

Exhibit A

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Advancing Climate-Neutral Agriculture in New York *Viewpoint*

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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) emissions¹⁰ 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-Scoping-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 emissions²² 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.aag0216>.

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.

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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>.

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Exhibit B

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

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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 $\delta^{15}\text{N}$; 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 $\delta^{15}\text{N}$ 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 $\leq 15\%$ 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; $\delta^{15}\text{N}$ natural abundance; land-use history; light fraction organic matter; nitrification; nitrogen mineralization; pastures; phosphorus; reforestation; vegetation effects; woodlots.

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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 $\delta^{15}\text{N}$ 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 influ-

enced 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	<i>Tsuga canadensis</i> , <i>Pinus strobus</i>
	Woodlot	132	Canton	1890	44	<i>T. canadensis</i>
	Pastured	218	Canton	1880	57	<i>Pinus resinosa</i> , <i>P. strobus</i>
	Pastured	227	Scituate	1880	60	<i>P. resinosa</i>
	Cultivated	155	Scituate	1908	44	<i>P. strobus</i> , <i>Acer rubrum</i>
	Cultivated	215	Canton	1908	46	<i>T. canadensis</i> , <i>Picea rubens</i> , <i>P. strobus</i> , <i>P. resinosa</i>
Hardwood	Woodlot	135	Scituate	1890	37	<i>A. rubrum</i> , <i>Quercus rubra</i> , <i>T. canadensis</i>
	Woodlot	46	Canton	1890	37	<i>Q. rubra</i> , <i>A. rubrum</i> , <i>Fagus grandifolia</i>
	Pastured	43	Canton	1908	32	<i>A. rubrum</i> , <i>Betula alleghaniensis</i>
	Pastured	91	Canton	1908	30	<i>A. rubrum</i> , <i>Q. rubra</i>
	Cultivated	235	Canton	1908	28	<i>Q. rubra</i> , <i>Acer saccharum</i> , <i>P. strobus</i>
	Cultivated	134	Canton	1908	46	<i>Fraxinus americana</i> , <i>Prunus serotina</i> , <i>A. rubrum</i> , <i>Acer pensylvanicum</i>

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/cm³). 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 \geq four times until no floating material remained. The light and heavy fraction were washed over Whatman GF A/E filter paper with \geq 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 $\delta^{15}\text{N}$ 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.

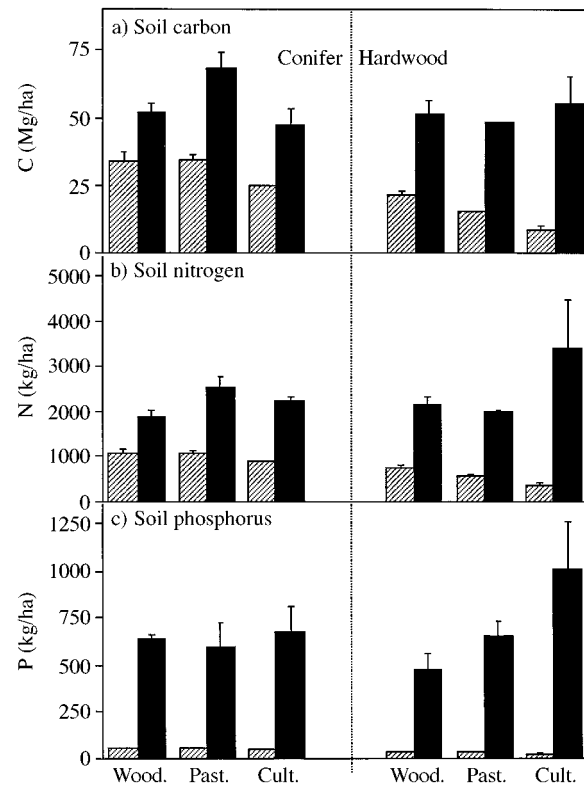


FIG. 1. Total C, N, and P in forest floor and mineral soil by prior land use (Wood. = woodlot; Past. = pastured; Cult. = cultivated) and present vegetation. Hatched bars depict data for the forest floor, and solid bars depict data for mineral soil; error bars indicate +1 SE ($n = 2$ sites).

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 (non-parametric 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

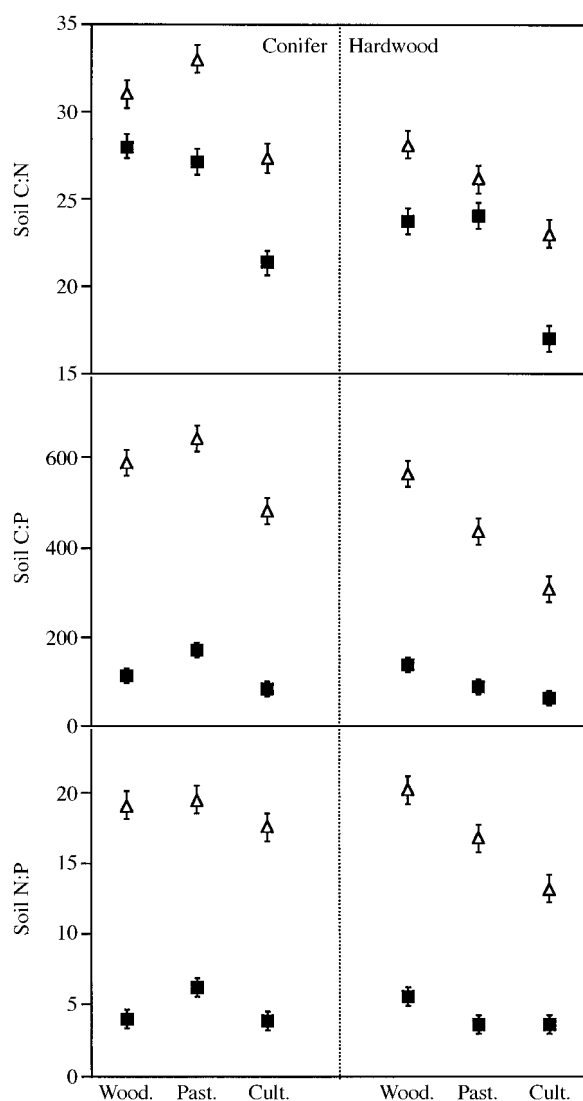


FIG. 2. Ratios of total C, N, and P in forest floor (open triangles) and 0–15 cm soil (solid squares) by prior land use (abbreviations as in Fig. 1) and present vegetation. Error bars indicate ± 1 SE ($n = 2$).

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 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 $\delta^{15}\text{N}$

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 $\delta^{15}\text{N}$ 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).

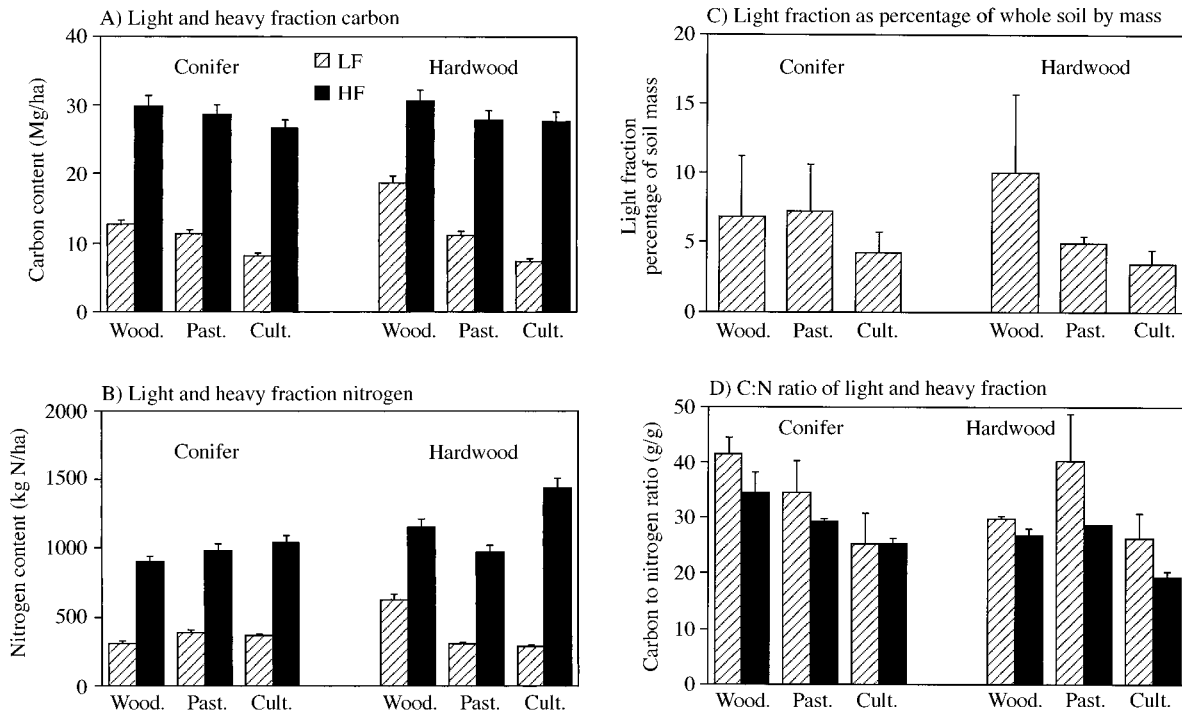


FIG. 3. Light-fraction (LF) and heavy-fraction (HF) mass, C:N ratio, and C and N content in mineral soil collected in May 1995 (0–15 cm depth). Error bars indicate + 1 SD. Prior land use abbreviations are as in Fig. 1.

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	1321 ^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	1009 ^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·ha ⁻¹ ·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·ha ⁻¹ ·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

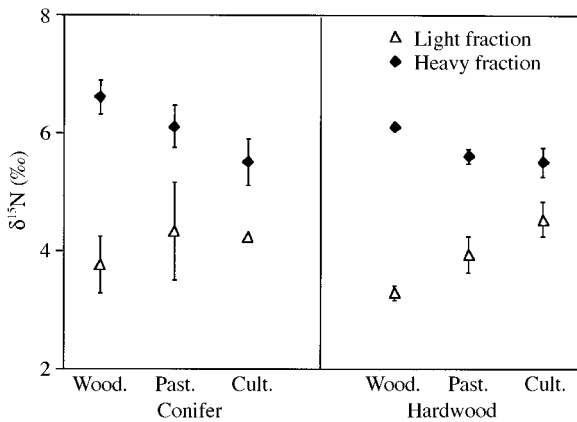


FIG. 4. Natural abundance of $\delta^{15}\text{N}$ in light and heavy fraction soil. Error bars represent ± 1 SD ($n = 2$ sites). Prior land-use abbreviations are as in Fig. 1.

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-

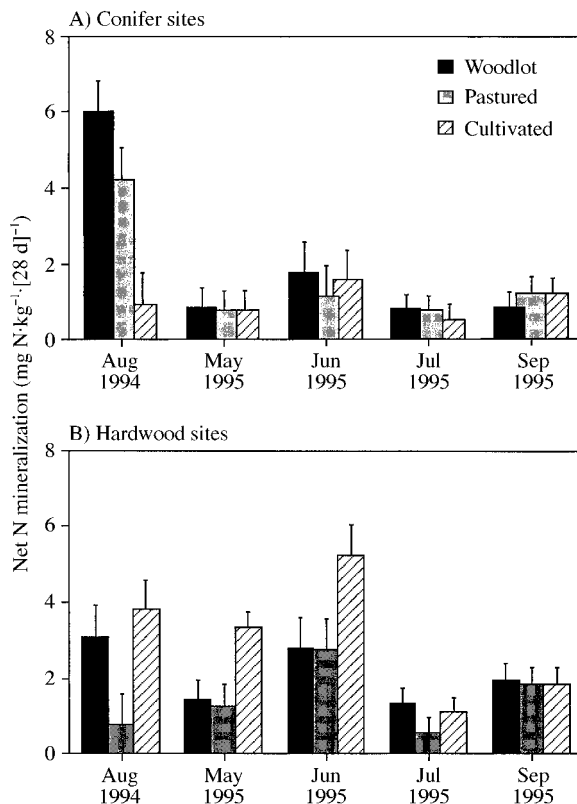


FIG. 5. In situ net nitrogen mineralization rates per gram soil in the 0–15 cm mineral soil. Error bars indicate +1 SE ($n = 2$ plots).

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·ha ⁻¹ ·yr ⁻¹)	Nitrification (kg N·ha ⁻¹ ·yr ⁻¹)	Nitrification as a percentage of N mineral- ization
Conifer			
Cultivated	7.9 ^b (3.5)	0.8 ^{ab} (0.5)	9.3 (2.4)
Pastured	8.2 ^b (0.6)	0.5 ^b (0.2)	6.0 (2.1)
Woodlot	7.7 ^b (0.1)	0.1 ^b (0.0)	1.6 (0.1)
Hardwood			
Cultivated	18.1 ^a (0.7)	4.3 ^a (0.3)	23.7 (2.5)
Pastured	11.8 ^{ab} (1.6)	0.6 ^{ab} (0.2)	4.6 (1.3)
Woodlot	14.7 ^{ab} (0.0)	0.6 ^{ab} (0.1)	4.2 (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·ha⁻¹·yr⁻¹ under hardwoods and 0.68 Mg·ha⁻¹·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 $\sim 65\%$ 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 $\sim 65\%$ 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-

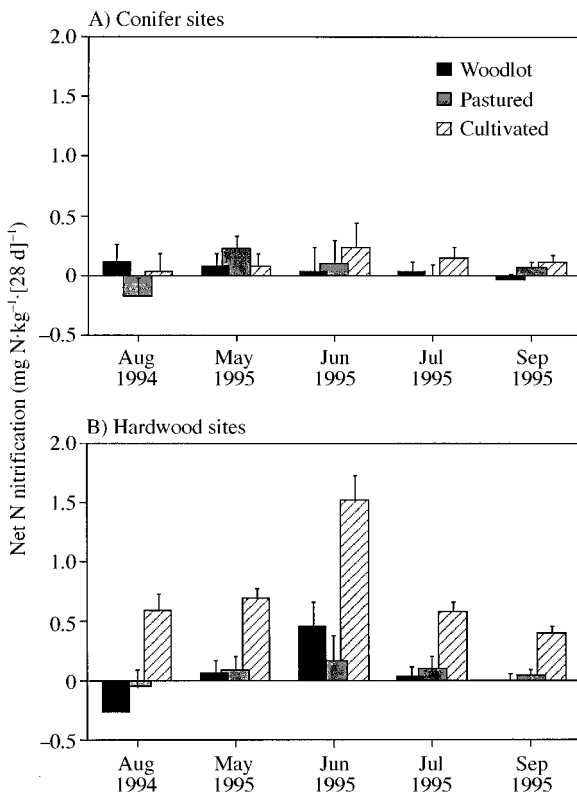


FIG. 6. In situ net nitrification rates per gram soil in the 0–15 cm mineral soil. Error bars indicate +1 SE ($n = 2$ plots).

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 $\delta^{15}\text{N}$ 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 $^{15}\text{N}/^{14}\text{N}$ 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 through leaching, denitrification, and ammonia volatilization result in preferential losses of ^{14}N from soils, resulting in an increase in soil $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$. Soil $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ in several tropical grasslands (Piccolo et al. 1994). The heavy fraction (mineral-associated) $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ values between the two fractions. Tiessen et al. (1984) reported that long-term cultivation of a grassland soil resulted in lower $\delta^{15}\text{N}$ in the bulk soil and coarse clay and silt fractions, but higher $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ values in the previously cultivated soils suggest a tighter coupling between the two fractions in the cultivated soils than in woodlots.

The $\delta^{15}\text{N}$ 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

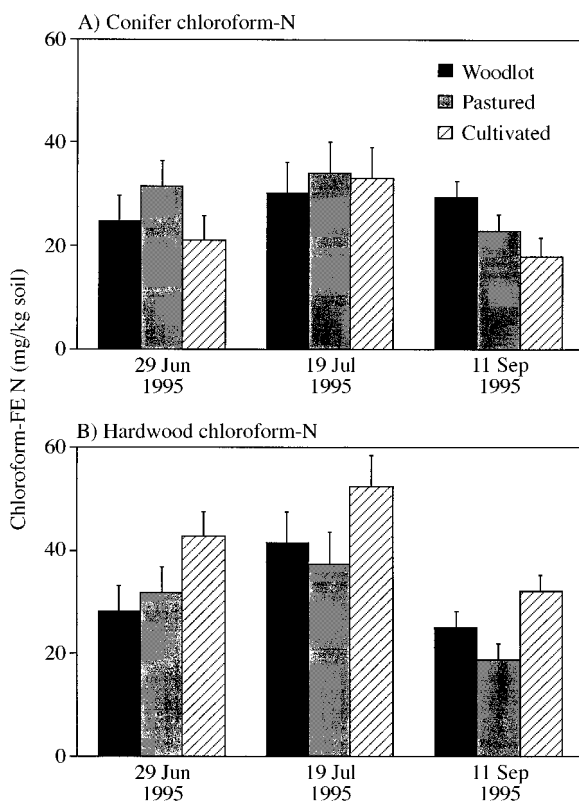


FIG. 7. Chloroform fumigation-extraction (FE) nitrogen as a relative estimate of microbial N by vegetation type and land use. Error bars represent +1 SD ($n = 2$ plots).

using these values. The $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ values of amendments to be relatively low. The $\delta^{15}\text{N}$ values of fresh animal wastes may not be outside the range of soil values (Macko and Ostrom 1994), therefore we cannot use the actual $\delta^{15}\text{N}$ 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 ef-

fect 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 $\delta^{15}\text{N}$ 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 $\delta^{15}\text{N}$ 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 $\text{kg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ (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 ≥ 100 yr by the Sandersons and subsequent land-owners (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.

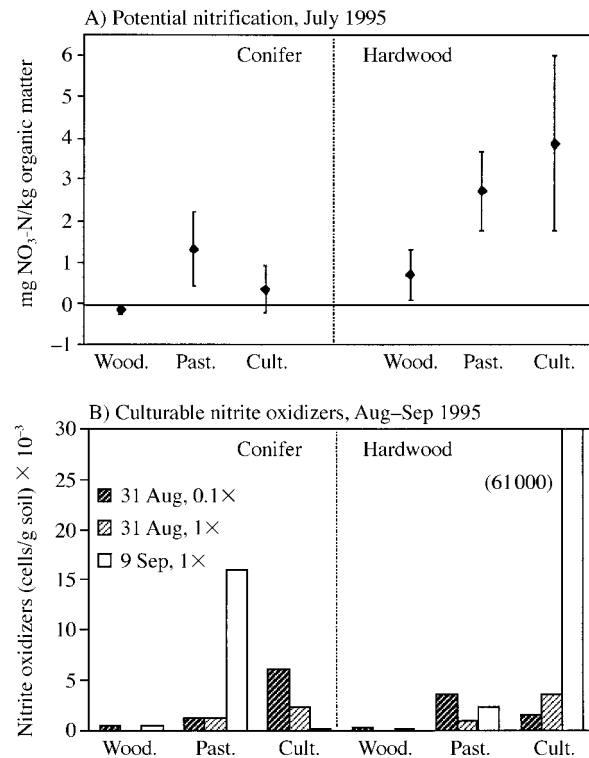


FIG. 8. Potential nitrification and most-probable-numbers (MPN) estimates of nitrite oxidizers. Error bars represent ± 1 SE for potential nitrification. For MPN estimates, media nitrite concentrations were full strength (1 \times) or diluted to one-tenth of those recommended (0.1 \times) in Schmidt and 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 ($3.2 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ 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 McLaugherty 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.

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Exhibit C



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

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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 km²). Tree abundance was averaged over town-wide samples of multiple forest types, integrating proportions of taxa at a local scale (10² km²). 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

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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, proprietary 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 proprietary 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 proprietary 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 spa-

tially 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 ($c.10^{-2}$ km² extent), through the regional assemblages of those communities in a landscape ($c.10^2$ km² extent), to groupings of those landscapes by similar physiognomy or constituent flora ($c.10^5$ km² 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 north-eastern 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 pre-settlement 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. Secondly, 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 km² 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

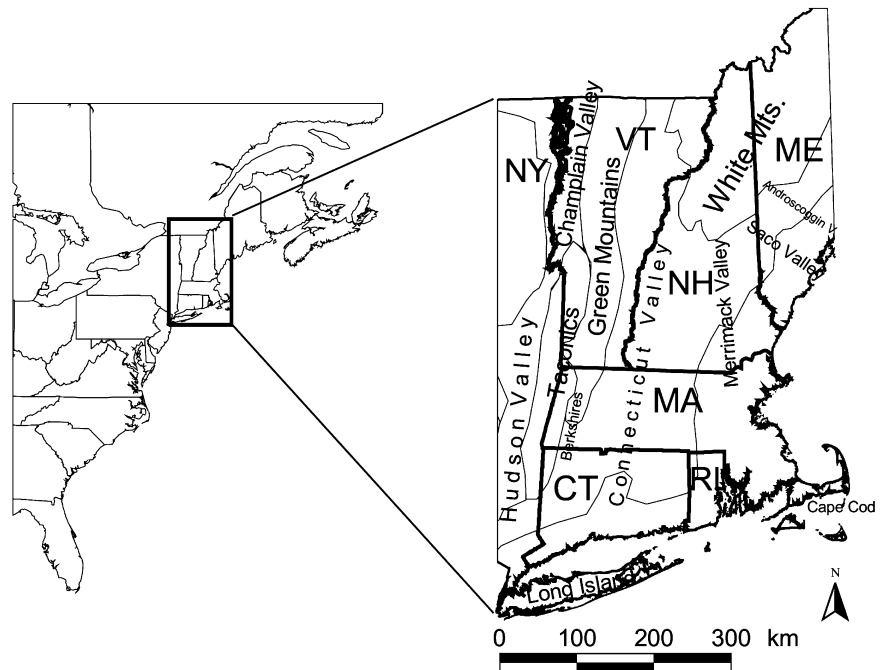


Figure 1 Study area on the east coast of North America and indicating the location of place names cited. State outlines in heavy lines New York (NY), Vermont (VT), New Hampshire (NH), Maine (ME), Massachusetts (MA), Connecticut (CT) and Rhode Island (RI). Outlines of the Physiographic Sections of Fenneman (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 (Melnik, 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 km².

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 km² 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 ($T_{\text{bar}} - T_{\text{amp}}$), mean July temperature ($T_{\text{bar}} + T_{\text{amp}}$), solved for the length of the frost-free growing season [$G_{\text{Season}} = \text{curve days } (d) > 10 \text{ } ^\circ\text{C}$], or integrated for Growing Degree Days [$\text{GDD } (^\circ\text{days}) = \text{area} > 10 \text{ } ^\circ\text{C}$].

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 km² is close to the classic 100 km² proprietary 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 km². 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						Overall	Range
	ME	NH	VT	MA	CT and RI	NY		
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

Taxa	ME	NH	VT	MA	CT and RI	NY	All trees	All Towns		
	Towns Sample stems (%)	21 4473 (%)	47 17,735 (%)	97 23,496 (%)	118 52,403 (%)	67 45,560 (%)		39 10,265 (%)	389 Mean (%)	389 Maximum (%)
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 gen-

era, 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 km²) 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

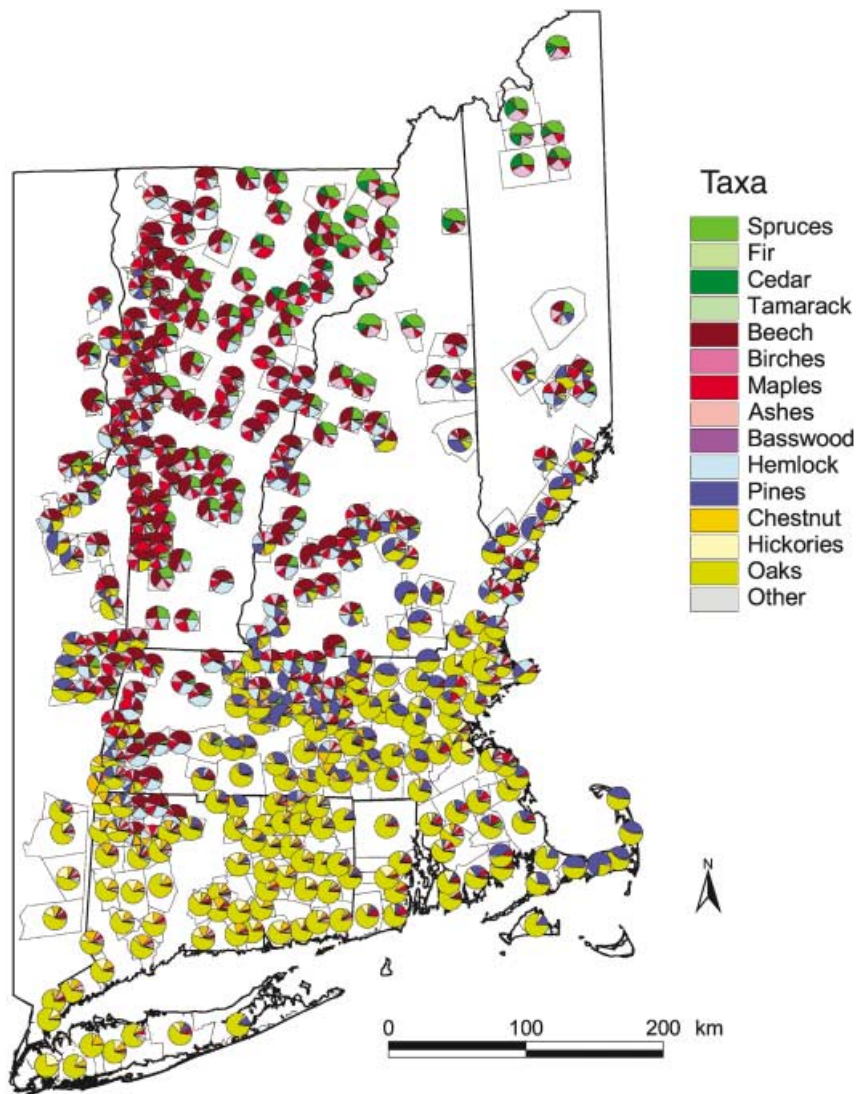


Figure 2 Spatial distribution of the proportions of genera in presettlement surveys arrayed on sample towns. Sample towns outlines indicated in light lines.

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

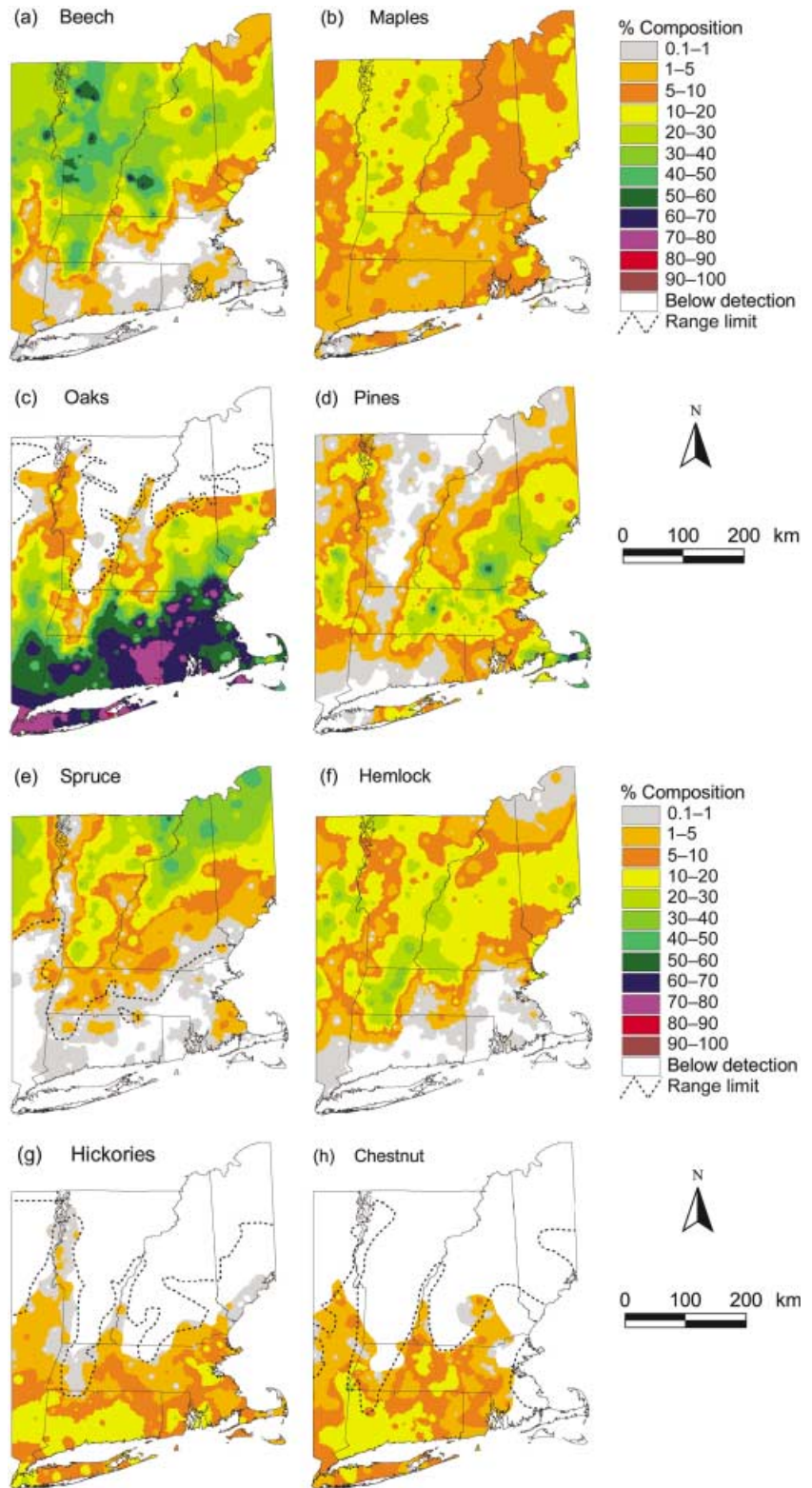


Figure 3 Maps of abundance surfaces of taxa: (a) beech, (b) maples, (c) oaks, and (d) pines, (e) spruces, (f) hemlock, (g) hickories and (h) chestnut in presettlement New England. The interpolated surfaces are shown divided into classes of relative frequency, from the general detection limit of 0.1%. The explicit minimum resolution is the town-scale (*c.*100 km²). 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 sig-

nificant 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 south-western 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 south-eastern 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

Taxa	Towns	Cluster							
		1	2	3	4	5	6	7	8
Trees		1441	11,230	17,004	17,603	21,149	11,753	50,022	23,730
		(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)
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 low-

land ‘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.

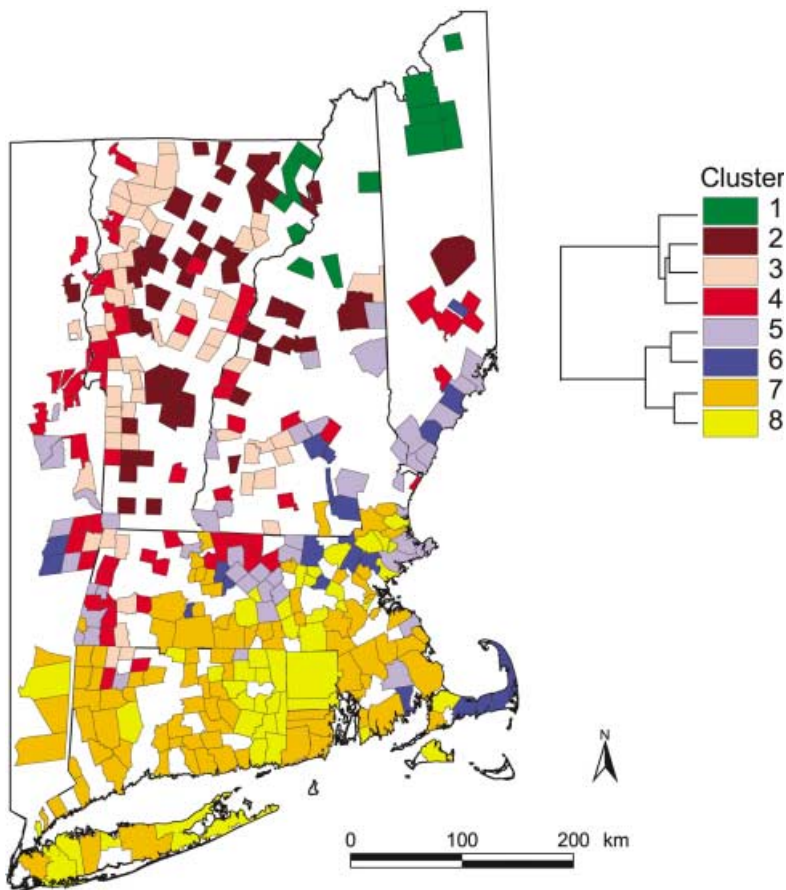


Figure 4 Geographical distribution of towns as classified by Cluster Analysis into eight clusters. The dendrogram next to the legend is scaled proportional to objective distance at which the cluster units are joined (i.e. 'fusion distance') in the hierarchy.

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.

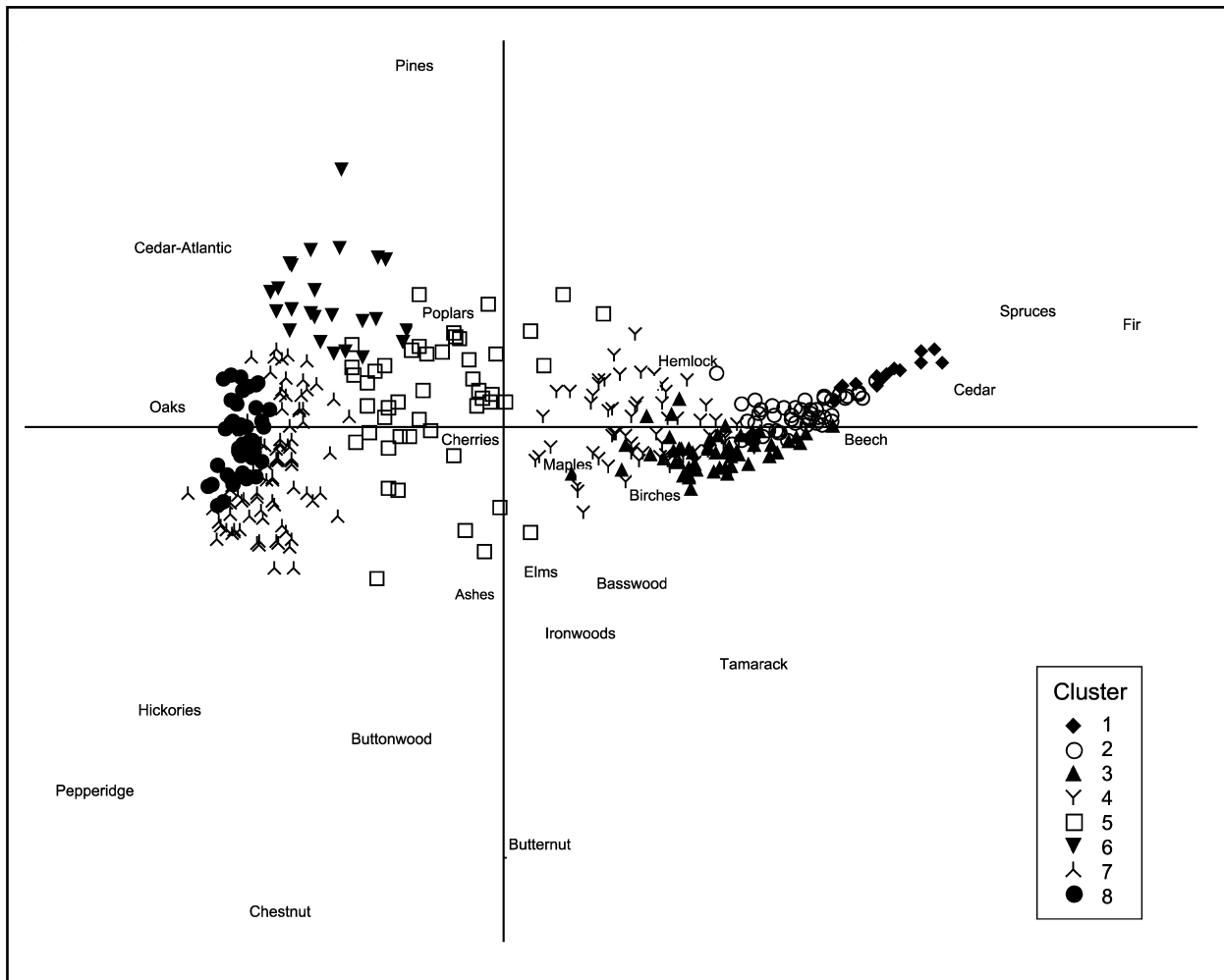


Figure 5 Detrended correspondence analysis ordination of twenty-two genera and 389 towns in the presettlement surveys of New England. Towns are classified by their cluster identity and the genus groupings are labelled at their centroid.

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 [$P \leq 0.01$, 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 ($P \geq 0.01$) 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

Parameter	n = 389	Cluster								F [†]	d.f. = [7,381]	
		All towns		1	2	3	4	5	6			7
	Mean	389	389	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean
		Min.	Max.	SD	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean
FCE-Elevation [‡] (m)	-610	-1044	41	244	-98	-265	-442	-543	-676	-781	-823	-842
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
GSeason ^{**} (days)	164	128	203	14	136	148	155	159	165	171	177	177
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
GDD ^{††} (days)	2230	1286	3294	334	1513	1821	2048	2156	2234	2366	2535	2527
Solar ^{‡‡} (Ly)	966	959	970	2	961	963	964	965	966	967	968	968
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
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
Altitude max. (m a.s.l.)	415	29	1351	298	890	844	586	441	327	162	224	179
Altitude mean (m a.s.l.)	229	8	709	170	542	460	317	255	172	91	120	98
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
Altitude min. (m a.s.l.)	118	0	457	108	360	236	167	129	97	45	49	42
Ruggedness ^{**} (m)	66	6	248	49	112	133	90	70	51	27	42	31
Precipitation (mm)	1091	843	1317	101	1016	1031	1015	1035	1115	1094	1169	1170
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
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
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

[†]Least significant difference between group means at P = 0.01, LSD = 2.58 × √(2 × n⁻¹ × 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 Langley's 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

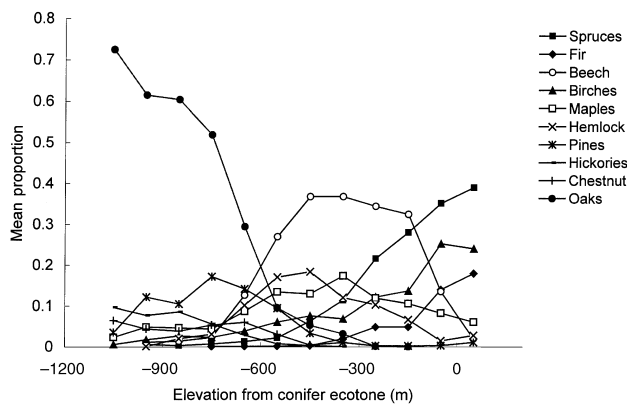


Figure 6 Coenocline plot of town average presettlement composition by classes of FCE-elevation (distance from the conifer ecotone). Mean of town-wide relative frequencies are plotted at the midpoint of 100-m classes.

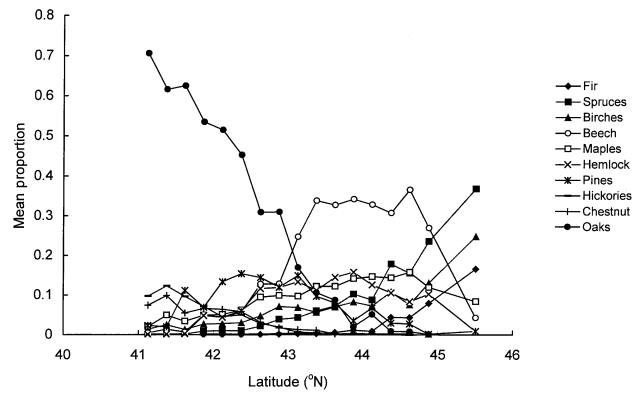


Figure 7 Coenocline plot of town average presettlement composition against classes of latitude. Mean of town-wide relative frequencies are plotted at the midpoint of 0.25 degree classes.

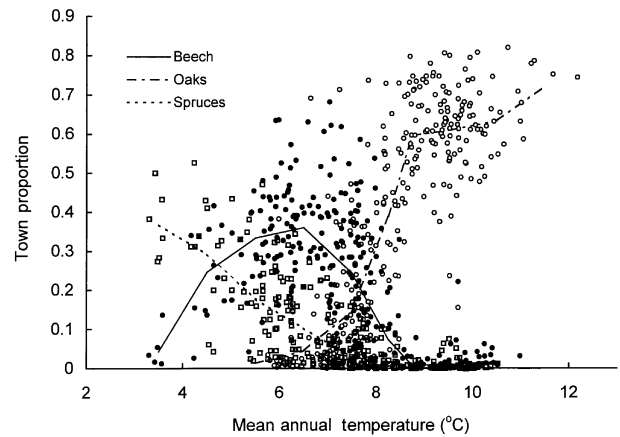


Figure 8 Plot of town-wide (open circle) oak, (solid circle) beech, and (open square) spruce presettlement abundances against mean annual temperature. Lines connect means in 0.5 °C classes plotted at the midpoint.

$r^2 = 0.48$, 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.

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).

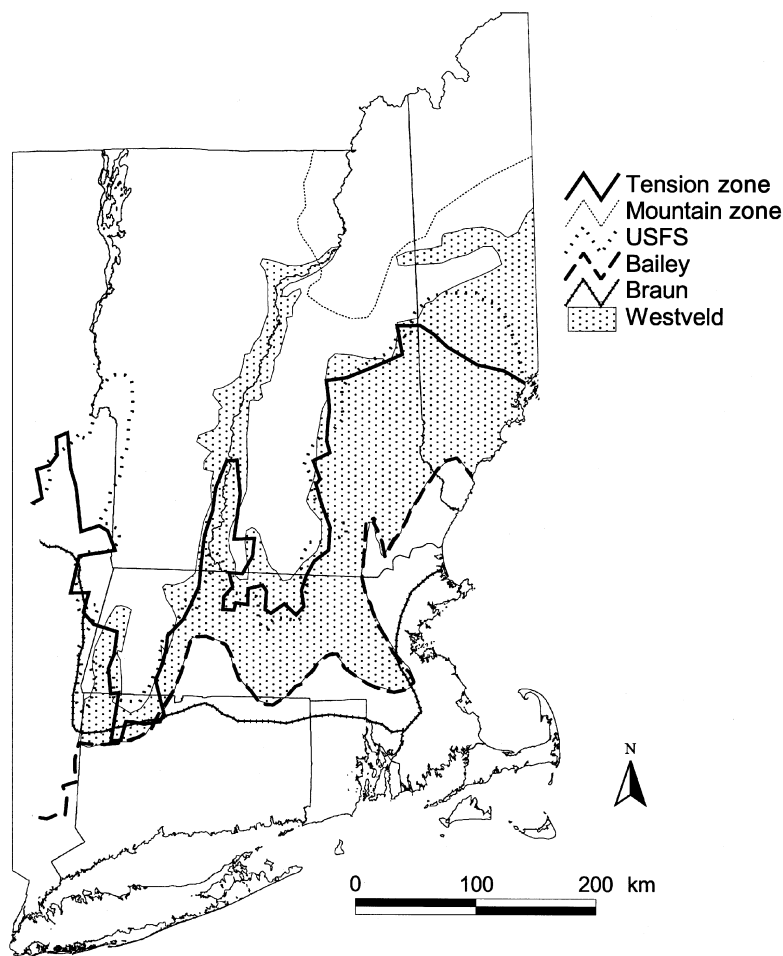


Figure 9 Location of New England Tension Zone, mapped as the town boundaries from presettlement surveys clusters. The shaded area is the extent of Westveld *et al.*'s (1956) Transition-White Pine-Hemlock Zone, which was also adopted by Kuchler (1964, 1978) as a transition zone between Northern Hardwoods 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 ($T_{\text{bar}} = 6.3\text{ }^{\circ}\text{C}$) 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 promi-

ence 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.

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

Appendix I *continued*

Surveyor name	Towns	Stems	Taxa [†] inferred	Genus group this study
Oak, Chestnut	41	174	<i>Quercus prinus</i>	Oaks
Oak, Grey	59	799	<i>Quercus rubra?</i>	Oaks
Oak, Mountain	3	7	<i>Quercus prinus?</i>	Oaks
Oak, Pin	18	28	<i>Quercus palustris</i>	Oaks
Oak, Red	237	7548	<i>Quercus rubra</i>	Oaks
Oak, Rock	37	551	<i>Quercus prinus?</i>	Oaks
Oak, Shrub	9	15	<i>Quercus ilicifolia?</i>	Oaks
Oak, Swamp	46	256	<i>Quercus bicolor</i>	Oaks
Oak, White*	271	32,635	<i>Quercus alba</i>	Oaks
Oak, White Swamp	18	54	<i>Quercus bicolor</i>	Oaks
Oak, Yellow	32	145	<i>Quercus prinus</i>	Oaks
Oilnut	5	16	<i>Juglans cinerea</i>	Butternut
Peach	2	2	<i>Prunus</i> sp.?	Cherries
Pear, Wild	11	23	<i>Prunus</i> sp., [<i>Amelanchier</i> sp.]	Cherries
Pepperidge*	34	114	<i>Nyssa sylvatica</i>	Pepperidge
Pine*	259	9629	<i>Pinus</i> sp.	Pines
Pine, Black	3	4	<i>Pinus</i> sp.?	Pines
Pine, Candle	4	39	<i>Pinus</i> sp.?	Pines
Pine, Norway	16	44	<i>Pinus resinosa</i>	Pines
Pine, Pitch	119	2642	<i>Pinus rigida</i>	Pines
Pine, Red	2	2	<i>Pinus resinosa</i>	Pines
Pine, Spruce	7	13	<i>Tsuga canadensis?</i>	Pines
Pine, Swamp	3	3	<i>Pinus</i> sp.?	Pines
Pine, White	189	2403	<i>Pinus strobus</i>	Pines
Pine, Yellow	15	29	<i>Pinus rigida</i> , [<i>P. resinosa</i>]	Pines
Plum	9	16	<i>Prunus</i> sp., [<i>Amelanchier</i> sp.]	Cherries
Poplar*	194	1188	<i>Populus</i> sp.	Poplars
Poplar, Water	3	22	<i>Populus</i> sp.?	Poplars
Popple	25	86	<i>Populus tremuloides</i> & <i>P. grandidentata</i>	Poplars
Remmond	4	15	<i>Ostrya virginiana?</i>	Ironwoods
Roundwood	4	7	<i>Sorbus</i> sp., [<i>Acer pensylvanicum</i>]	Other
Shagbark	4	7	<i>Carya ovata</i>	Hickories
Spruce*	229	4533	<i>Picea rubens</i> [<i>P. mariana</i> , <i>P. glauca</i>]	Spruces
Spruce, Black	12	77	<i>Picea rubens</i> & [<i>P. mariana</i>]	Spruces
Tamarack*	21	58	<i>Larix laricina</i>	Tamarack
Walnut*	230	7927	<i>Carya</i> sp.?	Hickories
Walnut, Bitter	3	4	<i>Carya cordiformis</i>	Hickories
Walnut, Black	2	2	<i>Juglans nigra</i>	Other
White Tree	5	6	<i>Liriodendron tuliperfera?</i> , [<i>Tilia americana</i>] & [<i>Populus deltoides?</i>]	Other
Whitewood	46	187	<i>Liriodendron tuliperfera?</i> , [<i>Tilia americana</i>] & [<i>Populus deltoides?</i>]	Other
Wicerpee	10	32	<i>Tilia americana?</i>	Basswood
Odd Oaks (<i>Quercus</i> 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; Raue 1, 1			

*Lead taxa in one of the twenty-two lumped genus categories.

[†]Nomenclature follows Gleason & Cronquist (1991).

Exhibit D

Regional carbon dioxide implications of forest bioenergy production

Tara W. Hudiburg^{1*}, Beverly E. Law¹, Christian Wirth² and Sebastiaan Luyssaert³

Strategies for reducing carbon dioxide emissions include substitution of fossil fuel with bioenergy from forests¹, where carbon emitted is expected to be recaptured in the growth of new biomass to achieve zero net emissions², and forest thinning to reduce wildfire emissions³. 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 fuels⁶. 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 removed⁷. The effects of changes in management on CO₂ 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 thinning⁵, with the added benefit of preventing catastrophic wildfire and concurrent carbon loss³. 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 CO₂ 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 (LCA⁶) 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 ecoregions¹¹ 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 forests¹². Forest land ownership is divided fairly evenly between public and private sectors having different management histories and objectives that affect forest carbon dynamics¹³.

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 average¹⁴ and estimates for the member states of the European Union¹⁵. 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

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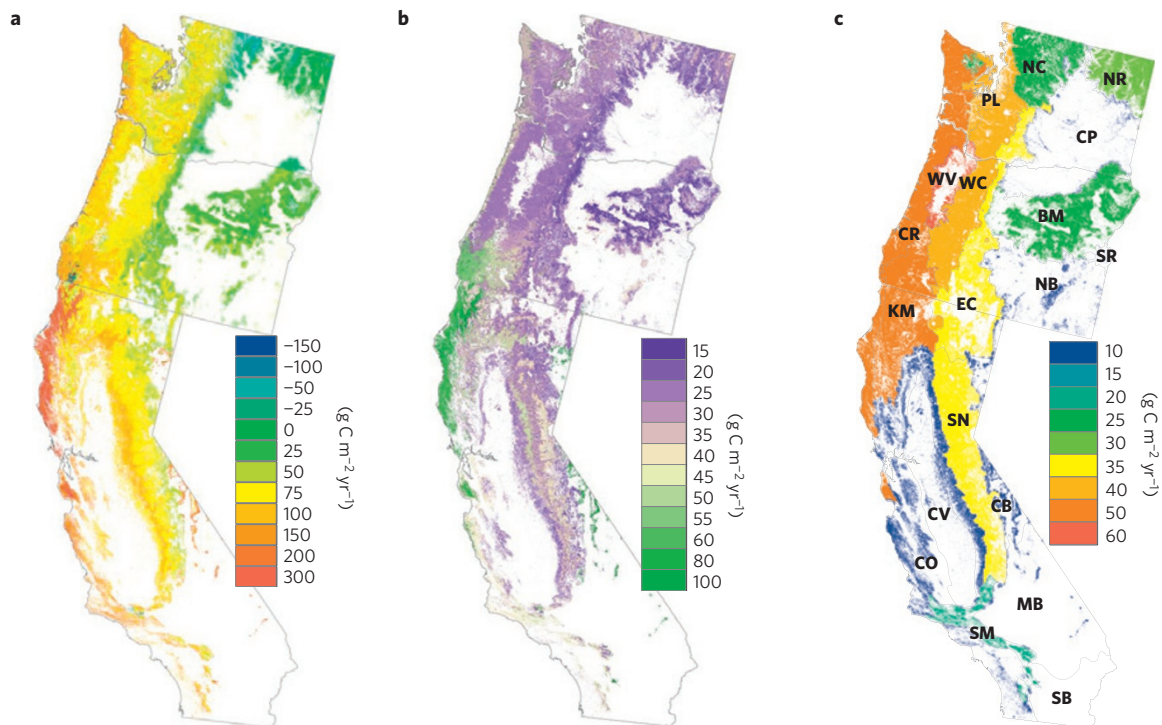


Figure 1 | Maps of US West Coast NBP and uncertainty for current and threshold conditions. a, Current NBP or BAU; positive values (warm colours) indicate forest sinks whereas negative values (cool colours) are carbon sources to the atmosphere. **b**, The current NBP uncertainty estimates that were calculated 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 CO₂ 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 future¹⁹, 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 Northwest²⁰ or increasing the thinning removals of temperate forests is likely to deteriorate the forest and wood

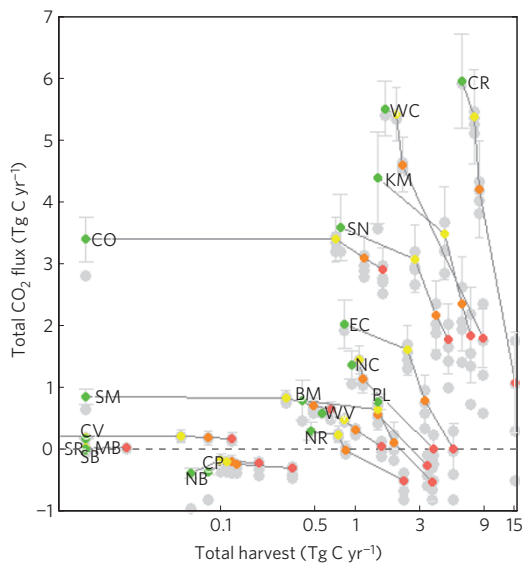


Figure 2 | Life-cycle assessment carbon emission trends by ecoregion under various management scenarios. The x axis is the total harvest (BAU + treatment) and the y axis is the total CO₂ flux in Tg C yr⁻¹ for each ecoregion. Coloured circles represent each scenario (green, BAU; yellow, fire prevention; 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 stock²¹. 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 CO₂ emissions compared to BAU as long as the regional sink persists.

To determine a threshold NBP for which bioenergy management reduces atmospheric CO₂ 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 CO₂ 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 CO₂ to the atmosphere and/or fire emissions are high for BAU, total CO₂ 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 work²². 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 climate²³. 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

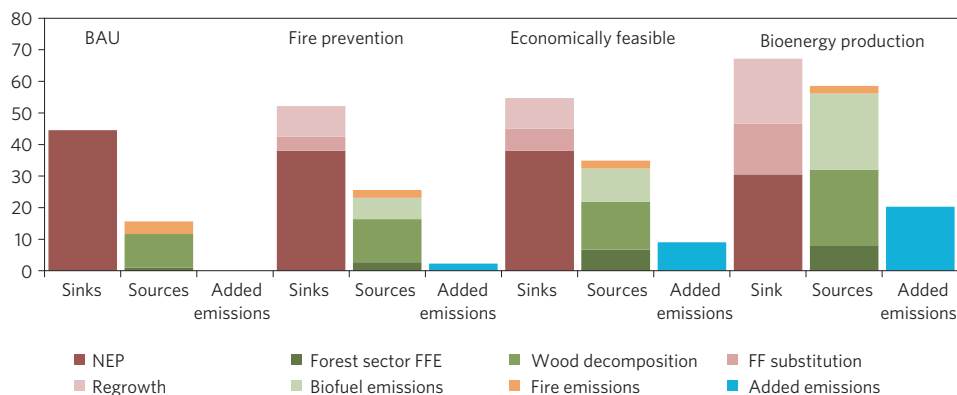


Figure 3 | Total US West Coast forest sector carbon sinks, sources and added emissions relative to BAU under various management scenarios. Units are in Tg C yr⁻¹. Life-cycle assessment estimates account for changes in carbon on land in addition to emissions associated with production, transport and usage 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 CO₂ 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 CO₂-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 risk²⁶, a global data compilation of wood decomposition data and 200 supplementary plots¹³ 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 Northwest²⁸, contrary to other studies that report much higher emissions because they assume 30% of all aboveground woody biomass is consumed²⁷. Although the latter contradicts extensive field observations^{28,29} and modelling studies³⁰ 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 CO₂ 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 CO₂ 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).

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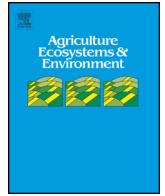
Author contributions

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



Review

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

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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 $0.32 \pm 0.08 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ 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 $16.7 \pm 1.5 \text{ Mg ha}^{-1}$ 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 $0.12 \pm 0.03 \text{ Pg C yr}^{-1}$, 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.

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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 km² 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	150	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	Orthic haplohumod	20	23	Hansen et al., 2000
Denmark	24	626	7.8	Mollic luvisol	13	15	Schjøning 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

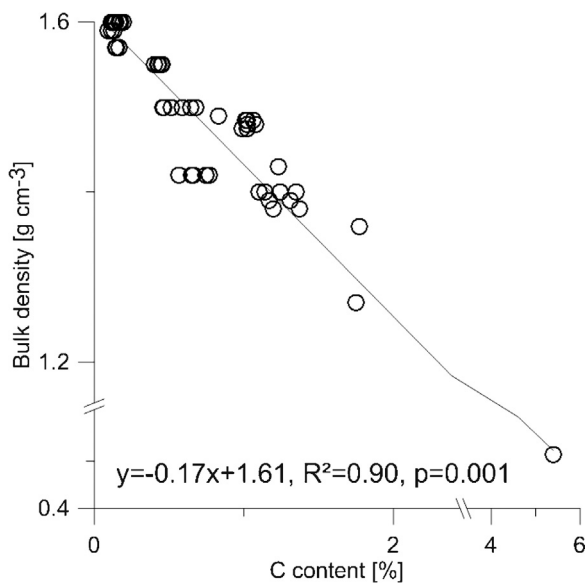


Fig. 1. Correlation of carbon content (%) and bulk density (g cm^{-3}) with regression line and equation for the compiled studies which reported both parameters.

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 $1.87 \pm 0.22 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, 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 ($1.65 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ and $2.09 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, 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 $0.01 \text{ Mg ha}^{-1} \text{ yr}^{-1}$.

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 $0.32 \pm 0.08 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ ($R^2 = 0.17$). 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 $0.35 \text{ Mg ha}^{-1} \text{ yr}^{-1}$). 102 out of 139 observations had an annual change rate between 0 and $1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, while at 13 plots a SOC stock depletion was measured and at 24 plots the change rate exceeded $1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ (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

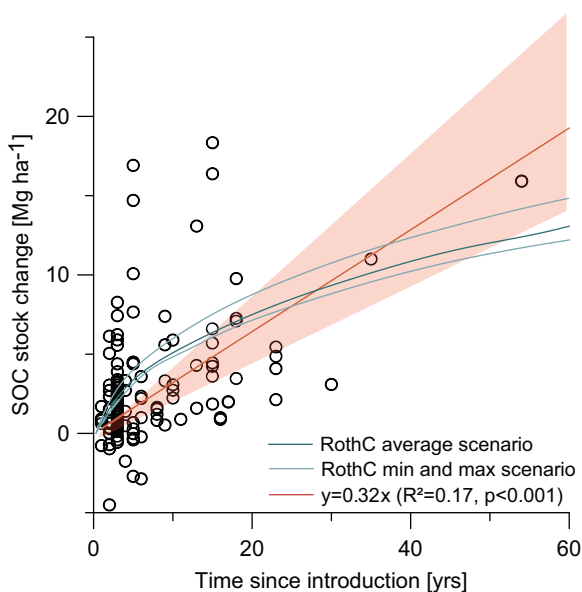


Fig. 2. SOC stock change after cover crop introduction as a function of time with linear regression (with 95% confidence interval) and the RothC simulated average cropland (with min and max scenario).

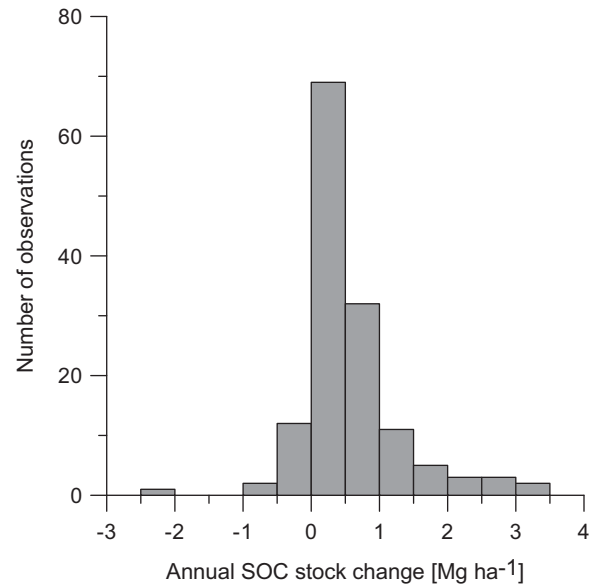


Fig. 3. Histogram of annual change of soil carbon due to cover cropping in comparison to fallow winter.

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:

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

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

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

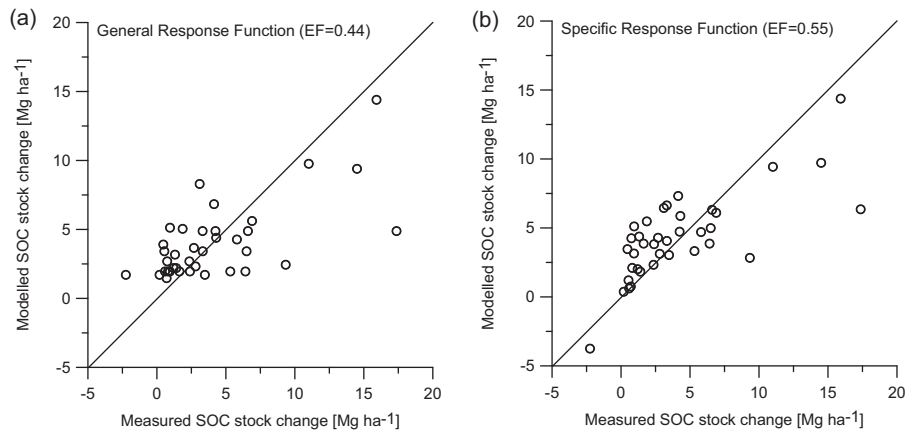


Fig. 4. Modeled vs. measured SOC stock changes for (a) the general “carbon response function” (CRF) applied to site means and (b) the specific CRF applied to site means.

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 $16.7 \pm 1.5 \text{ Mg ha}^{-1}$ and a relative SOC change of $+58 \pm 5\%$. However, half of this new equilibrium (a change of 8.35 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 $0.12 \pm 0.03 \text{ Pg C yr}^{-1}$ (annual change rate for about 50 years) or in total $6.7 \pm 0.6 \text{ Pg C}$ (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

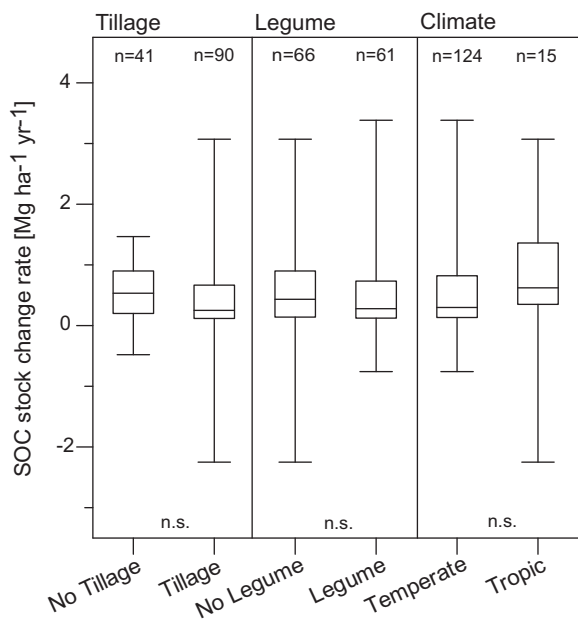


Fig. 5. Boxplots with number of observations for the categorical variables tillage vs. no tillage, plant functional type (no legume vs. legume) and climate zone (temperate vs. tropic).

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 $0.32 \pm 0.08 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ 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).

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 $>2 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. 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 $16.7 \pm 1.5 \text{ Mg ha}^{-1}$ 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 $17 \pm 5 \text{ Mg ha}^{-1}$ in the 0–30 cm soil increment and $18 \pm 9 \text{ Mg ha}^{-1}$ in the 0–80 cm increment after a mean time of 33 years since grassland establishment. Relatively, the estimated SOC stock change of $16.7 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ 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 $0.12 \pm 0.03 \text{ Pg C yr}^{-1}$ ($0.44 \text{ Pg CO}_2 \text{ yr}^{-1}$) 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 $0.32 \pm 0.08 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ to an average maximum increase of 16.7 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.

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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>.

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