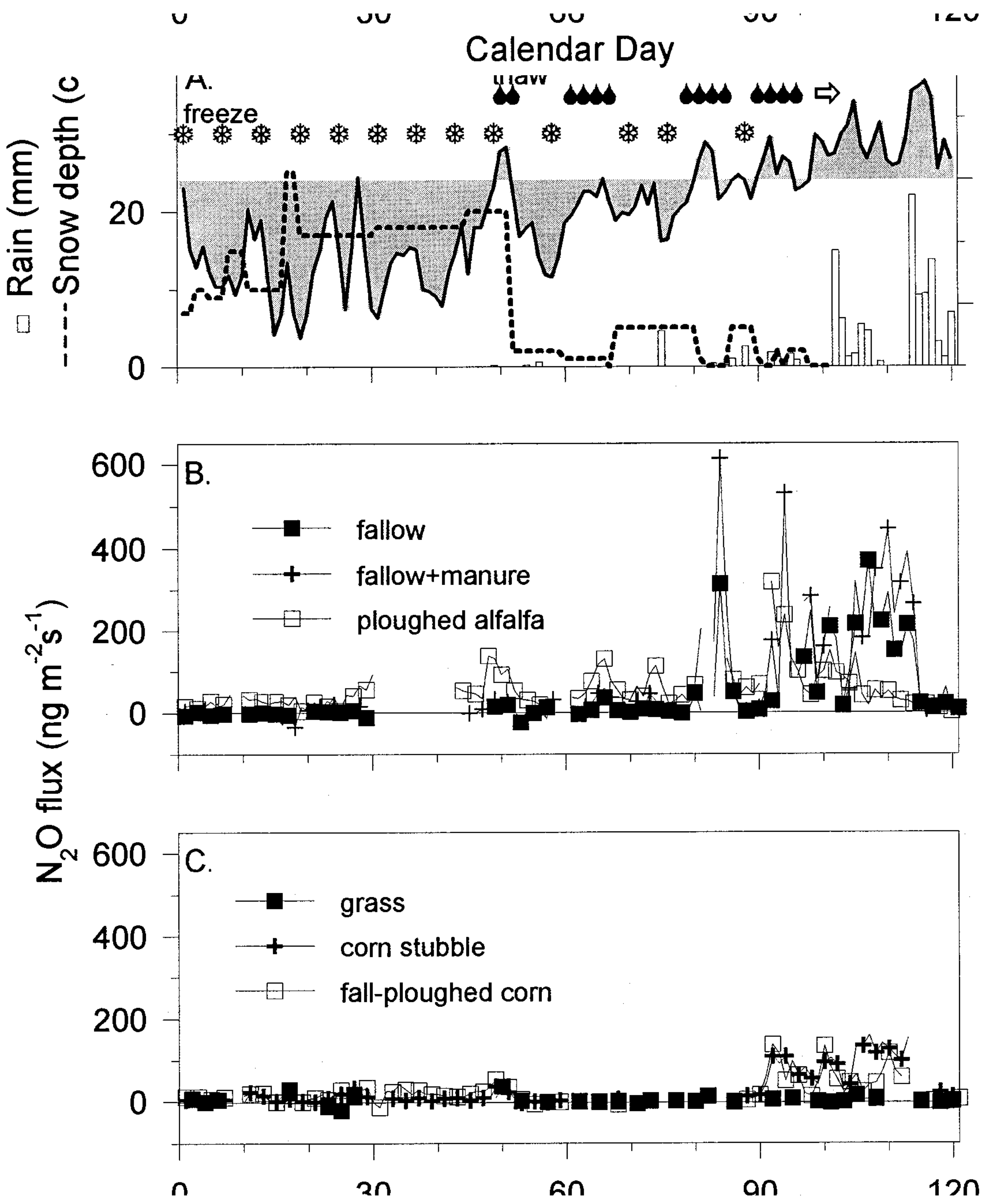


Figure 1. A. Snow depth on the ground, daily rainfall and daily air temperature recorded at Elora during January to April 1994. Freeze periods with air temperature below 0 °C and significant snow cover are indicated with the symbol ⊛, and thaw periods with air temperature above 0 °C (including hourly values) and decreasing snow cover due to melting are indicated with the symbol 💧. The start of the final thawing period is indicated by the symbol ⇒. Daily N_2O flux values measured for the same period over B. fallow soil, fallow soil that received liquid dairy cattle manure in 1992 and 1993, and an alfalfa crop ploughed down in 1993, and C. bluegrass last fertilized in 1992, corn stubble, and corn stubble ploughed-down in 1993. Only every second data point measured is shown. Note that N_2O flux data are missing from day 70 to 85 for corn plots due to equipment failure.

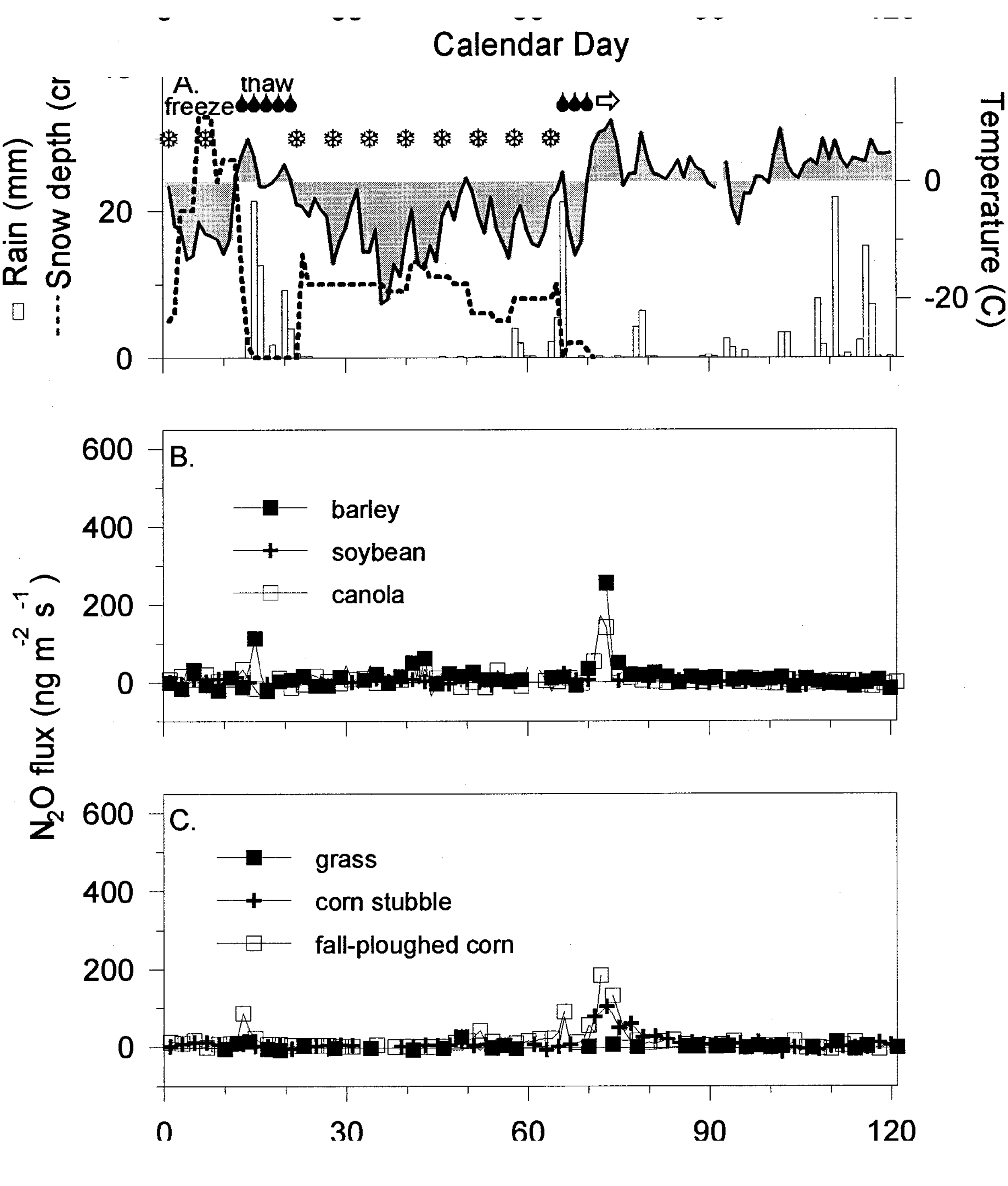


Figure 2. A. Snow depth on the ground, daily rainfall and daily air temperature recorded at Elora during January to April 1995. Freeze periods with air temperature below 0 °C and significant snow cover are indicated with the symbol ❄, and thaw periods with air temperature above 0 °C (including hourly values) and decreasing snow cover due to melting are indicated with the symbol 💧. The start of the final thawing period is indicated by the symbol ⇒. Daily N_2O flux values measured for the same period over B. plots cropped with barley, soybeans, and canola in 1994, and C. bluegrass last fertilized in 1994, corn stubble, and corn stubble ploughed-down in 1994. Only every second data point measured is shown.

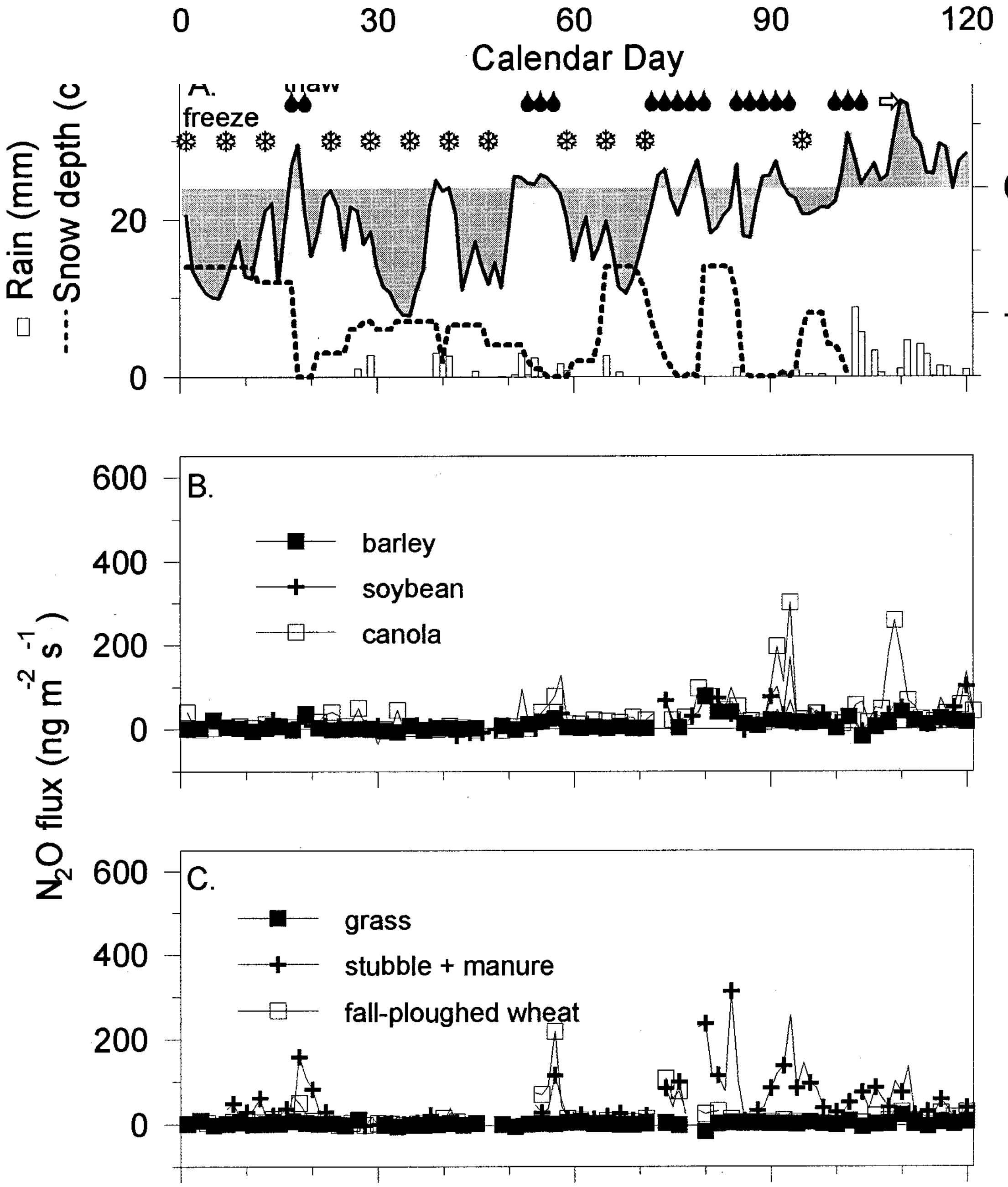


Figure 3. A. Snow depth on the ground, daily rainfall and daily air temperature recorded at Elora during January to April 1996. Freeze periods with air temperature below 0 °C and significant snow cover are indicated with the symbol ❄, and thaw periods with air temperature above 0 °C (including hourly values) and decreasing snow cover due to melting are indicated with the symbol 💧. The start of the final thawing period is indicated by the symbol ⇒. Daily N_2O flux values measured for the same period over B. plots cropped with barley, soybeans, and canola in 1995, and C. bluegrass last fertilized in 1994, wheat stubble that received liquid swine manure in 1995, and wheat stubble ploughed-down in 1995. Only every second data point measured is shown.

Table 2. Mean N_2O flux values during freezing and thawing periods from January to April 1994 measured over fallow, fallow that received manure in 1993, fall-ploughed alfalfa, grass, corn stubble and fall-ploughed corn plot. All plots except grass refer to residues from the previous year's crop. Standard deviation of N_2O flux means, and number of hourly flux values averaged over each period are given in brackets. Thawing periods are characterized as temporary, and as a final thawing period. Mean temperature is given for each riod for air and for soil depths of 1 and 10 cm under fallow soil. Total N_2O–N loss from January to April 1994 is given in kg N ha^{-1}

	Period									
	First freeze	First thaw	Second freeze	Second thaw	Third freeze	Third thaw	Fourth freeze	Fourth thaw	Final Thaw	
Day of year	1–49	50–53	54–60	61–67	68–78	79–85	86–89	90–97	98–120	
Temperature (°C)										
air	-13.2	-0.3	-10.9	-2.8	-5.2	0.7	-0.7	1.9	6.8	
1 cm	-2.0	-1.4	-5.2	-1.1	-1.9	0.1	0.2	1.7	6.6	
10cm	-1.4	-1.1	-4.7	-1.3	-1.7	-0.3	0.1	1.1	6.1	
					N_2O Flux (ng m^2 s^{-1})					Total loss (kg N ha^{-1})
Plots										
fallow	0.9	2.1	7.9	14.2	5.6	78.3	17.8	57.5	149.0	2.627
	(10.4, 113)	(22.2, 14)	(5.73, 20)	(13.3, 26)	(5.1, 43)	(118.1, 25)	(22.3, 14)	(46.8, 27)	(104.9, 96)	
fallow+manure	8.2	19.6	9.5	36.1	16.3	265.8	20.1	164.8	198.2	4.837
	(13.6, 80)	(16.2, 16)	(18.6, 13)	(18.3, 19)	(14.6, 37)	(247.8, 18)	(22.7, 12)	(185.4, 23)	(144.1, 85)	
ploughed alfalfa	38.8	71.6	16.6	75.1	53.4	107.7	57.8	153.6	55.0	3.792
	(30.3, 156)	(33.7, 20)	(20.7, 21)	(38.2, 29)	(28.4, 53)	(77.0, 17)	(16.6, 14)	(94.7, 32)	(40.3, 117)	
grass	3.4	18.5	-2.1	-0.9	-0.8	4.1	-1.8	4.4	4.8	0.209
	(11.8, 67)	(25.3, 6)	(1.6, 11)	(0.4, 7)	(2.6, 15)	(5.4, 13)	(-, 3)	(2.7, 12)	(7.1, 41)	
corn stubble	7.7	11.3	-1.2		4.5		9.7	71.3	85.1	1.666
	(8.8, 363)	(12.9, 37)	(2.2, 56)		(-, 7)		(3.6, 10)	(32.6, 69)	(51.6, 167)	
ploughed corn stubble	15.4	25.4	-2.5		-7.4		6.0	69.8	47.2	1.334
	(12.0, 378)	(18.6, 42)	(2.3, 55)		(-, 8)		(2.6, 15)	(43.8, 70)	(38.2, 167)	

period, that is, on day 75 N_2O fluxes had decreased to values smaller than 50 ng m^{-2} s^{-1} (Figures 2B and C). Consequently, N_2O flux averages during the final thaw period were lower in 1995, when compared to 1994 and 1996, with total loss of N_2O–N during the January to April period lower for most plots during 1995 (Table 3). A direct comparison of the effect of years is possible for the corn stubble and ploughed corn stubble plots. While in 1994 total N_2O–N losses from January to April for these plots were, respectively, 1.666 and 1.334 kg N ha^{-1} (Table 2), the losses over the same period in 1995 were 0.523 and 0.924 kg N ha^{-1}. Note also that flux data for corn are missing for the third and second thaw in 1994, due to equipment problems. If average N_2O fluxes from the fourth thaw period are assumed for these periods, N_2O–N total losses for 1994 are underestimated by approximately 0.27 kg N ha^{-1}. Therefore, an increased number of freeze/thaw cycles in 1994 when compared to 1995 (four versus two) resulted in an approximate doubling of N_2O emissions.

The 1996 winter and spring seasons presented a total of five cycles between freezing and thawing temperatures (Figure 3A and Table 4). While emissions increased and decreased during each consecutive freeze/thaw cycle up to day 80, emissions continued high during the fourth freeze when air temperature decreased to -5.3 °C, snow depth was larger than 10 cm, but soil temperatures stayed around the 0 °C level (Table 4). Similar results were observed for the fifth freeze cycle that occurred between day 95 and 99. Daily N_2O fluxes from all plots in 1996 were not as high as recorded for the two fallow plots in 1994, but total N_2O–N losses for the wheat stubble that received liquid swine manure in the fall of 1995 were comparable (Tables 2 and 4). In addition, soybean and canola plots also recorded higher N_2O emissions in 1996 when compared to 1995.

Table 3. Same as for Table 2 but for January to April 1995 over barley, soybean, canola, grass, corn, and ploughed corn stubble plots. Mean soil temperatures at 1 and 10 cm measured under the barley plot

	Period					
	First freeze	First thaw	Second freeze	Second thaw	Final Thaw	
Day of year	1–12	13–21	22–65	66–70	71–120	
Temperature (° C)						
air	-8.2	2.2	-8.1	-5.9	3.0	
1 cm	0.0	2.1	-1.0	-1.3	3.3	
10 cm	0.2	2.2	-0.5	-1.0	3.2	
			N_2O Flux ($ng\ m^2\ s^{-1}$			Total loss ($kg\ N\ ha^{-1}$)
Plots						
barley	8.0	18.7	8.7	11.1	16.1	0.828
	(20.9, 37)	(39.3, 39)	(20.3, 149)	(16.5, 17)	(39.3, 212)	
soybean	4.2	1.3	2.5	11.4	2.6	0.197
	(6.0, 45)	(8.3, 18)	(6.0, 118)	(7.7, 17)	(8.1, 129)	
canola	9.8	-3.3	8.9	8.2	10.9	0.585
	(6.3, 60)	(21.2, 40)	(15.4, 183)	(12.0, 22)	(31.7, 229)	
grass	0.0	1.8	-1.2	6.4	1.0	0.026
	(8.0, 13)	(8.4, 28)	(8.8, 60)	(7.7, 8)	(3.9, 105)	
corn	7.0	5.5	2.5	22.8	11.9	0.523
	(3.5, 111)	(5.5, 49)	(4.9, 295)	(28.8, 23)	(23.1, 357)	
ploughed corn	9.4	21.5	10.1	42.1	14.4	0.924
stubble	(6.3, 104)	(27.3, 64)	(11.8, 241)	(38.1, 21)	(32.5, 323)	

Table 4. Same as for Table 2 but for January to April 1996 over barley, soybean, canola, grass, wheat stubble with fall-manure, and ploughed wheat stubble plots. Mean soil temperature at 5 and 10 cm measured under grass at the weather station

	Period											
	First freeze	First thaw	Second freeze	Second thaw	Third freeze	Third thaw	Fourth freeze	Fourth thaw	Fifth freeze	Fifth thaw	Final Thaw	
Day of year	1–16	17–20	21–51	52–58	59–71	72–80	81–84	85–94	95–99	100–104	105–120	
Temperature (°C)												
air	-11.1	-0.5	-8.6	0.7	-9.2	0.1	-5.3	-0.7	-3.1	2.9	5.4	
5 cm	-5.2	-0.4	-5.0	-0.4	-3.1	-1.1	-0.2	0.0	-0.2	1.5	4.6	
10 cm	-4.5	-0.5	-4.5	-0.4	-2.6	-1.0	-0.1	0.0	0.0	1.0	3.9	
						N_2O Flux ($ng\ m^2\ s{-}1$)						Total loss ($kg\ N\ ha^{-1}$)
Plots												
barley	6.5	11.5	1.1	30.3	6.7	34.3	40.9	22.5	18.1	9.1	19.6	0.897
	(6.9, 71)	(18.1, 21)	(4.4, 143)	(29.9, 30)	(3.5, 61)	(33.5, 16)	(9.5, 22)	(19.9, 51)	(3.4, 22)	(17.4, 21)	(11.3, 75)	
soybean	7.8	13.2	1.4	16.0	5.4	37.6	56.0	46.0	27.2	6.0	36.6	1.196
	(6.8, 50)	(6.2, 18)	(6.4, 137)	(14.3, 24)	(3.2, 49)	(27.7, 25)	(21.1, 20)	(54.3, 40)	(13.7, 21)	(8.9, 15)	(31.3, 75)	
canola	9.6	17.8	8.0	45.7	19.2	47.3	74.9	86.4	33.4	41.7	78.3	2.343
	(10.4, 50)	(12.9, 21)	(17.9, 147)	(46.4, 30)	(5.8, 66)	(29.4, 29)	(21.9, 22)	(95.8, 48)	(13.0, 17)	(24.5, 16)	(71.0, 81)	
grass	1.5	3.3	0.0	0.6	1.1	-0.6	3.8	2.8	-0.2	0.1	3.7	0.084
	(2.7, 67)	(2.9, 22)	(3.7, 129)	(1.5, 23)	(2.2, 59)	(8.1, 22)	(2.8, 21)	(2.9, 51)	(5.9, 22)	(4.7, 23)	(6.1, 76)	
stubble + manure	28.4	90.0	8.7	56.3	18.0	119.3	156.1	90.1	80.0	45.7	50.0	3.156
	(15.9, 52)	(61.6, 18)	(11.6, 102)	(34.8, 18)	(5.9, 58)	(92.7, 24)	(107.1, 15)	(72.1, 47)	(45.9, 21)	(22.3, 11)	(34.5, 79)	
ploughed wheat	6.6	36.9	3.1	99.0	9.4	55.6	24.4	9.1	3.9	6.2	18.9	1.163
stubble	(3.2, 41)	(25.7, 21)	(4.0, 101)	(69.0, 18)	(4.9, 61)	(32.6, 21)	(10.3, 13)	(7.0, 45)	(1.4, 19)	(2.3, 15)	(8.6, 75)	

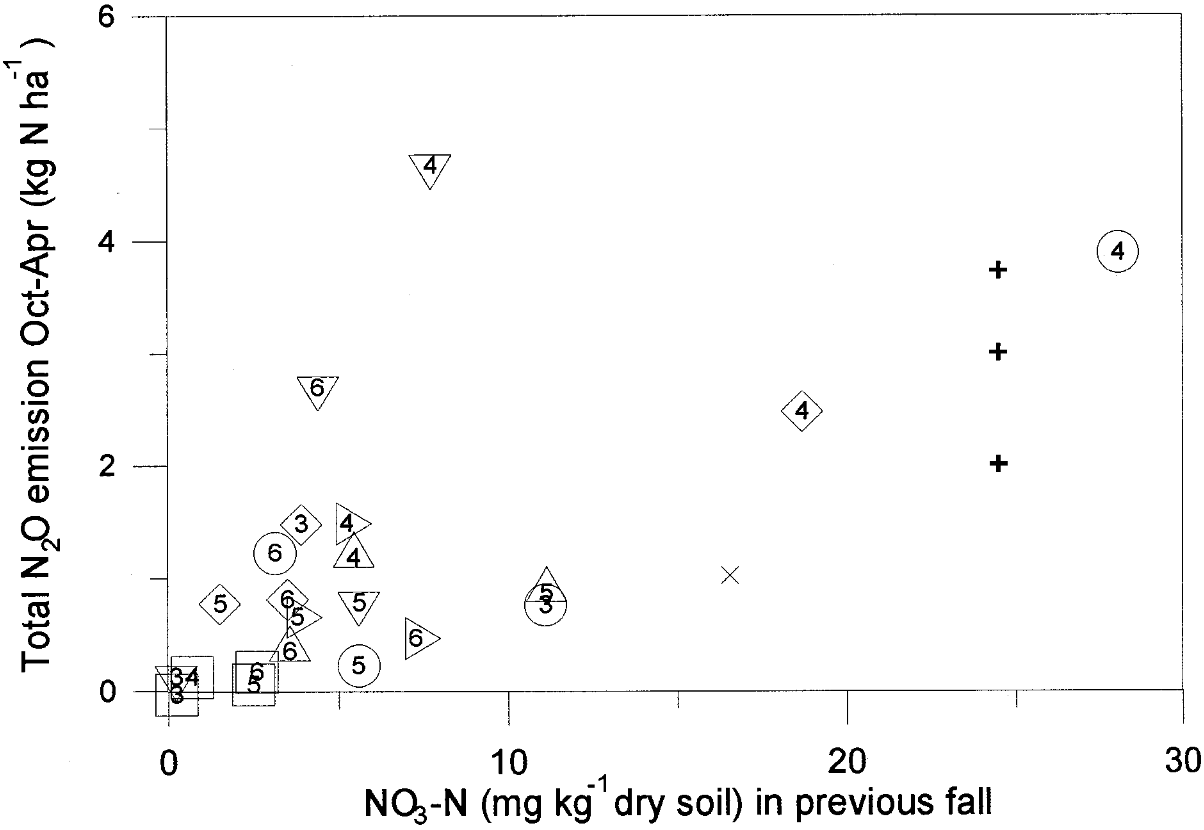


Figure 4. Total N_2O emission accumulated between January and April of each studied year for all plots studied as a function of the soil nitrate concentration in the previous fall. Soil sampling dates varied between September and November. Data points for Elora are shown with: ◇ plot 1 (fallow/barley), ○ plot 2 (manure/soybean), ▽ plot 3 (ploughed alfalfa/canola), □ plot 4 (grass), ▹ plot 5 (spring-ploughed corn/no-till), and △ plot 6 (fall-ploughed corn). Data points for Arkell 1996 are shown with: + plot 1, 2, 3 (wheat stubble + manure) and × plot 4 (ploughed stubble). Labels on each data point indicate the winter and spring thaw year at Elora (3=93, 4=94, 5=95 and 6=96).

From our results it is clear that both freeze/thaw cycles and management practices play a role in the magnitude of winter and spring emissions. The total N_2O–N emissions from October to April in each studied year were correlated to the nitrate concentration in the soil measured during the previous fall (r=0.70, Figure 4). For vegetated plots, such as grass, or alfalfa before ploughing in 1993, soil nitrate concentrations in the fall were less than 5 mg kg^{-1} dry soil and total N_2O–N emissions less than 0.2 kg N ha^{-1}. In the other extreme, the fallow and manured plots presented NO_3–N soil concentrations in the fall that were higher than 20 mg kg^{-1} dry soil, contributing to the larger than 2 kg N ha^{-1} of N_2O–N loss between January and April of the following year. The only plot that did not fit this pattern very well was plot 3, where an alfalfa crop was ploughed down in the fall of 1993. Although NO_3–N soil concentrations were less than 10 mg kg^{-1} dry soil, N_2O–N losses during October to April in 1994 and 1996 exceeded 2 kg N ha^{-1}. Rapid cycling between nitrogen in the organic form and NO_3 could explain the low NO_3 detected in the soil solution. Surprisingly, organic N from alfalfa ploughing in 1993 still seemed to be affecting N_2O emissions two years after incorporation into the soil.

It is believed that increased emissions due to freeze/thaw cycles are a combination of physical release of N_2O, and N_2O production in the surface soil, and N_2O diffusion from the subsurface soils (Goodroad and Keeney, 1984b). Christensen and Christensen (1991) showed that organic matter becomes available for denitrification when microbes are subjected to a killing freeze and aggregates are disintegrated due to freeze/thaw cycles. Flessa and Dörsch (1995) concluded that the marked increase in N_2O emissions was very likely due to increased denitri-

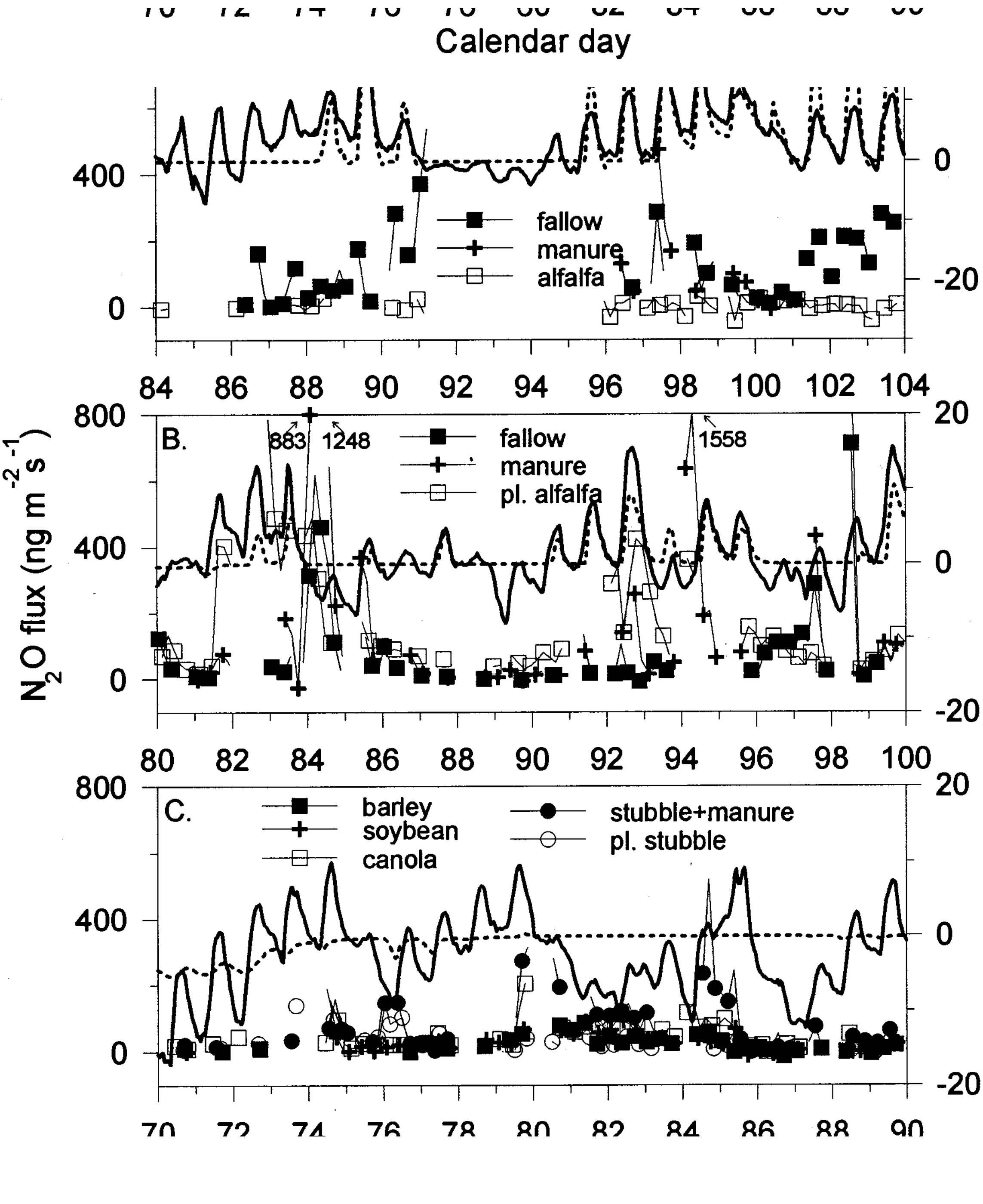


Figure 5. Hourly air (solid line) and soil temperatures at 1 cm depth (or 5 cm for 1996) (dotted line) for the fallow or barley residue plot (or weather station for 1996), and hourly N_2O fluxes for A. fallow, manured fallow, and alfalfa crop in 1993; B. fallow, manured fallow, and ploughed alfalfa in 1994; and C. plots previously cropped with barley, soybeans, canola, and wheat (including manure stubble and ploughed-down stubble), in 1996. Labels for C. indicate values that are off-scale. Only every second data point measured is shown.

fication activity in the uppermost thawed soil layer. During their study year, peak emissions were observed during the first freeze/thaw cycle, and citing Melin and Nommik (1983) they suggested that the suppression of nitrous oxide reductase at low temperatures could explain the initial peak observed.

In contrast, we observed N_2O fluxes peaking at the third or fourth freeze/thaw cycle, during some years, respectively in 1994 and 1996. Upon close examination of the hourly N_2O fluxes and air and soil temperatures, it can be seen that maximum N_2O fluxes occurred frequently during periods of cooling or minimum temperatures (Figure 5). Typically, a three to four-day period of increasing air temperatures, with soil temperatures increasing above freezing, followed by a cold period when soil temperatures returned to 0 °C, resulted in increased fluxes. Peaks occurred during the cooling phase of the diurnal temperature wave, as on day 91 (5:00 h) in 1993 (Figure 5A), days 84 (6:00 h) and 94 (7:00 h) in 1994 (Figure 5B), and days 76 (9:00 h) and 79 (21:00 h) in 1996 (Figure 5C). But also on the warming phase of the diurnal temperature wave, as on days 97 (10:00 h) in 1993, days 85 (10:00 h), 94 (7:00 h), 97 (13:00 h) and 98 (13:00 h) in 1994, and day 84 (17:00 h) in 1996. Two possible mechanisms might be playing a role for this lack of coupling between the soil temperature and the occurrence of N_2O emission episodes: (1) a shortage of substrate for denitrification (carbon and/or nitrate) may be limiting the production of N_2O as temperatures increase on the ascending phase of the temperature wave; and (2) a suppression of nitrous oxide reductase at low temperatures (Melin and Nommik, 1983) may be delaying the reduction of N_2O produced during the ascending phase of the temperature wave. This second mechanism may be also allowing for the accumulation of N_2O once temperatures decrease and the reduction of N_2O to N_2 stops on the descending part of the temperature wave. It is also obvious that level of nitrate in soils was limiting N_2O production by denitrification, since plots with low nitrate levels, such as alfalfa in 1993 (Figure 5A) did not present any increased fluxes. In addition, Figure 5 clearly shows that emissions occur after a change in hourly temperature from above freezing to below freezing, or vice-versa, but that these cycles do not always result in high emissions even in plots with high nitrate levels.

Conclusions

Nitrous oxide emissions from agricultural fields from January to April over four years in Ontario, Canada, ranged between 0 and 4.8 kg N ha^{-1}. Compared to yearly estimates ranging between 0.5 and 4.1 kg N ha^{-1} based on fertilizer applications (Wagner-Riddle et al., 1997), these thaw emissions are substantial and should be considered in the nitrous oxide budgets in regions where thaw periods occur.

While other studies have measured high emissions during soil thaw, our study indicates that agricultural management can play a role in mitigating these emissions. Our data show that fallowing, manure application and alfalfa incorporation in the fall lead to high spring emissions, while the presence of plants (as in the case of alfalfa and grass) can result in negligible emissions during thaw. This presents an opportunity for mitigation of N_2O emissions through the use of over-wintering cover crops.

Emissions from plots cultivated with crops that received nitrogen fertilizer additions at planting time were much smaller than those resulting from fall applications of manure or alfalfa on uncropped plots, but it is not clear from our results if a fall nitrogen application on a vegetated plot would also result in high thaw emissions.

Hourly fluxes monitored in this study indicated a lack of coupling between the N_2O emission peaks and soil temperature maxima. In addition, the temporal variability of N_2O fluxes on an hourly basis highlighted the importance of continuous monitoring for the quantification of seasonal emission totals.

Acknowledgements

This research project was funded by grants from the Environmental Research Program of the Ontario Ministry of Environment and Energy, the Natural Sciences and Engineering Research Council of Canada, the Ontario Ministry of Agriculture, Food and Rural Affairs, and by research contracts through the Green Plan Initiative/Agriculture Canada. The technical assistance of Mr. R. Sweetman was invaluable for the conduction of the field experiments. The assistance of Ms. Adriana Furon in data analysis was greatly appreciated.

References

Berard R & Thurtell GW (1990) Soil temperature measurements. In: Goel NS & Norman JM (eds) Instrumentation for Studying Vegetation Canopies for Remote Sensing in Optical and Thermal Infra-red Regions, pp 293–300. Glasgow, UK: Remote Sensing Reviews 5, Harwood Academic

Cates RL & Keeney DR (1987) Nitrous oxide production throughout the year from fertilized and manured maize fields. J Environm Qual 16: 443–447

Chen Y, Tessier S, MacKenzie AF & Laverdière MR (1995) Nitrous oxide emission from an agricultural soil subjected to different freeze-thaw cycles. Agric Ecosys Environm 55: 123–128

Christensen S & Christensen BT (1991) Organic matter available for denitrification in different soil fractions: effect of freeze/thaw cycles and straw disposal. J Soil Sci 42: 637–647

Christensen S & Tiedje JM (1990) Brief and vigorous N_2O production by soil at spring thaw. J Soil Sci 41: 1–4

Denmead OT & Raupach MR (1993) Methods for measuring atmospheric gas transport in agricultural and forest systems. In: Harper LA et al. (eds) Agricultural Ecosystem Effects on Trace Gases and Global Climate Change, pp 19–44. ASA Spec Publ 55. Madison, Wisc: ASA, CSSA, and SSSA

Edwards GC, Neumann HH, den Hartog G, Thurtell GW & Kidd GE (1994) Eddy correlation measurements of methane fluxes using a tunable diode laser at the Kinosheo Lake tower site during the Northern Wetlands Study (NOWES). J Geophys Res 99: 1511–1517

Flessa H & Dörsch P (1995) Seasonal variation of N_2O and CH_4 fluxes in differently managed arable soils in southern Germany. J Geophys Res 100: 23115–23124

Goodroad LL & Keeney DR (1984a) Nitrous oxide emission from forest, marsh, and prairie ecosystems. J Environm Qual 13: 448–452

Goodroad LL & Keeney DR (1984b) Nitrous oxide emissions from soils during thawing. Can J Soil Sci 64: 187–194

Goodroad LL, Keeney DR & Peterson LA (1984) Nitrous oxide emissions from agricultural soils in Wisconsin. J Environm Qual 13: 557–561

Isermann K (1994) Agriculture's share in the emission of trace gases affecting the climate and some cause-oriented proposals for sufficiently reducing this share. Environ Pollut 83: 95–111

Melin J & Nommik H (1983) Denitrification measurements in intact soil cores. Acta Agri Scand 33: 145–151

Mosier AR, Duxbury JM, Freney JR, Heinemeyer O & Minami K (1996) Nitrous oxide from agricultural fields: assessment, measurement and mitigation. Plant and Soil 181: 95–108

Sommerfeld RA, Mosier AR & Musselman RC (1993) CO_2, CH_4 and N_2O flux through a Wyoming snowpack and implications for global budgets. Nature 361: 140–142

Tel DA & Heseltine C (1990) The analyses of KCl soil extracts for nitrate, nitrite and ammonium using a TRAACS 800 analyzer. Commun Soil Sci Plant Anal 21: 1681–1688

Van Bochove E, Jones HG, Pelletier F & Prévost D (1996) Emission of N_2O from agricultural soil under snow cover: a significant part of N budget. Hydrol Proc 10: 1545–1549

Wagner-Riddle C, Thurtell GW, King KM, Kidd GE & Beauchamp EG (1996) Nitrous oxide and carbon dioxide fluxes from a bare soil using a micrometeorological approach. J Environ Qual 25: 898–907

Wagner-Riddle C, Thurtell GW, Kidd GK, Beauchamp EG & Sweetman R (1997) Estimates of Nitrous Oxide Emissions from Agricultural Fields over 28 months.Can J Soil Sci 77: 135–144

Watson RT, Rodhe H, Oeschger H & Siegenthaler U (1990) Greenhouse gases and aerosols. In: Houghton JT et al. (eds) Climate Change: The IPCC Scientific Assessment, pp 7–40. Cambridge Univ Press, Cambridge, UK