2016 BILLION-TON REPORT

Advancing Domestic Resources for a Thriving Bioeconomy

Volume 2: Environmental Sustainability Effects of Select Scenarios from Volume 1

January 2017

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ENERGY

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Advancing Domestic Resources for a Thriving Bioeconomy

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Bioenergy Technologies Office

Volume 2:
Environmental Sustainability Effects of Select Scenarios from Volume 1

January 2017

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Availability

This report, as well as supporting documentation, data, and analysis tools, can be found on the Bioenergy Knowledge Discovery Framework at bioenergykdf.net. Go to https://bioenergykdf.net/billionton2016/vol2reportinfo for the latest report information and metadata.

Additional Information

The U.S. Department of Energy, Office of Energy Efficiency and Renewable Energy's Bioenergy Technologies Office and Oak Ridge National Laboratory provide access to information and publications on biomass availability and other topics. The following websites are available:

- energy.gov
- eere.energy.gov
- bioenergy.energy.gov
- web.ornl.gov/sci/transportation/research/bioenergy/

Front cover images courtesy of ATP[3], Oak Ridge National Laboratory, Abengoa, Solazyme, and BCS, Incorporated.

Preface

On behalf of all the authors and contributors, it is a great privilege to present the *2016 Billion-Ton Report* (*BT16*), *volume 2: Environmental Sustainability Effects of Select Scenarios from volume 1*. This report represents the culmination of several years of collaborative effort among national laboratories, government agencies, academic institutions, and industry. *BT16* was developed to support the U.S. Department of Energy's efforts towards national goals of energy security and associated quality of life.

As director of the U.S. Department of Energy's Bioenergy Technologies Office (BETO), I would like to thank Kristen Johnson, sustainability technology manager who served as one of the leads on this report, Alison Goss Eng, the program manager of Advanced Algal Systems and Feedstocks Supply and Logistics, and Mark Elless, technology manager in the Feedstock Supply and Logistics team for their leadership on crafting this document with the numerous contributors and reviewers. I would especially like to express gratitude to the additional report leads: Rebecca Efroymson, research scientist at Oak Ridge National Laboratory; Matthew Langholtz, research scientist at Oak Ridge National Laboratory; and Bryce Stokes, senior advisor of Allegheny Science and Technology.

This product builds on *BT16 volume 1*, which evaluated the most recent estimates of potential biomass resources that could be available for new industrial uses in the future (up to 2040). Consistent with prior versions of the *Billion-Ton Study, BT16 volume 1* identified potential biomass resources of one billion tons or more per year in the United States. While volume 1 focused on potential resource analysis, volume 2 is a pioneering effort at evaluating changes in land management and environmental indicators associated with select production scenarios derived in volume 1. Addressing a critical knowledge gap, this report uses environmental models to investigate how particular 2017 and 2040 scenarios from volume 1 affect greenhouse gas emissions, soil organic carbon, water quality and quantity, air emissions, and biodiversity. Volume 2 also discusses potential qualitative environmental effects of algae production, and strategies to enhance environmental outcomes.

The results from volume 2 are not meant to be predictions or final answers, but they provide rich quantitative and spatially explicit information revealing potential benefits and challenges that may need to be considered as biomass production increases in the U.S. *BT16 volume 2* will soon be incorporated into BETO's interactive Knowledge Discovery Framework (KDF) at bioenergykdf.net, providing an extensive online resource to inform future R&D as well as efforts to enhance positive effects and reduce potential challenges. Data from the report's rigorous studies will be available to the public, and users can leverage the platform to explore relationships between potential biomass production and potential environmental effects and visualize results to gain new insights. We invite the user community to take a step forward with us and use this report and associated data to perform further analyses, join the vibrant discussion of the latest understanding of environmental indicators and land management, ask more questions, and inform strategies to enhance environmental outcomes of a growing bioeconomy.

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Contributors

Many people contributed to the analyses and reporting in the *2016 Billion-Ton Report (BT16*) volume 2. In addition to completing the analyses, researchers composed report chapters, while also contributing their expertise and rigorous efforts towards enhancing the overall quality and effectiveness of the report. Others contributed technical, managerial, and production skills and knowledge, both to the accuracy and comprehensiveness of the analyses and to the delivery of the information and data in text and electronic formats. The many contributors are listed below by their organizations.

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This report is the product of multidisciplinary collaboration among numerous institutions that contributed to the process—defining assumptions, providing data, conducting analysis, reviewing methods and results, and developing the report. This collaboration ranges from federal agency scientists and national laboratories to universities and contractors. A special thanks and recognition goes to the U.S. Forest Service in the U.S. Department of Agriculture (USDA) for their long-established support and contributions to the Billion-Ton studies and major contributions to this study in particular. USDA has always been a significant collaborator. We also acknowledge the use of USDA data for both agricultural and forestry analyses.

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The authors of chapter 4 on greenhouse gas emissions would like to give special thanks to the study leads, as well as Craig Brandt of ORNL, Jeongwoo Han of Argonne National Laboratory, Michelle Wander of the University of Illinois at Urbana-Champaign, and Ho-Young Kwon of the International Food Policy Research Institute, as well as participants of the peer review workshop who provided written and oral comments on the chapter.

The water footprint analysis (chapter 8) is built upon cumulative efforts developing the Water Analysis Tool for Energy Resources (WATER) model and its components and data inventory development, data dissemination and analysis, and scenario development. The authors of chapter 8 would like to recognize the substantial contributions of Sheshikanth Yalamanchili (Argonne National Laboratory [Argonne]) in the model and database development and in assisting scenario implementation. A special thanks goes to Craig Brandt (ORNL) for intensive data processing and management, Christina Canter (Argonne) for scenario data management, Laurence Eaton (ORNL) for detailing critical assumptions in agriculture scenarios, and Ge Sun (USDA Forest Service) for verifying technical approaches in forestry water analysis. The authors appreciate those who reviewed the chapter on the water consumption footprint. These include Alex Mayer (Michigan Technological University), Tom Richard (Penn State University), Allison Thomson (Field to Market), Marilyn Buford (USDA Forest Service), Steve Kaffka (University of California, Davis), Steve Evett (USDA Agricultural Research Service), and Bob Rose (U.S. Environmental Protection Agency [EPA]).

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The algae chapter authors appreciate the assistance from the Algae Biomass Organization and its leadership, with special thanks to Matt Carr for finding reviewers for this study. The scope of the chapter was discussed with reviewers of *BT16* volume 1. The chapter authors would like to thank all of the peer reviewers, as well as Amanda Barry from the DOE Bioenergy Technologies Office and Colleen Tomaino from BCS, Incorporated. Susan Schoenung (Longitude 122 West, Inc.) and Ryan Davis (NREL) contributed to the methods used to quantify potential biomass and prices in volume 1 and helped clarify assumptions used in this report.

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Reviewers

The workshop titled "Presentation and Expert Review of the *2016 Billion-Ton Report Volume 2*" was held May 11, 2016, in Washington, D.C. Contributors presented an overview of their methods and assumptions, and reviewers provided verbal comments at the workshop and electronic written feedback after the workshop. Full draft chapters were then distributed to reviewers on July 21, 2016, and reviewers, including some who did not attend the workshop, responded with electronic comments. Reviewer comments were addressed during the subsequent revision of the report.

We would like to thank Argonne National Laboratory for hosting the workshop and BCS, Inc., for organizing the workshop. In particular, we thank Joe Ralbovsky, Megan Lucas, Lauren Illing, Ali Minarcik. and Jackson Stubbs.

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Executive Summary

Synopsis

With the goal of understanding environmental effects of a growing bioeconomy, the U.S. Department of Energy (DOE), national laboratories, and U.S. Forest Service research laboratories, together with academic and industry collaborators, undertook a study to estimate environmental effects of potential biomass production scenarios in the United States, with an emphasis on agricultural and forest biomass. Potential effects investigated include changes in soil organic carbon (SOC), greenhouse gas (GHG) emissions, water quality and quantity, air emissions, and biodiversity. Effects of altered land-management regimes were analyzed based on select county-level biomass-production scenarios for 2017 and 2040 taken from the *2016 Billion-Ton Report: Advancing Domestic Resources for a Thriving Bioeconomy* (*BT16*), volume 1, which assumes that the land bases for agricultural and forestry would not change over time. The scenarios reflect constraints on biomass supply (e.g., excluded areas; implementation of management practices; and consideration of food, feed, forage, and fiber demands and exports) that intend to address sustainability concerns. Nonetheless, both beneficial and adverse environmental effects might be expected. To characterize these potential effects, this research sought to estimate where and under what modeled scenarios or conditions positive and negative environmental effects could occur nationwide. The report also includes a discussion of land-use change (LUC) (i.e., land management change) assumptions associated with the scenario transitions (but not including analysis of indirect LUC [ILUC]), analyses of climate sensitivity of feedstock productivity under a set of potential scenarios, and a qualitative environmental effects analysis of algae production under carbon dioxide (CO_2) co-location scenarios. Because *BT16* biomass supplies are simulated independent of a defined end use, most analyses do not include benefits from displacing fossil fuels or other products, with the exception of including a few illustrative cases on potential reductions in GHG emissions and fossil energy consumption associated with using biomass supplies for fuel, power, heat, and chemicals.

Most analyses in volume 2 show potential for a substantial increase in biomass production with minimal or negligible environmental effects under the biomass supply constraints assumed in *BT16*. Although corn ethanol has been shown to achieve GHG emissions improvements over fossil fuels, cellulosic biomass shows further improvements in certain environmental indicators covered in this report. The harvest of agricultural and forestry residues generally shows the smallest contributions to changes in certain environmental indicators investigated. The scenarios show national-level net SOC gains. When expanding the system boundary in illustrative cases that consider biomass end use, reductions in GHG emissions are estimated for scenarios in which biomass—rather than oil, coal, and natural gas—is used to produce fuel, power, heat, and chemicals. Analyses of water quality reveal that there could be tradeoffs between biomass productivity and some water quality indicators, but better outcomes for both biomass productivity and water quality can be achieved with selected conservation practices. Biodiversity analyses show possible habitat benefits to some species, with other species showing potential adverse effects that may require additional safeguards. Increasing productivity of algae can reduce GHG emissions and water consumption associated with producing algal biomass, though the effects of water consumption are likely of greater concern in some regions than in others. Moreover, the effects of climate change on potential biomass production show gains and losses in yield among feedstocks across the continental United States. Key research gaps and priorities include actions that can enhance benefits and reduce potential for negative effects of increased biomass production. The results from this report will help DOE, the bioenergy industry, and other institutions continue important discussions on environmental effects and will help chart a path toward a more environmentally sustainable bioeconomy.

Introduction

For more than a decade, DOE has been quantifying the potential of U.S. biomass resources for production of renewable energy and bioproducts. *BT16* volume 1 (released in July 2016) estimates potential biomass that could be available for use in the future at specified prices, assuming a future market for the biomass. Volume 2 (this volume) is a first effort to analyze a range of potential environmental effects associated with illustrative near-term and long-term biomass-production scenarios from volume 1. Environmental effects of biomass production, including effects on SOC, GHG emissions, water quality, water quantity, air emissions, and biodiversity, are modeled. Land management changes associated with the scenario transitions are described and discussed, but modeling ILUC is outside the scope of this report.

As estimated in *BT16* volume 1, 0.8 billion dry tons or 1.2 billion dry tons of biomass are potentially available annually by 2040 at $60 per dry ton or less,[1] under base-case and high-yield production scenarios,[2] respectively, In addition, an estimated 365 million dry tons of currently used resources were used in 2015 (e.g., corn for ethanol, wood waste) and are assumed to remain constant through the simulation period to 2040 (see table ES.1 in *BT16* volume 1). These potential and current supplies include forestry, agricultural, and waste resources. *BT16* volume 2 focuses primarily on the largest categories of these total potential supplies, i.e., agricultural and forest biomass (see descriptions of feedstock types below). Although energy crops are scarce in the near term, they represent the greatest source of potential biomass in future scenarios.

BT16 assumptions hold total forestland and total agriculture lands constant throughout the 2017–2040 simulation period. The primary type of LUC implied in *BT16* supply scenarios involves land management within agricultural land. When total land allocation in 2015 (agricultural baseline) is compared to land allocation in 2040 under biomass scenarios, 24 or 45 million acres (net) transition from annual crops to perennial crops under the *BT16* base case or high-yield scenarios, respectively. An additional 37 to 39 million acres of agricultural land transitions from pasture to perennial energy crops (about 8% of total pasture area in the 2015 agricultural baseline).

The potential biomass supplies in *BT16* volume 1 reflect guiding principles for environmental and socioeconomic considerations. These principles are consistent with DOE's mission to develop biomass as a sustainable resource and with other research that applies environmental constraints to resource analysis (Schubert et al. 2009; Beringer, Lucht, and Schaphoff 2011). For example, simulations in *BT16* volume 1 aim to promote food security and incorporate projected future demands for food, feed, forage, and fiber in the simulations from 2017 through 2040. Constraints are embedded in the scenario assumptions to minimize land-use transitions of highest concern (e.g., the loss of forestlands or productive cropland). Land management constraints that promote environmental quality, such as reduced tillage and residue-retention practices, minimal irrigation (see chapter 2), and reserved land areas to protect biodiversity and soil quality, are assumed in the biomass supply scenarios (see chapter 1). The use of these constraints effectively reduces potential adverse environmental effects and the potential biomass supply itself, compared to biomass that could be available otherwise.

The guiding principles and supply constraints embedded in volume 1 illustrate biomass production opportunities that could minimize or avoid key environmental concerns. However, it is important to further investigate the potential environmental implications of land management changes portrayed in volume 1. This knowledge gap is the motivation behind *BT16* volume 2.

[1] This price is at farmgate or roadside, marginal cost. In GHG emissions analyses and air emissions analyses, supplies delivered to the biorefinery (up to a price of $100 per dry ton at the reactor throat) are included.

[2] Scenarios are specific to *BT16* as described under "Scenarios and Data Inputs" and further elaborated in chapter 2.

Goals of Volume 2

In addition to investigating potential environmental effects associated with select biomass production scenarios in volume 1, *BT16* volume 2 also seeks (1) to advance the discussion and understanding of environmental effects that could result from significant increases in U.S. biomass production and (2) to accelerate progress toward a sustainable bioeconomy by identifying actions and research that could enhance the environmental benefits while minimizing negative impacts of biomass production.

Scenarios from 2017 and 2040 were selected to examine effects of a large increase in biomass production with an emphasis on cellulosic biomass in the future, as well as effects of increasing biomass yield. Key environmental indicators were modeled in the categories of SOC, GHG emissions, water quality, water quantity, air emissions, and biodiversity (see section 1.3). Most results are presented at the county level. Results primarily focus on cellulosic biomass, although some analyses include corn grain to estimate how future cellulosic biomass might compare to conventional biomass production. This volume also presents a qualitative analysis of environmental effects of algae production under a set of scenarios from volume 1 in which algae production is co-located with sources of waste CO_2. An analysis of climate sensitivity of agricultural feedstock productivity under a set of potential future scenarios is also included.

BT16 volume 2 provides a spatially explicit illustration of potential biomass production opportunities and associated environmental implications, rather than a prediction of biomass production and environmental effects that will inevitably occur. It is important to note that the biomass supply estimates presented in *BT16* are policy independent and based on specified price and yield scenarios that assume a market demand. This report differs from efforts that seek to depict potential biomass demand and related market, environmental, and land-use interactions under specifically defined business-as-usual or policy conditions. Assumptions used in *BT16* regarding land transitions and supply constraints have implications for the environmental effects analyses, and modifying these assumptions would likely result in different environmental effects.

Scenarios and Data Inputs

A small subset of the agricultural and forestry assessment scenarios and scenario years from *BT16* volume 1 was selected for analysis in *BT16* volume 2. The scenarios in volume 2 include a low- and a high-yield scenario and near-term and long-term estimates from volume 1. "Yield" refers to annual improvements in crop yield for commodity crops and energy crops. The $60 per-dry-ton price model runs of the base-case[4] (i.e., 1% annual yield increase, referred to as "BC1" in *BT16* volume 1) and high-yield (i.e., 3% annual yield increase, referred to as "HH3") scenarios were chosen from the agricultural assessment in volume 1. From the forestry assessment, the baseline (moderate housing, low wood energy demand, referred to as "ML") and high housing–high wood energy ("HH") scenarios were selected.[5]

Most chapters in volume 2 analyze county-level outputs from the following volume 1 biomass scenar-

[4] The terms base case and baseline have specific meanings in *BT16* that may differ from the use of these terms in other studies.

[5] In the forestry assessment, biomass availability decreases from 2017 to 2040. Furthermore, biomass is lower in the HH 2040 scenario than the ML 2040 scenario because of the high demand assumed for housing.

ios, all assuming a roadside price of up to $60 per dry ton[6] (table ES.1; fig. ES.1; and table ES.2):

1. **BC1&ML 2017**: 2017 base-case agricultural combined with baseline forestry scenarios: 326 million dry tons
2. **BC1&ML 2040**: 2040 base-case agricultural combined with baseline forestry scenarios: 807 million dry tons
3. **HH3&HH 2040**: 2040 3% high-yield agricultural combined with HH forestry scenarios: 1.1 billion dry tons.

Many chapters analyze agricultural biomass only or forestry biomass only. Although the use of wastes for energy has potential environmental benefits, quantifying these effects is beyond the scope of this analysis. These effects are considered qualitatively in the final chapter of this report.

Table ES.1 | Biomass Supplies Identified in *BT16* volume 1 and Evaluated in volume 2 for Select Scenarios and Years (in Million Dry Tons)

	Identified in volume 1			Evaluated in volume 2		
Scenario	BC1&ML 2017	BC1&ML 2040	HH3&HH 2040	BC1&ML 2017	BC1&ML 2040	HH3&HH 2040
New potential	343	826	1,154	192	669	997
Currently used	365	365	365	134	138	139
Total	709	1,192	1,520	326	807	1,136
Notes	New potential and currently used resources include agricultural and forest biomass and waste resources.			New potential includes agricultural and forest biomass only. Currently used resources include only corn ethanol and soybean biodiesel portions. Waste resources are excluded.		

[6] GHG and air emission analyses are limited to supplies at $100 or less delivered to the biorefinery.

Figure ES.1 | Biomass resources of the three primary scenarios evaluated in this volume[7]

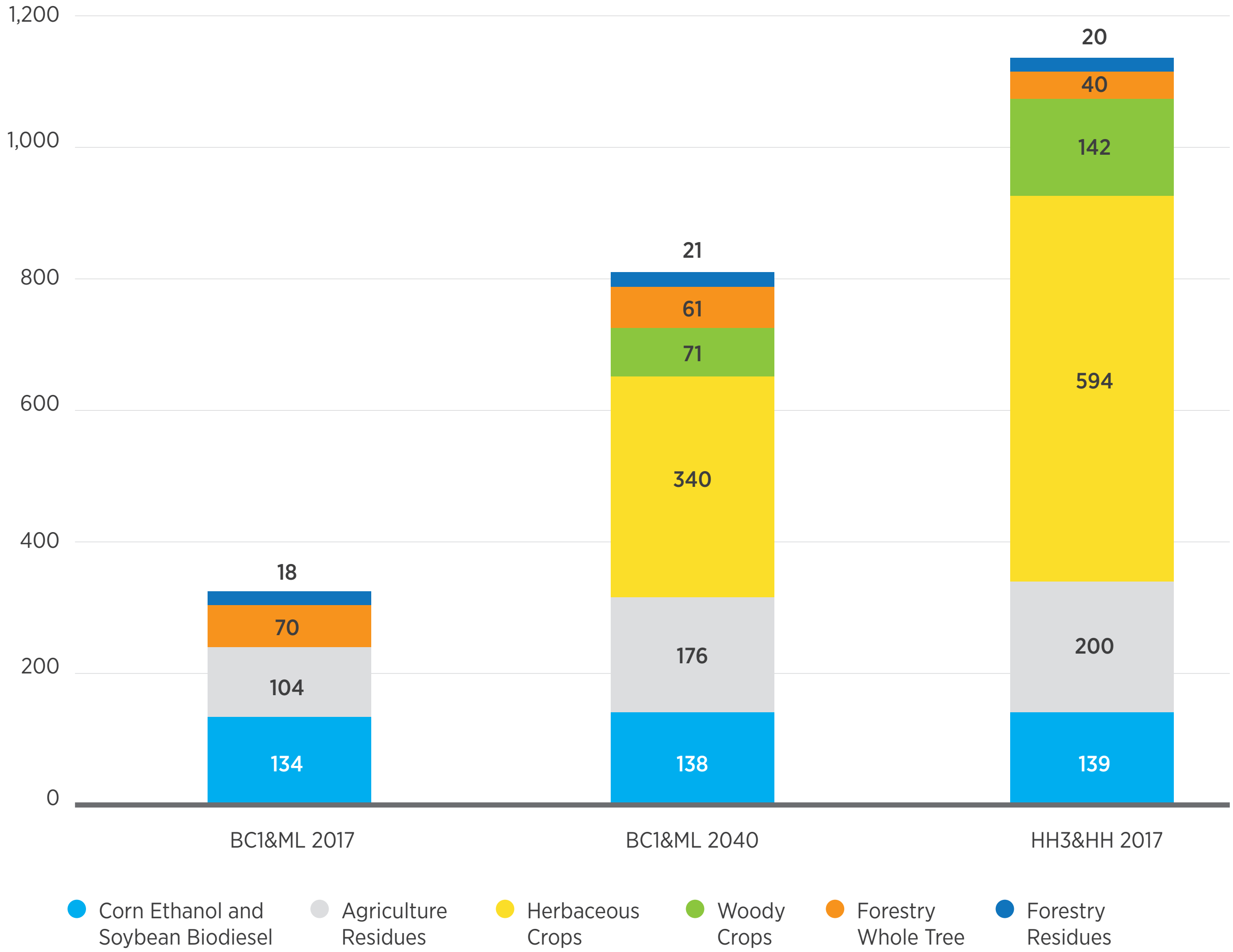


Table ES.2 describes the agricultural and forestry scenarios; chapter 2 provides more details on these scenarios and a brief summary of the methodology used to generate data in volume 1 that are analyzed in volume 2.

[7] The supplies analyzed in volume 2 exclude about 230 million dry tons of currently used resources (current uses beyond corn ethanol and soybean biodiesel) and about 140 million dry tons of additional waste resource potential reported in volume 1. In the forestry assessment, biomass availability decreases from 2017 to 2040. Furthermore, biomass is lower in the HH 2040 scenario than the ML 2040 scenario because of the high demand assumed for housing.

Table ES.2 | Scenarios Considered in *BT16* Volume 2 Analyses

Combined agricultural and forestry scenarios		Agricultural scenarios			Forestry scenarios			
Combined identifier	Year	Identifier	Energy crop annual yield increase[a]	Corn annual yield increase	Identifier	Description	Housing starts	Wood energy demand
BC1&ML 2017	*2017*	BC1 (base-case yield)	1%	0.8%	ML (baseline)	Moderate housing–low wood energy	Returns to long-term average by 2025	Increases by 26% by 2040
BC1&ML 2040	*2040*	BC1 (base-case yield)	1%	0.8%	ML (baseline)	Moderate housing–low wood energy	Returns to long-term average by 2025	Increases by 26% by 2040
HH3&HH 2040	*2040*	HH3 (high yield)	3%	1.9%	HH (high demand)	High housing–high wood energy	Adds 10% to baseline in 2025 and beyond	Increases by 150% by 2040

[a] Yield improvements are only applied at establishment and are not applied after year one for perennial crops until replanting

The following is a summary of results from chapters 3 through 13 in this report.

Land Allocation and Management

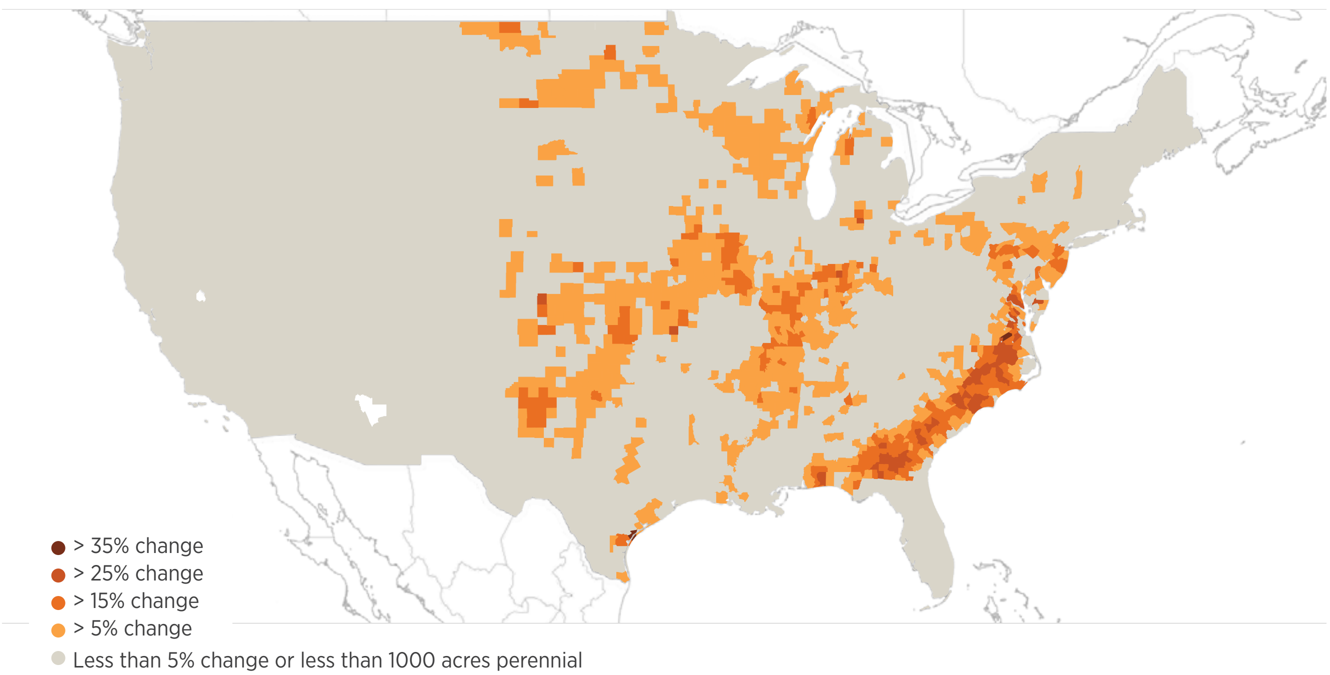
Chapter 3 of *BT16* volume 2 aims to clarify LUC implications of the select *BT16* scenarios. Unlike most LUC studies, volume 2 does not analyze the LUC effects of a policy. *BT16* assumptions hold the forestland and agricultural land base constant throughout the 2017–2040 simulation periods. Supply constraints limit the total land available for energy crops in *BT16* based on rainfall, rates of transition, and caps on total area allowed to transition to new crops (see chapter 2).

The primary type of LUC associated with *BT16* supply scenarios involves changes in agricultural land management practices. For example, the area that would be managed as perennial cover in 2040 is 24 and 45 million acres greater under BC1 and HH3 (respectively) than the area of perennial cover in the U.S. Department of Agriculture (USDA) 2015 agricultural baseline. Additional changes in management occur on pasture: 37–39 million acres, or about 8% of total pasture area in the 2015 agricultural baseline, would undergo changes in management for energy crops by 2040. Fencing and pasture rotation are management practices that are assumed to intensify production on another 60 million acres of pasture.

The geospatial distribution of the net change from annual to perennial cover in BC1 is illustrated in figure ES.2. By 2040, changes in land management affect about 3% of total cropland (e.g., transition from annual to perennial cover) and 19% of total pastureland, with 11% being intensified and 8% being managed for energy crops (percentages here are relative to the total areas of cropland and pastureland in the 2015 agricultural baseline). As with any model, input parameters and assumptions regarding land classes, land area available for different uses, and productivity influence how land is allocated among traditional and energy crops over time.

Chapter 3 includes a review of LUC studies and concludes that clear definitions of land parameters and effects are essential to improve LUC analyses. The large variability in results from previous LUC analyses associated with increased biomass production underscores the need for more consistent and transparent approaches.

Figure ES.2 | Geospatial distribution of changes in perennial cover under the base-case (BC1) scenario



Change in perennial cover by county is the difference between the percentage of total agricultural acres (cropland +pasture +idle land) managed as perennial cover in the 2040 base case (BC1) and the percentage managed as perennial cover in the 2015 agricultural baseline. The maximum county-level increase in perennial cover in BC1 was 38%. The light grey shading over the majority of counties indicates that change was below 5% (either an increase or decrease in perennial cover). Larger increases in percentage of perennial cover occur on agriculture land in the Southeastern Plains and in areas where simulated returns from conventional crops are not as competitive with energy crops under the conditions defined in the base-case scenario.

Greenhouse Gas Emissions, Soil Carbon, and Fossil Energy Consumption

The GHG emissions and fossil energy consumption associated with producing potential biomass supply in the select *BT16* scenarios include emissions and energy consumption from biomass production, harvest/collection, transport, and pre-processing activities to the reactor throat. Emissions associated with energy, fertilizers, and agricultural chemicals that are consumed in biomass production are also included. Energy consumption and emissions for biomass logistics are considered only for biomass with delivered costs below $100 per dry ton. The contribution of changes in SOC to GHG emissions as a result of producing agricultural biomass is also considered. Changes in forestry soil carbon are not analyzed because the land area in forestry stayed constant and no major forestry land management changes were considered. However, a review of potential impacts of using forest biomass as a bioenergy feedstock on soil carbon is discussed. This analysis indicates potential GHG-emissions hotspots from producing biomass and illuminates drivers for these emissions, which can inform efforts to reduce the GHG emissions and energy consumption of biomass-derived fuels, products, and power.

Generally, results show that conventional crops would have a higher share of GHG emissions per ton than energy crops, and the GHG intensity (emissions per mass) of biomass production would be lower in higher-yield scenarios (e.g., HH3 and HH 2040). Emissions from the production of forestry biomass would be, in general, lower than for other crops because not all forestry plots undergo site preparation, which consumes diesel fuel, and because fertilizers are used more sparingly than for agricultural crops. Overall, forest residues would be a minor contributor to both biomass tonnage and GHG emissions in these scenarios. Other factors besides yield that influence GHG-emissions intensity include advanced logistics operations and SOC changes. The latter factor varies in importance by region, yield, and by final and initial land allocations. In general, growing energy crops on historical cropland typically leads to SOC gains. When pasture is used to produce biomass, however, only a few energy crops sequester soil carbon. This analysis found that under the two *BT16* 2040 scenarios, changes in SOC could result in a net soil carbon sink nationally, largely due to the land transition to energy crops (particularly miscanthus).

It is important to note that *BT16* is not a life-cycle analysis of fuels, products, or power produced from the biomass. However, a few illustrative case studies were completed to estimate displacement of fossil-derived GHG emissions and energy. Life-cycle GHG intensities for both biomass- and fossil fuel–derived fuel and energy products were applied to specific scenarios based on potential growth in energy, power, and chemical production between now and 2030. These cases illustrate that GHG-emissions reductions (between 4%–9%) and fossil energy consumption reductions could be expected as compared to a scenario in which all U.S. energy and conventional products are produced from fossil fuels in that year. Results depend on these GHG intensities, the biomass supply, and how the biomass supply is allocated to different end uses.

Water Quality (Agriculture)

A water-quality analysis addressed the question: how can future biomass production be managed to protect water quality with minimal decreases to feedstock yield? Two tributary basins of the Mississippi River that have contrasting future biomass-feedstock profiles under the BC1 2040 scenario were selected for analysis. The Iowa River Basin (IRB) supports corn-soy-dominated agriculture with corn stover as the dominant potential cellulosic feedstock. The Arkansas-White-Red (AWR) River Basin grows a broader diversity of cellulosic feedstocks including perennial grasses in the 2040 scenario; sorghum; and residues from wheat, corn, and grain sorghum. This analysis found that suitable combinations of conservation practices improved water quality with relatively small decreases in feedstock yield in both river basins. Results for the IRB suggest that four practices (i.e., riparian buffer, cover crop, slow-release nitrogen fertilizer, and tile-drain control), if additive, could reduce nitrogen loading by more than 65% for watersheds planted in corn. In the AWR River Basin, higher fertilizer levels produced higher yields of perennial grasses and short-rotation woody crops (SRWCs), higher nitrate loading, and lower levels of sediment and phosphorus draining into this basin. Thus, the challenge is to balance the other three indicators (i.e., productivity, sediment, and phosphorus) against nitrate. In addition, the results reflected a water-quality benefit of coppiced willow, which minimized trade-off between nutrient and sediment reduction and biomass yield. Filter strips also provided water-quality benefits from SRWCs. Results from this analysis can be used to identify location-specific management practices that can achieve simultaneous biomass production and water-quality goals.

Water Quality (Forestry)

Despite decades of research into forest harvest effects on water quality, longterm and consistently collected data to parameterize process-based models of water-quality related to biomass removal in forests are scarce. Therefore, this analysis developed a simple, empirical modeling approach to estimate sediment and nutrient response to the total acres harvested for biomass within a given county. Results were aggregated to three regions of the United States: the South, West, and North (see chapter 6, fig. 6.1, for regional divisions). Modeled estimates show there could be regional variation in how biomass harvest would influence water quality. Sediment loads often increase after intensive site preparation in plantations. Because these practices are most common in the South, results indicate that absolute sediment loads and percent increases over reference conditions could be greatest in the South, with smaller increases in the West and North. Alternatively, results indicate that absolute nitrate loads could increase most in the North; however, when considered as an increase over regional reference, the highest increase occurs in the South, followed by the North and then the West in ML 2017. In the ML 2040 and HH 2040 scenarios, the largest percent increase is still in the South, but the North is surpassed by the West. For the scenarios investigated, sediment flux is the most dynamic water-quality parameter, as it could increase nearly 40% or more after biomass harvests, particularly in areas where mechanical site preparation is common prior to planting. Responses for nitrate and total phosphorus tend to be less dynamic, with high-yield scenarios typically resulting in <10% increase over baseline loadings. Continued adherence to and increased adoption of best management practices on lands on which silviculture is practiced should minimize biomass-harvest impacts.

Water Quantity (Forestry)

The amount and distribution of live forest biomass is closely related to water yield (outflow from a drainage basin) and water supply. Biomass harvesting has the potential to alter water quantity indicators by altering the ecohydrological processes (evapotranspiration in the ecosystem in particular). This analysis investigated how prescribed forest-harvesting scenarios affect mean seasonal and annual water yield at the county level. The three scenarios modeled all have minor impacts on water quantity at the county level, with water-yield responses increasing 0.3% or less, largely because of the small areas of harvesting (<5%) in most counties. The small magnitude of hydrological response to biomass removal may not have much significance, positive or negative, in terms of water supply at the county level; however, concentrated biomass-removal activities may cause substantial local impacts on watershed hydrology, such as increasing stormflow volume and potentially causing water-quality concerns. County-level estimates of biomass harvesting do not provide the spatial information sufficient for watershed-scale assessment. However, this assessment identifies regions that are most likely to experience hydrological impacts under the scenarios investigated. Future watershed-scale studies should focus on these regions. Also, other ecologically relevant hydrologic parameters, such as base flow and peak flow rates, should be examined in addition to annual water yield.

Water Consumption Footprint (Agriculture and Forestry)

BT16 volume 1 showed the potential for increasing biomass production without reliance on irrigation. This water footprint analysis investigated water-resource demand for the three select *BT16* scenarios (agricultural combined with forestry scenarios) by estimating the water footprint and conducting geospatial analyses to examine the interplay between feedstock mix and water consumption at three scales: county, state, and national. Biomass requires water from irrigation or rainfall, and some deep-rooted energy crops, such as perennial grasses and SRWCs, as well as forest biomass, can grow without irrigation, which was the assumption in *BT16* volume 1. The water footprint analysis illustrated greater rainfall use on a volume basis for both agricultural and forest biomass in 2040 scenarios, compared to 2017, with more biomass produced and harvested in the 2040 scenarios. Lower consumption of irrigation water is associated with the water footprint of 2040 scenarios compared to 2017. Irrigation for corn was attributed to the grain rather than the residues. Overall, water consumption to produce a ton of biomass remains unchanged in the scenarios. Although both rain water and irrigation water are consumed, rain water is generally preferred because of its low cost, especially in the water-rich regions. Additional research is needed to place water consumption findings in the context of regional water needs.

Air Pollutant Emissions (Agriculture and Forestry)

This analysis developed county-level emission inventories for seven non-GHG, regulated air pollutants[8] for the three biomass supply scenarios (agriculture combined with forestry). These inventories consider emissions from field preparation through harvest, including chemical application and on-farm (or on-forest) transportation, along with transportation and preprocessing for a selected portion of feedstock to the biorefinery. Upstream air emissions (e.g., emissions associated with fertilizer production) and air emissions avoided by displacing other products or fuels with biomass-derived products or fuels were beyond the scope of this study. However, emissions reductions from displacement or upstream emissions may be substantial and should be the focus of future study.

The results indicate that although the air pollutant emissions per dry ton of feedstock produced would vary by county and pollutant, they are generally lower for cellulosic feedstocks than for corn grain. However, this study also shows that the emissions resulting from increased biomass feedstock produc-

[8] The seven pollutants include ammonia, nitrogen oxides, volatile organic compounds, particulate matter ($PM_{2.5}$, PM_{10}), carbon monoxide, and sulfur oxides.

tion could pose challenges for local compliance with air-quality regulations. The variability in county-level emission estimates suggests that certain practices and production locations result in much lower emissions than others. Higher yields, lower tillage requirements, and lower fertilizer and chemical inputs are important factors that contribute to lower air emissions. In addition, using biomass more locally or using more fuel-efficient long-distance transportation methods (e.g., rail or densified biomass) could potentially decrease emissions from truck transport.

For the *BT16* scenarios analyzed, about a quarter of the counties are estimated to emit direct and precursor criteria pollutant mass emissions around 1% to 10% of the current National Emissions Inventory (NEI) (see chapter 9). Emissions in areas currently in attainment could pose challenges in the future or for surrounding areas. In areas currently in non-attainment for the Clean Air Act's National Ambient Air Quality Standards (NAAQS), the absolute increase in mass emissions under *BT16* scenarios is estimated to be small (a few percentage points of the current NEI baseline emissions; see chapter 9) relative to current attainment counties. Emissions in non-attainment counties are more likely to pose challenges to meeting the Clean Air Act's NAAQS in the context of population and economic growth.

The emission estimates provided in this study could be coupled with air-quality screening tools to evaluate changes in emission concentrations, to assess human health impacts, and to help inform future air-quality planning.

Biodiversity (Agriculture)

Bird species habitat and species richness in agricultural landscapes were modeled as a way to investigate questions about potential effects to biodiversity resulting from increased energy crop production. The approach used species-distribution modeling to model bird probabilities of occurrence in different geographic locations as a function of climate and land use/land cover. For the majority of counties, grassland, forest, and generalist birds showed no change in occupancy under the base-case scenario (BC1) in 2040. For the other counties, nearly equal percentages of species were estimated to occupy fewer and more counties. However, decreases in richness were larger than increases for forest and generalist species. The analysis showed that grassland birds would respond positively to switchgrass in comparison to row crops, but the responses to miscanthus in the United States are less well understood. Because many species are affected by the type and timing of management activities, as well as by land cover, guidelines for managing bioenergy crops may be needed to maintain biodiversity of grassland birds and other species as biomass production increases. This analysis is useful in showing where energy crops could be grown with potential benefits to bird species and where more research is needed to understand the wildlife consequences of adopting particular energy crops and management practices.

Biodiversity (Forestry)

Using harvest acres generated in volume 1 of *BT16*, this analysis assesses and compares implications for biodiversity of potential forest biomass produced in the near term (2017) and long term (2040). A coarse-filter approach was taken to assess effects of woody-biomass harvesting on biodiversity. Woody-biomass harvest in the examined scenarios would primarily affect biodiversity through changes in forest structure, both at the stand (e.g., loss of canopy cover and residues) and landscape scales (e.g., distribution of stand ages from clearcutting smaller-diameter trees). Species could be negatively or positively affected at the ecoregion scale based on the primary forest-habitat type sourcing the feedstock, and at the local scale based on species distributions, specific habitat requirements, and the proportion of forest types affected by biomass harvest. Case studies of taxonomic groups or single species with life-history traits that rely functionally on dead and downed wood or changing canopy cover are discussed. This information may be used in conjunction with other finer-scaler biodiversity assessments (e.g., state wildlife action plans, county project planning, etc.) to identify species that may be vulnerable to changes. Conservation of species amidst an increasing national demand for woody biomass will require taking a multi-scale planning approach and continued monitoring of species that are functionally dependent on the material to fulfill their life-history requirements.

Qualitative Analysis of Environmental Effects of Algae Production

The environmental effects analysis for algae emphasizes scenarios from volume 1 of *BT16*, wherein open-pond biomass-production facilities are co-located with coal-fired power plants, natural gas power plants, or ethanol-production plants to reduce cost and to use waste CO_2 that would otherwise be emitted directly into the air. GHG emissions and water-quality indicators are emphasized, though other indicators are discussed. Variables include freshwater and saltwater strains, current and future high-productivity scenarios, and fully and minimally lined ponds. Few examples of commercial algae production exist, and few environmental indicators have been measured for systems resembling those that were modeled. However, some qualitative results are clear: (1) increasing productivity has benefits for water consumption on a per-mass basis; (2) GHG emissions are generated from plastic liner production and piping CO_2 in flue gas to production facilities, so minimizing that infrastructure can minimize GHG emissions; and (3) water consumption can be reduced through the use of sealed systems or recycling, but the broader significance of doing so depends on the regional context, including weather and climate, competing water uses, type of water used, and requirements of regional biota. Enclosed photobioreactors would have different environmental effects, such as lower water consumption because of very low evaporation, but these were not examined in *BT16* volume 1.

Climate Sensitivity of Feedstock Productivity

The modeling of potential biomass feedstock responses to alternative climate change scenarios indicates that, much like conventional crops and other vegetation, biomass feedstocks are sensitive to climatic conditions. The U.S. climate is projected to change significantly in coming decades, particularly for regions such as the Midwest and Southeast, which are considered priority landscapes for the development of biomass resources. Projections of biomass-yield responses to climate change scenarios indicate that the expected warmer climate could alter yields and shift the geographic distribution of commercially important feedstocks (e.g., sugarcane could be grown in more northerly latitudes than is done currently). Projections show that both significant increases and decreases in feedstock yields could occur in future decades, given the current genetic composition of feedstocks, the levels of technology and management associated with feedstock production, and the biomass supply chain. These changes may have greater significance at the regional level than at the national level. Variability in feedstock response is a function not only of geographic variability in current climate and future climate change, but also variability in the inherent sensitivity of different feedstocks and cultivars to particular changes in climate. The development of a more process-based understanding of biomass feedstock responses to changing climatic conditions that includes factors such as climate variability and extremes, the effects of CO_2 fertilization, and different management practices and economic constraints would assist in reducing uncertainties associated with purely empirical methods.

Synthesis and Conclusions

BT16 volume 1 demonstrates the technical and economic potential for increasing national biomass production to support a thriving U.S. bioeconomy. Volume 2 of *BT16* is a first effort to quantify potential environmental effects associated with illustrative near-term and long-term biomass-production scenarios from *BT16* volume 1. Taken together, this collection of analyses reveals benefits, opportunities, challenges, and tradeoffs that should be considered as biomass production increases.

The results must be interpreted in light of uncertainties. As with *BT16* volume 1, results presented in *BT16* volume 2 are neither predictions nor final answers, and they pertain only to the select scenarios. For example, the scenarios reflect the assumption that the agricultural land base and the forest land base do not change between the present and 2040. This assumption has implications for all of the environmental effects analyses, and modifying scenarios to allow transitions between these major land classes could result in environmental changes of different types, magnitudes, or direction than the comparisons presented here.

Although environmental effects vary by location and biomass type, several general conclusions across indicators are apparent from the simulated results and analyses. Most counties analyzed in the scenarios show potential for a substantial increase in biomass production with minimal or negligible effects on water quality, water quantity, air pollutant emissions, and biodiversity (for avian species analyzed in agricultural scenarios) under the biomass supply constraints assumed in *BT16*. Cellulosic biomass generally shows, favorable performance relative to conventional feedstocks, with harvest of agricultural and forestry residues generally showing the smallest contributions to changes in environmental indicators.

As evaluated in volume 2, biomass produced and delivered to the reactor throat generates GHG emissions because fuel, fertilizer, and agricultural chemicals are consumed. In some counties SOC gains from producing deep-rooted cellulosic feedstocks offset these emissions. Furthermore, as shown in illustrative cases, displacing fossil fuel-derived fuels and products with biomass-derived fuels and products can reduce GHG emissions on a full life-cycle basis that takes into account all life-cycle stages: biomass production and transportation, biomass conversion, and biofuel combustion.

In some locations and under some biomass supply scenarios, challenges may arise for SOC, air quality, water availability, and water-quality management, all of which would benefit from further research and technological improvements. For example, conclusions regarding water consumption by algae in production ponds improve if the recycling of process water is considered. The significance of biomass-related water quality and air quality changes for human health and ecosystems would need to be studied. Biodiversity analyses show a range of outcomes depending on species and location, with possible benefits to richness and range for some species and with other species showing potential adverse impacts that may require additional safeguards and development of wildlife-friendly practices.

This collection of analyses illustrates that biomass production should be integrated into agricultural and forestry systems with consideration of local and regional environmental contexts. Estimates of environmental effects for the scenarios considered in this volume can help the research community, industry, and other decision makers in prioritizing research efforts and data collection, as well as moving toward recommendations of priority locations for biomass production and location-specific best management practices. Research, science-based monitoring, and adaptive management can be used to further enhance environmental benefits of biomass production while mitigating potential negative effects. Strategies to enhance environmental outcomes from biomass production (e.g., landscape design, precision agriculture, the use of waste, and biomass production in conjunction with wastewater remediation) are discussed in chapter 14. Although this study focuses on environmental effects, it is important that future studies investigate environmental, social, and economic effects in a more integrated manner to provide a broader view of sustainability with respect to expanded biomass production in the United States.

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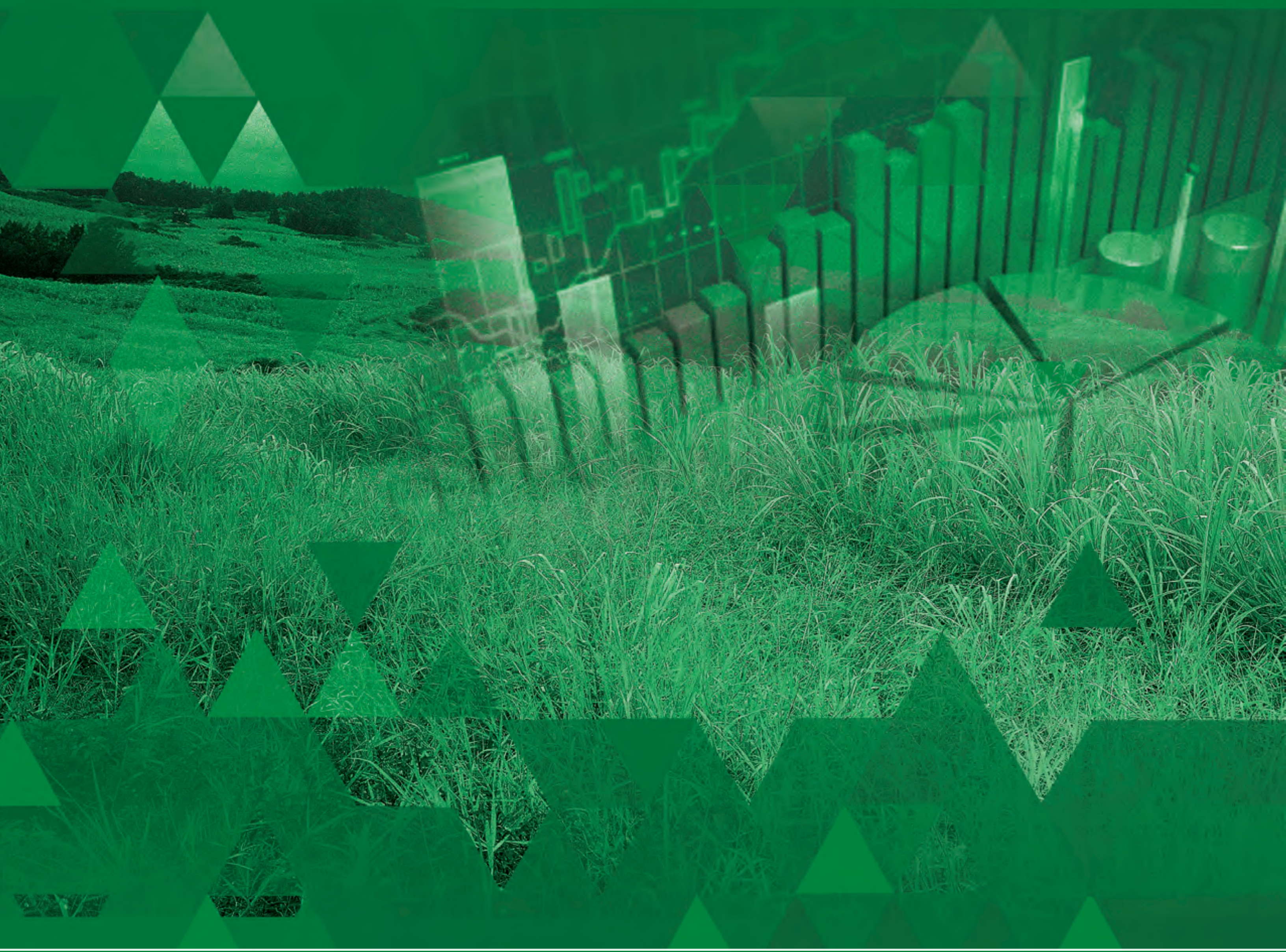
01 | Introduction



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1.1 Background

With the goal of informing national bioenergy and bioproducts research, development, and deployment strategies, the 2016 *U.S. Billion-Ton Report: Advancing Domestic Resources for a Thriving Bioeconomy* (*BT16*), is the third in a series of national biomass resource assessments commissioned by the U.S. Department of Energy (DOE). The *BT16* report is composed of two volumes. Volume 1 focuses on biomass resource analysis (i.e., the potential economic availability of cellulosic and other feedstocks under specified market scenarios) as an update to the two previous Billion-Ton reports, i.e., the *2005 Billion-Ton Study* (Perlack et al. 2005) and the *2011 Billion-Ton Update* (*BT2*) (DOE 2011). In *BT16* volume 1, supplies are quantified under specified constraints. *BT16* volume 2, this report, investigates potential environmental effects of producing biomass supplies for a small set of scenarios simulated in volume 1.

Increasing biomass use can create economic opportunities, enhance energy security, and provide environmental benefits (Rogers et al. 2016). Federal policies aim to foster increased biomass utilization, focusing on growth of second-generation cellulosic biofuels. A report by EPA (2011) concluded that environmental effects of biomass use in the future will be determined by the choice of feedstock, land use change, cultivation, and conservation practices. *BT16* volume 2 investigates a range of these factors to improve understanding of potential environmental outcomes associated with increased biomass production.

Most analyses in volume 2 simulate environmental effects of potential agricultural and forestry biomass production at the county level.[1] The land-use (i.e., land management) change assumptions associated with the scenario transitions are described and discussed, including the assumption and modeling constraint that the agricultural and forestry land bases remain constant during the simulation period. This volume also presents a qualitative analysis of environmental effects of algae production under carbon dioxide (CO_2) co-location scenarios, as well as an analysis of climate sensitivity of agricultural feedstock productivity under a set of potential future scenarios. Finally, strategies to enhance environmental outcomes are described.

Several constraints designed to maintain aspects of environmental quality are employed in volume 1, carried over from the 2011 *BT2*. These constraints include assumptions about tillage classes, residue availability, irrigation, and land-exclusion areas. Supply constraints are summarized in chapter 2 and are described in more detail in BT16 volume 1. Some of these constraints reduce the national potential biomass supply estimates in volume 1 when compared to biomass potential without these constraints. Despite these supply reductions, volume 1 illustrates a situation where large volumes can be produced while not using environmentally sensitive lands or exacerbating soil erosion. However, more thorough analyses are required to estimate possible environmental effects of producing the potential biomass supplies simulated in *BT16* volume 1, and to determine how different types of environmental effects could vary across locations, years, biomass type, biomass yield increase rates, and management practices.

[1] The potential benefits of utilizing biomass wastes for energy (after reduce, reuse, and recycling options have been exhausted) are described in chapter 14 but are not evaluated quantitatively in this volume. Environmental effects of algae biomass are described qualitatively in chapter 12.

1.2 Objectives

BT16 volume 2 seeks to (1) advance the discussion and understanding of environmental effects that could result from significant increases in U.S. biomass production and (2) accelerate progress toward a sustainable bioeconomy by identifying actions and research that could enhance the environmental benefits while minimizing negative impacts of biomass production.

In previous DOE-funded research, indicators were identified that support evaluation of environmental sustainability for a variety of bioenergy systems (McBride et al. 2011; Efroymson and Dale 2015). For this study, environmental indicators were selected in the categories of soil carbon, greenhouse gas (GHG) emissions, water quality, water quantity, biodiversity, and air emissions (see section 1.3). *BT16* volume 2 also includes a discussion of land-use (i.e., land management) change assumptions associated with the scenario transitions (but not including analysis of indirect land-use change [LUC]), analyses of climate sensitivity of feedstock productivity under a set of potential scenarios, and a qualitative assessment of environmental effects of algae production under CO_2 co-location scenarios.

BT16 volume 2 is not a prediction of environmental effects of growing the bioeconomy, but rather, it evaluates specifically defined biomass-production scenarios to help researchers, industry, and other decision makers identify possible environmental benefits, opportunities, and limitations related to increasing biomass production at the local, regional, and national levels. For example, the analyses in this volume can help identify where care should be taken when producing certain feedstocks or where further safeguards are needed to prevent or mitigate potential negative impacts of commercial scale production. Results can also help stakeholders identify locations that are more or less appropriate for certain feedstocks given local conditions, or possible issues that will require further research, monitoring, and adaptive management.

Terrestrial biomass supply projections were simulated in volume 1 using the Policy Analysis System model for agriculture and the Forest Sustainable and Economic Analysis Model for forestry. *BT16* assumptions hold total forestland and total agriculture lands constant throughout the simulation period. Chapter 2 provides a summary of the methodology used to generate the data in volume 1 that are analyzed in volume 2.

It is important to note that the biomass supply potentials presented in volumes 1 and 2 are policy-independent and based on specified price and yield scenarios as well as guiding principles that reflect certain environmental and socioeconomic considerations. For example, some principles aim to maintain environmental quality, such as improved tillage and residue-removal practices, exclusion of irrigation, and reserved land areas to protect biodiversity and soil quality. In this sense, this report may differ from other efforts seeking to depict potential biomass demand and related market, environmental, and land-use interactions under business-as-usual (BAU) scenarios or other specific policy conditions. Further, the scenarios represent total potential biomass production at a market price of $60 per dry ton regardless of end use. Because future end uses may be some unknown mix of biofuels, biopower, and bioproducts, this report presents the biomass supplies as being potentially available for these end uses, but the analysis of environmental effects is limited to production, preprocessing, and delivery of the supplies.

1.2.1 Scenarios

Most chapters in volume 2 analyze three biomass scenarios from volume 1 or a subset of these, such as focusing only on agricultural or only on forestry scenarios. These scenarios assume a price of up to $60 per dry ton at the roadside (i.e., prior to transport, storage, and processing at a biorefinery). This price point is potentially viable and could provide more than 1 billion tons[2] of biomass by 2040. The scenarios include

- **BC1&ML 2017**: 2017 base-case agricultural combined with baseline forestry scenarios: 326 million dry tons[3]
- **BC1&ML 2040**: 2040 base-case agricultural combined with baseline forestry scenarios: 807 million dry tons
- **HH3&HH 2040**: 2040 3% high-yield agricultural combined with HH forestry scenarios: 1.1 billion dry tons.

In these scenarios, BC1 and HH3 are agricultural scenarios and ML and HH are forestry scenarios.

Chapter 2 provides a description of these scenarios. The scenarios were selected to assess and compare potential environmental effects during two time periods with two potential agricultural yield-increase assumptions for the latter year (2040). Potential near-term biomass production is represented in the 2017 scenarios, and significantly expanded biomass production that could occur is represented in the 2040 scenarios. Differences in environmental effects between relatively low and potentially high levels of annual biomass production can be considered by comparing the 2017 and 2040 scenarios. Yield-based environmental effects can be shown by comparing the two 2040 scenarios, given that future biomass availability would greatly depend on yield growth and other technological improvements. For more information on the base-case and high-yield scenarios, see chapter 2 or volume 1. Alternative future scenarios are possible.

In the scenarios identified above, resources evaluated in volume 2 are a subset of the potential resources identified in volume 1. The resources evaluated in volume 2 exclude waste resources and include only corn ethanol and soybean biodiesel portions of currently used resources. Total potential supplies identified in volume 1 and the subset of those supplies analyzed in volume 2 are identified in table 1.1.

Table 1.1 | Biomass Supplies Identified in *BT16* volume 1 and Evaluated in volume 2 for Select Scenarios and Years (in Million Dry Tons)

	Identified in volume 1			Evaluated in volume 2		
	BC1&ML 2017	BC1&ML 2040	HH3&HH 2040	BC1&ML 2017	BC1&ML 2040	HH3&HH 2040
New potential	343	826	1,154	192	669	997
Currently used	365	365	365	134	138	139
Total	709	1,192	1,520	326	807	1,136
Notes	New potential and currently used resources include agricultural and forest biomass and waste resources.			New potential includes agricultural and forest biomass only. Currently used resources include only corn ethanol and soybean biodiesel portions. Waste resources are excluded.		

[2] Here and elsewhere in the report, tons are reported as dry short tons, unless specified otherwise.

[3] The terms base case and baseline have specific meanings in *BT16* that may differ from definitions in other studies.

This study does not include a simulated 2040 BAU scenario because of data limitations and uncertainties about multiple sectors in the future that are outside the scope of this study. The 2017 scenario may represent some characteristics of a future BAU scenario because the former scenario includes only currently available resources (i.e., agricultural residues and forestland resources) with production of conventional crops maintained at current levels. However, the scenario does not include several important characteristics of a BAU case, such as future changes in overall demand, market impacts, and crop yields.

The distribution of potential biomass across the nation in the scenarios reflects the assumption that the total agricultural-land base and the total forestland base do not change between the present and 2040. Modifying scenarios to allow transitions between these major land classes could result in different estimates of environmental effects.

Certain indicators evaluated in this report, including air emissions and GHG emissions, could be affected not only by biomass production, but also by biomass harvest and transportation. To enable analyses of these indicators, logistics inputs (e.g., diesel) were estimated using the Supply Characterization Model (SCM). For the three scenarios, SCM was used to simulate distribution of potential biomass resources to a national grid of hypothetical biorefinery locations and to simulate associated fossil fuel consumption based on current road networks. The application of SCM is described in chapter 6 of *BT16* volume 1 and costs estimated in the model are described in section 2.4.4 of this volume.

1.2.2 Research Questions

BT16 volume 2 investigates and reports on the following questions related to potential biomass production in select scenarios:

- What are the LUC implications of the scenarios over time?
- What are the estimated values of environmental indicators and how do those compare among scenarios?
- What are the potential negative environmental effects, and how might they be managed or mitigated?
- What environmental benefits are possible, and under what conditions do they occur?
- Where is more research needed with regard to quantifying effects, enhancing benefits, and preventing negative consequences?
- How sensitive is feedstock productivity to climate?

Comparisons and insights are based on quantification of environmental indicators for the select scenarios.

1.3 Environmental Indicators of Bioenergy Sustainability

Sustainability is an aspirational concept that denotes the capacity to meet current needs while maintaining options for future generations to meet their needs. Enhancing sustainability of bioenergy systems is part of the mission of the DOE Bioenergy Technologies Office. Specifically, the Office's strategic goal for bioenergy sustainability is to understand and promote the positive environmental, economic, and social effects and reduce the potential negative impacts of bioenergy production activities (DOE 2016). To make the concept of sustainability operational, consistent approaches are required that facilitate comparable, science-based assessments using measurable indicators of environmental, economic, and social processes (Hecht et al. 2009; McBride et al. 2011; Dale et al. 2013). Progress toward defined sustainability objectives can be estimated using these indicators, which can guide behavior toward those intended outcomes.

Many institutions and researchers have proposed indicators to evaluate sustainability of bioenergy pathways (e.g., Roundtable on Sustainable Biomaterials [RSB 2010]; Global Bioenergy Partnership [GBEP 2011]; and the Council on Sustainable Biomass Production [CSBP 2012]). Building from these efforts, researchers at Oak Ridge National Laboratory selected a generic and practical set of indicators to support environmental sustainability of biomass and bioenergy (McBride et al. 2011). Most of these indicators are modeled in this study (table 1.2). These include indicators of soil carbon, water quality and quantity, GHGs, biodiversity, and air emissions. For the purposes of *BT16* volume 2, these indicators are termed "environmental indicators."

Appropriate indicators for a particular application depend on the context for their intended use (Efroymson et al. 2013); therefore, the set of indicators from McBride et al. (2011) in table 1.2 is appropriate for some but not all uses. The context of an assessment of environmental effects typically includes the purpose of the assessment, biomass production and distribution systems, end use, policy conditions, stakeholder values, location, temporal influences, spatial scale, baselines, and reference scenarios. This study adopts a slightly modified list of the indicators proposed in McBride et al. (2011) for the purpose of this initial effort to analyze environmental effects of select terrestrial biomass scenarios from volume 1 (table 1.2). Furthermore, a slightly different set of indicators has been proposed to evaluate the environmental effects of algal biofuels (Efroymson and Dale 2015) and is described in chapter 12.

Table 1.2 | General Environmental Indicators from McBride et al. (2011) (Numbered) and Indicators Modeled for This Analysis (Light Green)

Indicator category	Indicator
Soil quality	1. Total organic carbon (TOC)
	2. Total nitrogen (N)
	3. Extractable phosphorus (P)
	4. Bulk density
Water quality and quantity	5. Nitrate concentration in streams (and export)
	6. Total phosphorus (P) concentration in streams (and export)
	7. Suspended sediment concentration in streams (and export)
	8. Herbicide concentration in streams (and export)
	9. Storm flow
	10. Minimum base flow
	11. Consumptive water use
	Additional: Water yield
Greenhouse gases	12. CO_2 equivalent emissions (CO_2 and N_2O)
Biodiversity	13. Presence of taxa of special concern
	14. Habitat area of taxa of special concern
Air quality	15. Tropospheric ozone
	16. Carbon monoxide
	17. Total particulate matter less than 2.5µm diameter ($PM_{2.5}$)
	18. Total particulate matter less than 10µm diameter (PM_{10})
	Additional: VOCs, SO_x, NO_x
Productivity	19. Aboveground net primary productivity or Yield

1.4 Scope and Scale

The scope of the report is summarized in table 1.3. Agricultural feedstocks include conventional crops, energy crops, and crop residues (fig. 1.1) while forestry feedstocks include logging residues and whole-tree biomass (fig. 1.2). A subset of these feedstocks is considered in various chapters in this volume. In addition, microalgae are the subject of a qualitative analysis. Most analyses consider production and harvest, while analyses of air emissions and GHG emissions consider transport to the biorefinery as well.

Table 1.3 | Scope of Terrestrial Biomass Chapters in *BT16* volume 2

Chap	Indicator category	Indicator	Spatial Extent	Biomass	Scenario	Model	Output
4	Soil quality	Soil organic carbon	Contermi-nous United States	Corn and soybeans for biofuels, wheat, switchgrass, mis-canthus, willow, poplar (surrogates for barley, cotton, oats, sorghum, biomass sorghum)[a]	BC1 2017 BC1 2040 HH3 2040	Surrogate CENTURY Soil Organic Carbon model	Soil organic carbon emissions factor (Mg C/ha/yr)
4	GHGs	CO_2 equivalent emissions (CO_2 and nitrous oxide [N_2O])	Contermi-nous United States	Corn and soybeans for biofuels, biomass sorghum, energy cane, eucalyptus, loblolly pine, miscanthus, poplar, switchgrass, willow, barley straw, corn stover, oats straw, sorghum stubble, wheat straw, hard-wood lowlands (tree), hardwood uplands (tree), mixed wood, softwood natural, softwood planted	BC1&ML 2017 BC1&ML 2040 HH3&HH 2040	Greenhouse gases, Regulated Emissions, and Energy use in Transpor-tation Model (GREET)	GHG intensity (g CO_2e/dt), GHG emissions (g CO_2e, tons CO_2e)
5	Water quality	Total nitrogen load-ing, nitrate loading, total phosphorus loading, sediment loading	Arkansas -White-Red River Basin (AWR) and Iowa River Basin (IRB)	Corn stover (IRB), miscanthus, willow, switchgrass, energy sorghum, sorghum stubble, poplar, willow, (AWR)	BC1 2040 with con-servation practices added	Soil and Water Assessment Tool (SWAT)	Total nitrogen loadings (kg/ha), nitrate loadings (kg/ha), total P loadings (kg/ha), total suspended sediment loading (t/ha), water yield (mm), productivi-ty (t/ha)
6	Water quality	Nitrate loading, total phosphorus loading, sediment loading	Contermi-nous United States	Whole trees (thinnings and clearcuts)	ML 2017 ML 2040 HH 2040	Empirical model	Regional nitrate, phosphorus, and sediment load response curves (kg/ha), increase over pre-harvest reference
7	Water quantity	Water yield	Contermi-nous United States	Whole trees (thinnings and clearcuts)	ML 2017 ML 2040 HH 2040	Water Supply Stress Index (WaSSI) Ecosystem Services Model	Annual water yield (gal/yr), seasonal water yield (gal/month), water yield as an incremental percentage, compared to reference

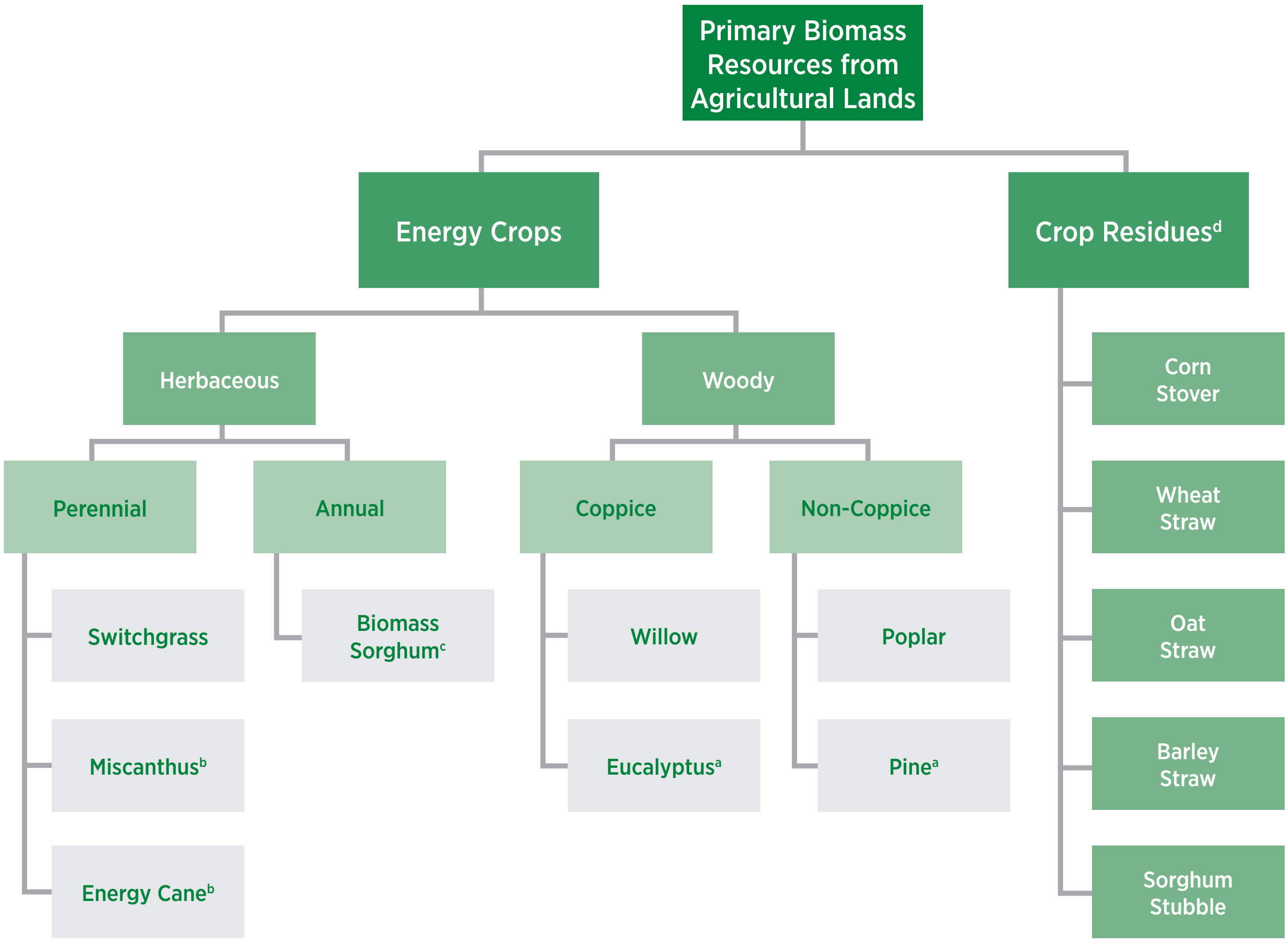
Chap	Indicator category	Indicator	Spatial Extent	Biomass	Scenario	Model	Output
8	Water quantity	Consumptive water use	Contermi-nous United States	Corn for biofuels, corn stover, soybean to biofuels, wheat straw, switchgrass, miscanthus, willow, poplar, southern pine, softwood and hard-wood resources	BC1&ML 2017 BC1&ML 2040 HH3&HH 2040	Water Analysis Tool for Ener-gy Resources (WATER)	Rainwater re-quirements (gal), (gal/acre); irriga-tion requirements (gal), (gal/acre)
9	Air emissions	Total particulate mat-ter less than 2.5µm diameter ($PM_{2.5}$), to-tal particulate matter less than 10µm diam-eter (PM_{10}), ammonia (NH_3), oxides of sulfur (SO_x), volatile organic compounds (VOCs), carbon mon-oxide (CO)	Contermi-nous United States	Corn, corn stover, sor-ghum stubble, wheat straw, barley straw, oats straw, switch-grass, miscanthus, hardwood trees, softwood trees, mixed wood trees, hardwood residues, softwood residues, mixed wood residues	BC1&ML 2017 BC1&ML 2040 HH3&HH 2040	Feedstock Production Emissions to Air Model (FPEAM)	Emissions per ton, emissions compared (as ra-tios) to emissions in the National Emissions Inven-tory
10	Biodiver-sity	Presence of avian species (grassland, forest, or generalist species), species richness, habitat area (range) of avian species	Contermi-nous United States	Switchgrass, mis-canthus, energy cane, pine, poplar, willow, eucalyptus, sorghum, corn, soybean, wheat	BC1 2040, reference 2014	Species distribution model, Bio-EST[b]	Percentage of counties occupied by grassland birds and forest birds, species richness
11	Biodiver-sity	Species among taxa of concern cate-gories: rare native species, keystone species that have a disproportionately large impact relative to abundance, bioindicator taxa that monitor the condition of the environment, species of commercial value, species of cultural importance or spe-cies of recreational value	Contermi-nous United States	Logging residue, whole trees (clearcuts and thinnings)	ML 2017 ML 2040 HH 2040	Habitat suitability framework	Harvest acres, qualitative analysis of hab-itat suitability at ecoregion scales

[a] Chapter includes appendix that discusses soil organic carbon changes that could result from biomass harvest in forests.

[b] Bio-EST – Bioenergy-biodiversity Estimation modeling framework

Abbreviations: Mg C/ha/yr – megagrams of carbon per hectare per year; g CO_2e/dt – grams of carbon dioxide equivalent per dry ton; kg/ha – kilogram per hectare; t/ha – ton per hectare; mm – millimeter; gal/yr – gallons per year; gal/month – gallons per month; gal/acre – gallons per acre

Figure 1.1 | Agricultural feedstocks considered in volume 1 of *BT16*, subsets of which are considered in analyses in volume 2



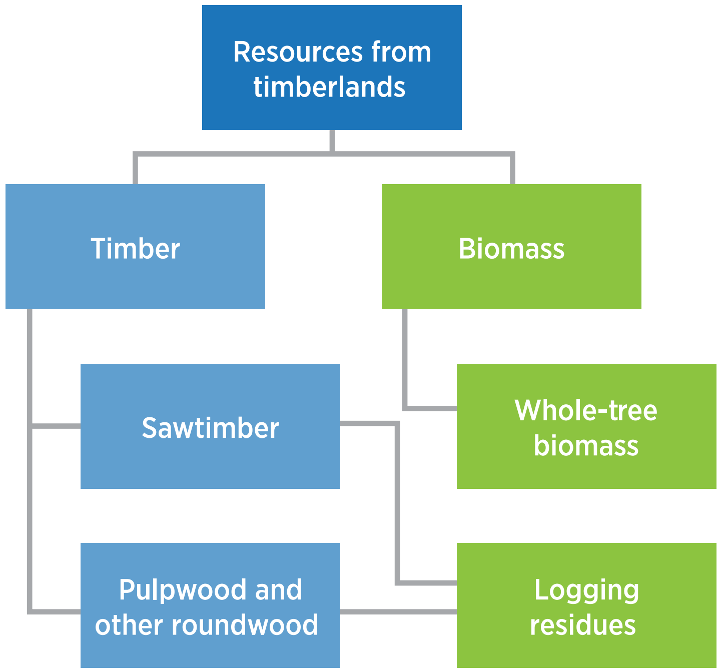
[a] Eucalyptus and pine are newly added feedstocks. They were generalized in the 2011 *BT2* as 8-year rotation, short-rotation woody crops under single-stem management.

[b] Energy cane and miscanthus are newly added feedstocks to the Billion-Ton reporting. They were generalized in the 2011 *BT2* as perennial grasses, along with switchgrass.

[c] The 2011 *BT2* discussed several types of sorghum. For the purposes of this report, "biomass sorghum" depicts any variety developed for high biomass yields, and neither for grain nor sugar content. Budgets for biomass sorghum can represent biomass sorghum, forage sorghum, or sweet sorghum. Modeled yields represent either biomass or forage sorghum; the variety with the highest productivity in a certain region was used.

[d] Agricultural resources already used for biofuels or bioenergy, such as sugar cane bagasse, are reported in volume 1, chapter 2.

Figure 1.2 | Forest feedstocks considered in volume 1 of *BT16*, subsets of which are considered in analyses in volume 2



The extent of analysis in volume 1 is the conterminous United States. Hawaii and Alaska were not included because of a lack of commodity crop data and scarce Forest Inventory Analysis data to support modeling. Most environmental analyses are performed at a national (conterminous United States) extent, with the exception of the water quality analysis for agriculture, which includes case studies focused on the Iowa River Basin and the Arkansas-White-Red Basin. As with volume 1, most analyses and reporting of results are at the county scale. Exceptions include watershed-level analyses for water quality and quantity.

1.5 Supply Constraints in *BT16* volume 1

Several supply constraints designed to reflect guiding principles that account for environmental and socioeconomic considerations were employed in *BT16* volume 1 as well as the 2011 *BT2*. These principles are consistent with DOE's mission to develop biomass as a sustainable resource, and with other research that applies environmental constraints to resource analysis (Schubert et al. 2009; Beringer, Lucht, and Schaphoff 2011). These constraints (summarized in fig. 1.3 and explained further in chapter 2) were carefully chosen to reflect practices that are commonly used in the industry or likely to be adopted in the future. Some of these practices are regulated while others are common industry practices with widespread compliance. Simulations are intended to fulfill projected needs for food, feed, forage, and fiber production, and some constraints are implemented to avoid production on lands with high ecological value.

When deciding which supply constraints to impose in *BT16* volume 1, it was deemed impractical and unrealistic to generate supply projections that are not technically feasible (e.g., removing all residue and debris) or cannot be sustained in the long term (e.g., harvesting residues at levels that exacerbate soil erosion). Using the potential biomass estimates from *BT16* volume 1 means that the same supply constraints are adopted in volume 2, but it is critical to recognize that the environmental effects results are contingent on these constraints. *BT16* volume 2 does not represent the full range of possible environmental effects of potential biomass in the United States; should biomass production practices not follow these modeled supply constraints (for example, using extensive irrigation in the western United States), there would likely be more adverse environmental effects. Analyzing the full range of worst- and best-case scenarios is outside the scope of volume 2. The potential biomass quantified in volume 1 represents a potential future that enables new insights into the environmental effects of biomass production. *BT16* volume 2 analyses will help determine whether the supply constraints applied in volume 1 are sufficient to protect many aspects of the environment or whether adverse effects remain and additional safeguards are needed.

Figure 1.3 | Supply constraints employed in *BT16* volume 1 and adopted in BT*16* volume 2

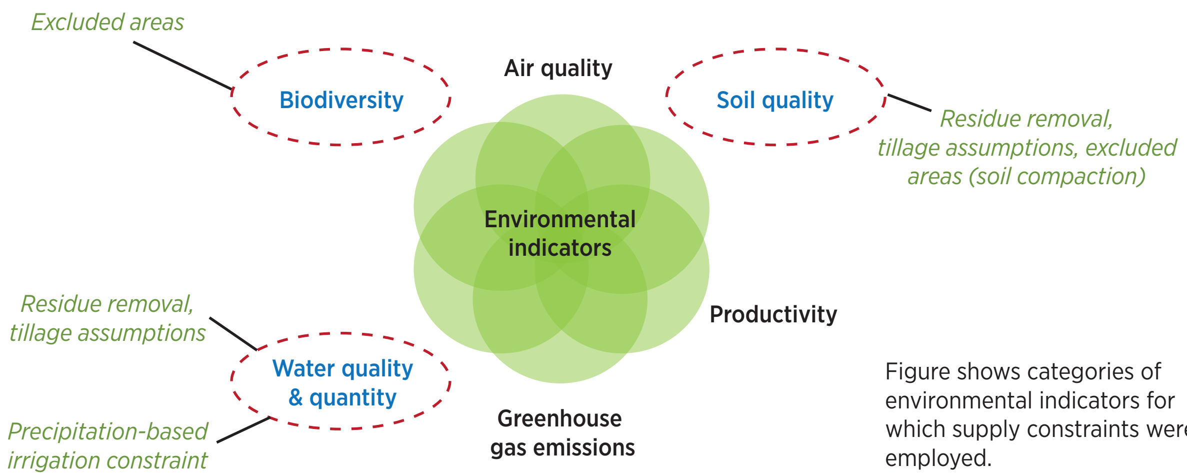


Figure shows categories of environmental indicators for which supply constraints were employed.

1.6 Limitations

Many types of environmental effects are not included in this initial environmental analysis of select *BT16* scenarios. For example, the scenario comparisons do not include an estimate of ecosystem-productivity changes or aquatic-biodiversity changes. In addition, many soil-quality effects (e.g., soil nitrogen, phosphorus, and bulk density) are not modeled. Peak-flow and base-flow indicators of water quantity are discussed but not estimated, and water yield for agriculture is not investigated in detail. The biodiversity analysis addresses select taxa in select regions or ecosystems. The potential for indirect LUC effects nationally and internationally from potential biomass expansion is not quantified in this volume, though issues and definitions are discussed. Environmental indicators for algae biomass for the scenarios in *BT16* volume 1 are not quantified, with the exception of water consumption estimates, but many types of environmental effects are addressed qualitatively. While some aspects of possible economic and social effects are mentioned, *BT16* volume 2 does not investigate these types of potential effects.

Efforts were made to coordinate the various analyses in *BT16* volume 2 to achieve consistency across scenarios and assumptions; however, this initial environmental effects analysis for a Billion-Ton report does not fully integrate results across categories, agriculture, or forestry. Further integration in future Billion-Ton reports will enable more robust understanding of the quantitative relationships—the synergies and trade-offs—between different types of potential environmental effects of biomass production.

1.7 *BT16* volume 2 Organization

The majority of chapters in this second volume of BT16 investigate environmental effects of potential agricultural and forest biomass produced in select 2017 and 2040 scenarios simulated in volume 1 (chapters 4–11). Chapter 2 describes the methodology used in volume 1 to estimate potential biomass supplies and summarizes the scenarios used in volume 2. Chapter 3 provides information to help readers interpret biomass supply results from *BT16* related to LUC (land management). Chapter 4 estimates fossil energy consumption and GHG emissions associated with producing biomass and considers the contribution of changes in soil carbon as a result of producing agricultural biomass on land that was previously in other states or under different management practices prior to production of biomass. Chapters 5 and 6 investigate effects on water quality, i.e., nutrient and sediment loadings associated with agricultural and forestry biomass production, respectively. Chapter 7 evaluates the potential effects of forest biomass harvesting on water yields, and chapter 8 examines the water footprint of agricultural and forest biomass as well as the interplay between feedstock mix and water use. Chapter 9 investigates air pollutant emissions associated with agricultural and forest biomass production and how the spatial distribution of air emissions could potentially impact local air quality. To investigate possible effects on biodiversity, chapters 10 and 11 consider habitat-related responses of select wildlife taxa to potential agricultural and forestry biomass production. Chapter 12 provides a qualitative assessment of environmental effects of microalgae in the context of scenarios in which algae production is co-located with CO_2 sources and that waste CO_2 is used for algae production. Chapter 13 evaluates the sensitivity of potential future biomass productivity to climate. Finally, chapter 14 summarizes and interprets results of previous chapters and explores strategies that could be used to enhance environmental outcomes of biomass production. These include strategies identified in this volume and strategies that are employed or under development elsewhere.

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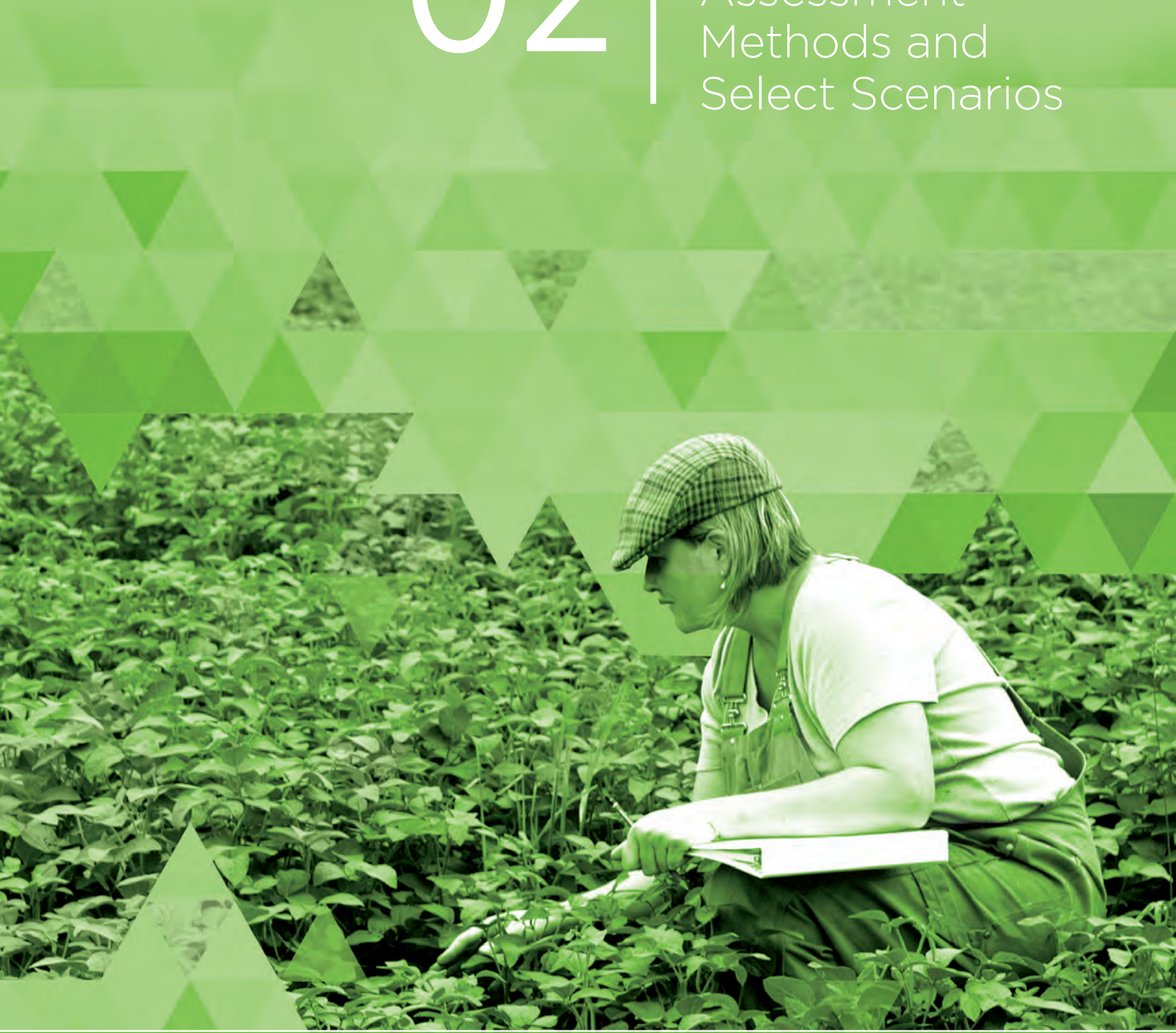
02 | *BT16* Feedstock Assessment Methods and Select Scenarios



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2.1 Introduction

The purpose of this chapter is to provide a brief summary of the methodology used to generate the data described in volume 1 of the 2016 Billion-Ton Report (*BT16*); these data form the basis of the analyses presented in *BT16* volume 2. This chapter is not intended to be a comprehensive description of the volume 1 methodology. For details not addressed here, the reader is referred to the appropriate chapter and associated appendices in volume 1. Furthermore, only the agricultural (chapter 4) and forestry (chapter 3) feedstock assessments from *BT16* volume 1 are summarized in this chapter (sections 2.1 and 2.2 respectively). The final section of this chapter (2.3) summarizes the data selected from volume 1 that are used in volume 2. The methodology used to simulate algae biomass is described succinctly in chapter 12 of this volume. Finally, waste resources, which were components of the biomass in *BT16* volume 1, are described briefly in chapter 14, which addresses approaches to enhance environmental outcomes.

2.2 Agricultural Feedstocks

BT16 employs the Policy Analysis System (POLYSYS), a policy simulation model of the U.S. agricultural sector (De La Torre Ugarte and Ray 2000), to evaluate the potential farmgate supplies of dedicated energy crops and agricultural (conventional crop) residues. POLYSYS uses linear programming models of crop supplies, as well as demand and price components to recursively estimate annual supply, demand, price, and income of conventional and dedicated energy crops for each county in the conterminous United States. Hawaii and Alaska are excluded from the model because significant quantities of conventional crops are not grown in these states.

POLYSYS is a system of interdependent modules that simulate 1) county conventional and dedicated energy crop production; 2) national crop demands and prices; 3) national livestock supply and associated feed demand; and 4) agricultural income. Variables that drive the modules include the planted and harvested area, production inputs, yields, exports, production costs, usage demands, commodity prices, government program outlays, and net realized income. An important component of POLYSYS is its ability to simulate how commodity markets balance supply and demand via price adjustments based on assumed economic relationships (e.g., price elasticities). POLYSYS estimates how agricultural producers may respond to new market opportunities, such as new demand for biomass, while simultaneously considering the effect on conventional crops.

Conventional crops considered in POLYSYS include corn, grain sorghum, oats, barley, wheat, soybeans, cotton, rice, and hay, which together comprise approximately 90% of the U.S. agricultural cropland area. Pastureland is included as permanent pasture and cropland used as pasture. Residues from corn, grain sorghum, oats, barley, and wheat are also estimated. Dedicated energy crops include four herbaceous crops (switchgrass, energy cane, miscanthus, and biomass sorghum) and two classes of short rotation woody crops (SRWCs) (coppice and non-coppice). The SRWC classes are designated as either poplar or pine for the non-coppice class and as willow or eucalyptus for the coppice class because the species assignment of these categories is unique at the county level. However, these individual species are renamed to either coppice or non-coppice in the POLYSYS output data. POLYSYS livestock categories (which contribute to the demand for conventional crops as feed) include cattle, hogs, chickens, turkeys, milk cows, horses, sheep, and goats.

POLYSYS uses a baseline simulation approach in which simulations are anchored to an established baseline of projections for the agricultural sector, and the model simulates scenarios that reflect the impact of changes to the baseline (De La Torre Ugarte and Ray 2000). Linking a scenario to a baseline enables a user to only consider the effect of changes in the economic conditions of interest. For *BT16*, the specified scenarios focused on various offered prices for cellulosic-biomass products (dedicated energy crops and agricultural residues) combined with improvements in energy crop yields, variations in conventional corn yield, and the flexibility of conventional crops to switch among tillage classes. Additional details about the user-specified scenario assumptions are discussed below in section 2.1.2. Section 2.1.1 summarizes the important model inputs, assumptions, and constraints that form the basis of the POLYSYS simulations.

2.2.1 Model Inputs, Assumptions, and Constraints

Baseline: The simulation period for the *BT16* volume 1 agricultural feedstock estimates is 2014 to 2040. POLYSYS anchors its simulations to a baseline that consists of two parts. For the period 2014 to 2023, the 2015 10-year U.S. Department of Agriculture (USDA) baseline projections of crop and livestock

[2] Biogas from animal manures and landfills is analyzed in chapter 5.

supply and demand for the agriculture sector (USDA 2015) is used. Beyond 2024, the USDA baseline is used as an average (linear) trend, and POLYSYS adjusts demand levels and prices to equilibrium around this trend. This approach is used for all food, feed, fiber, fuel, and export variables beyond 2024 except for domestic ethanol and biodiesel demand, which are extended beyond the USDA baseline by holding the 2024 USDA baseline estimate constant. Domestic ethanol and biodiesel demands are held fixed because of the assumption that the renewable fuel standard is met and maintained at the statue level (including 5.2 billion bushels of corn grain to ethanol and 365 million bushels of soy to biodiesel) from 2024 through the remainder of the projection period. This baseline is termed the ‘’extended agricultural baseline” and simplified as the “agricultural baseline.”

Conventional Crops: National Agricultural Statistics Service (NASS) data from USDA are used to generate initial estimates of a county’s planted area, harvested area, harvested-to-planted ratio, and yield for the conventional crops modeled in POLYSYS. Data sources include annual survey data obtained from the NASS Quick Stats database (USDA-NASS 2015) and the geospatial Cropland Data Layers (CDL) (Boryan et al. 2011). The survey data are the primary source of county-level estimates of area and yield. However, in some states and for some crops, survey data are only reported for the NASS Agricultural Statistics Districts (ASDs). In those cases where only ASD-level estimates exist, county-level estimates are calculated by multiplying the ASD planted and harvested areas by the county crop fractions in the ASD based on the crop areas reported in the CDL. The ASD harvested-to-planted ratio and yield are assigned to a county in the ASD if the CDL reports planted area in the county. Four years (2010–2013) of data are averaged to reduce inter-annual variability, and these averages are then used as input by POLYSYS. POLYSYS adjusts the initial estimates of a county’s planted areas proportionally so that the sum of these planted areas matches the USDA baseline (USDA 2015) total of 312.6 million acres (including 57.9 million acres of hay).

Conventional crop planted area and yield are assigned to one of three tillage categories of management: no-till production, reduced tillage, and conventional tillage based on 4 years of historical data (CTIC 2007). Tillage-specific yields are estimated from the corresponding 4-year historical averages by applying regression models (Toliver et al. 2012).

Agricultural Residues: Quantities of removable agricultural residues are based on estimates of total aboveground biomass produced as byproducts of conventional crops, which are then limited by supply constraints (see *BT16* volume 1, appendix C; constraints that are applied for environmental purposes are described in chapter 1 of this volume). Total aboveground biomass residue produced (before operational and other supply constraints are applied) is calculated in POLYSYS based on ratios of residue to grain for corn, barley, oats, sorghum, and wheat as described in table C-3 of appendix C in *BT16* volume 1.

The POLYSYS supply constraint consists of a sustainability constraint and an operational efficiency constraint that are combined to estimate the harvestable yield of residue. The amount of residue that can potentially be removed is limited to the lesser of the two supplies. The harvestable yield is subsequently removed if the price offered exceeds the residue production cost. The residue production cost is only based on the additional operations needed to harvest the residues and replace the nutrients removed; the establishment and maintenance costs of the residues are included in the budgets for corresponding conventional crops. If harvesting is not profitable, the residues are not removed.

The sustainability constraint for residues is designed to limit residue removal to ensure that the tolerable soil-loss limit of the USDA Natural Resources Conservation Service (USDA-NRCS 2016) is not exceeded. This constraint also prevents long-term reduction of soil organic carbon. The Revised Universal Soil Loss Equation – Version 2, the Wind Erosion Prediction System, and the Soil Conditioning Index are used to calculate county-level average-retention coefficients for wind, rain, and soil carbon for each rotation and tillage combination (Muth et al. 2013).

Operationally available residues are limited to 50% of the total-county residue yield starting in 2015, increasing linearly to 90% of available residue yield in 2040 but not exceeding the sustainably available residues (see 4.2.3 and discussion of model sensitivity to operational efficiency under 4.8.6 in *BT16* volume 1). The operational constraint is a function of the total residue yield. This constraint reflects the near-term technical challenges of harvesting variable levels of residue, while allowing for future technological advancements in harvesting equipment that could mobilize greater proportions of the available residue supply.

Dedicated Energy Crops: Energy crop yields are empirically modeled using yields calculated from field trial data collected under the Sun Grant Regional Feedstock Partnership and coupled with climate data generated by the PRISM (Parameter-elevation Relationships on Independent Slopes Model) interpolation method (Daly et al. 2008). Following six crop-specific workshops, data from more than 110 Sun Grant field trials were used to estimate county-specific, per-acre yields using a specialized version of PRISM developed for *BT16* PRISM Environmental-Model (PRISM-EM) (Halbleib, Daly and Hanaway 2012). PRISM-EM is based upon the biweekly values of precipitation, minimum temperature, and maximum temperature estimated by PRISM, and Soil Survey Geographic (SSURGO) Database's soil pH, drainage, and salinity. It uses crop-specific water-use and temperature-tolerance relationships to estimate yield as a function of PRISM climate and soils data. Initial calibrations for these functions are based on known, relative tolerances for warm- or cool-season crops and whether they are grown as annuals or perennials. These functions are coupled with data on soil characteristics and historical weather patterns to generate "first-guess," average, annual relative-yield values (0%–100%). The relative values are regressed with average field-trial yield values to create a transfer function that is used to estimate absolute yield. Since yield data are available for only a few years, in some cases, PRISM-EM is run for the individual years that match those of the data. The estimated yields are adjusted to reflect those under 1981–2010, 30-year average climate conditions. The process of modeling relative yield and estimating absolute yield was done in an iterative fashion during meetings with species experts. In these meetings, yield outliers from the regression function were examined, and model calibrations were modified as needed.

All energy crops are modeled as perennials except for biomass sorghum, which is modeled as an annual crop. Switchgrass is assumed to have a stand life of 10 years with 50% of the expected mature yield potential in year 1; 75% in year 2; and 100% of the expected mature yield potential in years 3–10. Miscanthus has a stand life of 15 years with no harvest of potential yield in year 1; 50% of mature yield potential in year 2; and 100% in years 3–15. Energy cane has a stand life of 7 years with 75% of the expected yield potential in year 1 and 100% yield potential during the remaining years of the stand. Non-coppice SRWCs (poplar and southern pine) are grown on an 8-year rotation with harvest occurring in year 8. Eucalyptus, a coppice SRWC, is grown on an 8-year rotation with harvesting every 4 years. Willow, also a coppiced crop, is grown on a 20-year rotation with harvesting every 4 years. The SRWC rotation lengths were chosen to reflect the shorter time needed to grow these feedstocks for energy use as compared to use for conventional products.

Harvest efficiency factors are also applied to the potential yield to reflect the factor that the harvesting equipment cannot remove all of the available biomass. A harvest efficiency of 90% is applied for switchgrass, miscanthus and energy cane. A 95% efficiency factor is for the SRWCs.

Pasture and Idle Land: The initial value for area of pasture used in POLYSYS is the sum of the cropland used as pasture (11.2 m acres), permanent pasture (402.1 m acres) and other pasture (33.1 m acres) as defined by USDA-NASS (2014). Pasture must meet certain requirements to be eligible for energy crop production including: the land must be rain-fed (not irrigated), there must be additional pastureland of similar quality available for intensified management at a ratio of 1.5 acres for each acre to be used for energy crops; and it must receive 25 inches or more of annual precipitation. This area of pasture is estimated to be 47.1 million acres nationally. Pasture is further classified as either permanent pasture or cropland pasture based on the census data.

The initial estimate of idle land area is also obtained from the 2012 USDA Census Data (USDA-NASS 2014). Land enrolled in the Federal Conservation Reserve Program is included in this initial estimate, but these areas are excluded in POLYSYS from the land base available for conversion to energy crops. The 2015 estimate of idle land used in POLYSYS is 12.3 million acres. The estimate of idle land in the 2040 agricultural baseline projection is 23.2 million acres.

Land Base and Transition Constraints: The total agricultural land base within POLYSYS is fixed throughout the 2015 to 2040 projection period. The land base represents the combination of area in conventional crops, pasture and idle land, as explained above. Natural, reserved, and environmentally sensitive areas such as wetlands, grasslands, and protected forests, as well as all public lands, are explicitly excluded from the agricultural land base. Military lands, powerline cuts, and other areas on which biomass crops could grow are also excluded from the land base.

Although the total land base is fixed, land is allowed to change annually among tillage practices for a crop; the land can also transition among crops and pasture to satisfy baseline demands for conventional crops, while also maximizing profit for dedicated energy crops. Transitions are primarily driven by the expected productivity of land, crop production costs, the expected economic return on the crop, and market conditions. However, for perennial dedicated energy crops, once land is allocated to such a crop, it will remain assigned to that crop for the duration of the crop's rotation period.

Transitions among crops are limited by a 10% maximum annual county-level area change constraint. This constraint is coupled with a tillage flexibility index to control switching among the tillage classes for each conventional crop. The index, which is specified as an input to POLYSYS, can take a value of 1, 2, or 3. A tillage index of 3 allows up to 2.5 times more area to change than an index of 1 as the price for agricultural residues increases. Index values associated with the scenarios are presented below.

Transitions from pasture to energy crops are constrained by annual and cumulative limits that set the maximum percentage of land that can transition. The annual limits are 5% of permanent pasture and 20% of cropland pasture. Cumulative limits are 40% of permanent pasture and 40% of cropland pasture for all energy crops. The exception is biomass sorghum, which is constrained to USDA land capability classes I and II. The pasture conversion constraints are further bounded by a requirement that for each acre of pasture converted to an energy crop, another acre of pasture must be managed for intensified grazing. The additional costs needed for this intensification are used by POLYSYS to determine the economic viability of converting pasture to energy crops. Cumulative cropland conversion to dedicated bioenergy crops is also constrained to 25% of total acreage.

Idle land cannot move into energy crop production in the model simulations. It is accounted for in baseline calibration to determine where, geographically, annual changes in crop acreage in the agricultural baseline either come into or go out of production. Land no longer needed for crop production can transition into idle land and idle land can convert to conventional crop production if needed.

Equipment, Material, and Cost Budgets: A database and associated computer programs are used to estimate the production costs, equipment usage, labor, and material usage (e.g., fertilizer) for the conventional and dedicated energy crops and residues simulated in POLYSYS. Land rents are not included. The database contains individual equipment costs and attributes such as engine horsepower and capacity, and material attributes such as quantities and types of fertilizers and chemicals. The information is based on 2014 costs and operations obtained from various literature sources and subject experts. The database also specifies how these machines and materials are assembled into systems to determine total enterprise budgets. Budgets and material usage for residues only include the additional operations needed to harvest the residues and replace the nutrients removed since the establishment of the associated crop; maintenance costs of the residues are costed in the corresponding conventional crop budgets.

Except for one case, budgets for dedicated energy crops do not include irrigation. The exception is the budget for energy cane in the Imperial Valley of California. However, none of the POLYSYS scenarios analyzed in *BT16* volume 1 included the production of energy cane in California. Also, no Regional Feedstock Partnership field trials that were irrigated were used to estimate energy crop yields.

The agricultural budget database specifies detailed enterprise crop budgets for up to 13 POLYSYS Farm Resource Regions (FRRs), which, in turn, are based on the nine USDA FRRs (USDA-ERS 2000). The additional POLYSYS FRRs arise from splitting the USDA Northern Crescent and Southern Seaboard FRRs into two subregions each, and dividing the Fruitful Rim into three subregions. For conventional crops, budgets are specified for conventional tillage and no-till. Budgets for reduced-till conventional crops are assumed to be the same as the budgets for conventional tillage. The costs and material usage contained in the enterprise budgets are interpolated to ASD-level values for input into POLYSYS using an inverse distance weighting interpolation method (Hellwinckel et al. 2016).

2.2.2 Scenarios

An exogenous price simulation in POLYSYS (hereafter "specified price" simulation) specifies a farmgate price (dollars per dry ton)($/dt) for dedicated energy crops and residues as an input. Such a simulation represents the potential biomass production that could occur if a national market were in place beginning in the near term and offering constant prices until 2040. The specified price (in 2014 dollars) is adjusted for inflation and applied to all counties for all years in the simulation period. POLYSYS then solves for the allocation of land, which produces a mix of biomass that maximizes the profit in response to this price after first satisfying the fixed demands for food, feed, forage, fiber, biofuel, and exports. For example, at a $60/dt specified-price, the resulting supply in 2040 is achieved by the constant presence of a $60/dt market price in all preceding years (2015 to 2040 for residues and 2019 to 2040 for dedicated energy crops).

One base case (BC1) and three alternative scenarios (HH2, HH3, and HH4) were developed in BT16 volume 1 to represent a range of assumptions that incorporate variations in the specified price; flexibility in tillage and crop transitions; yield improvements in dedicated energy crops; and increased yield of corn grain (table 2.1). In all scenarios, planting of dedicated energy crops is not allowed until 2019, but residues are available for the entire simulation period (2015 to 2040). Additional information about the scenario assumptions is presented after table 2.1. A sensitivity analysis of these assumptions is provided in section 4.8 of *BT16* volume 1.

Independent POLYSYS simulations were run at specified prices ranging from $30/dt to $100/dt in $5/dt increments for all conventional crops, dedicated energy crops, and residues together. This approach allows each dedicated energy crop to compete with both conventional crops and other dedicated energy crops for land. It provides an integrated assessment of the potential biomass availability from a mixture of dedicated energy crops and residues under the specified scenario.

The large volume of generated data prevented analysis of the results for every specified-price simulation, so only the results from the $40/dt, $60/dt, and $80/dt results were analyzed in *BT16* volume 1. The results of the $60/dt simulations from the base-case (BC1) and intermediate high-yield (HH3) scenarios were selected for analysis in *BT16* volume 2. The $60/dt-specified price was selected as an economically realistic price level.

Table 2.1 | Description of Agricultural Scenarios Analyzed in Volume 1 (Scenarios Used in Volume 2 Are Shown in Bold)

Scenario identifier	Description	Specified prices for energy feedstocks	Tillage flexibility constraint	Energy crop yield improvement (annual)	Conventional crop yields
BC1	**Base case (1%)**	$40, **$60**, $80	**1**	**1%**	**Baseline for all crops**
HH2	High yield (2%)	$40, $60, $80	3	2%	High corn grain, baseline for all other crops
HH3	**High yield (3%)**	$40, **$60**, $80	**3**	**3%**	**High corn grain, baseline for all other crops**
HH4	High yield (4%)	$40, $60, $80	3	4%	High corn grain, baseline for all other crops

Additional details regarding the scenarios are presented below.

Tillage Flexibility Constraints: As mentioned above, the tillage flexibility constraint controls the amount of land that changes tillage class annually for a given conventional crop. A tillage index of 3 allows up to 2.5 times more area to change than an index of 1, subject to an overall maximum annual change constraint of 10%.

Energy Crop Yield Improvements: Base-case and high-yield scenarios represent possible yield improvements over time that may be achieved with a mix of improved management practices and crop genotypes. These assumptions are derived from a series of workshops in 2010 drawing on expert opinion (INL 2009). Yield improvements are applied and compounded annually beginning in 2015.

Conventional Crop Yields: Yields for all conventional crops except corn are set to match their respective agricultural baseline values over the simulation period. For BC1, corn yield is also kept at its baseline values, but for HH2, HH3, and HH4, the corn yield is allowed to increase more rapidly to reach a national target of 265 bushels per acre in 2040. This increased yield allows for greater adoption of no-till management and a greater production of corn residues.

2.3 Forestry Feedstocks

The linear programming Forest Sustainable and Economic Analysis Model (ForSEAM) is used to estimate roadside forestland production over time to meet demands for both traditional forest products and biomass feedstock. The biomass feedstocks include forest residues and whole trees harvested explicitly for biomass uses. Wood wastes from sawmills and from landfills (e.g., construction and demolition waste) are not estimated by ForSEAM.

ForSEAM can be used to estimate the quantity of biomass that might be available as energy feedstocks for 305 production regions that correspond to the NASS ASDs (He et al. 2014). The model also estimates costs, land use, and competition among lands. ForSEAM seeks to determine the mix of harvested stand types that minimizes total cost (harvest and other costs) under a production demand target for wood products and biomass. The model requires that projected traditional timber demands be met first (i.e., traditional timber demands are fixed across scenarios). The mix of stand types used to meet the demand is subject to land, growth, and other constraints. The model estimates production based on location, stand type, stand's average tree diameter, slope of the land on which the stand occurs, harvest method, type of product that will be produced, and time of harvest. Regional model results are disaggregated to the county level using the ratio of the county planted area to the regional total planted area, calculated from the Forest Inventory and Analysis (FIA) program database (USDA-FS 2015).

ForSEAM requires estimates of projected demands for sawlogs and pulpwood. These demand levels are obtained from the U.S. Forest Products Module (USFPM) (Ince et al. 2011a). The USFPM is a global, forest-products, partial-equilibrium market model that operates within the Global Forest Products Model. USFPM provides detailed information on forest products production, trade, and prices for the North, South and West (see chapter 3 in BT16 volume 1) regions of the conterminous United States. In USFPM, wood energy demand can compete for supply sources also used to make lumber, panels, and paper; forest inventory responds to harvest and growth. U.S. demand for wood energy is specified at the national level, and the model determines the fuel feedstock-supply allocation among the North, South, and West regions by using the lowest-cost feedstock sources to meet the national demand. The U.S. demand for wood energy includes demands for residential and industrial fuel wood, as well as the potential for increased demand for wood pellets for export, and/or assumed domestic demands for biopower and biofuels. Weights based on inventory are used to develop state estimates of demand for these traditional wood products, which then serve as input for ForSEAM.

2.3.1 Model Inputs, Assumptions, and Constraints

Stand Types and Characteristics: Five stand types are simulated in ForSEAM: upland hardwood, lowland hardwood, natural softwood, planted softwood, and mixed wood. For each stand type, three diameter sizes are modeled: class 1 (stands with diameter at breast height (dbh) of >11 inches for hardwood and >9 inches for softwood); class 2 (stands with dbh between 5–11 inches for hardwood and dbh between 5–9 inches for softwood); and class 3 (stands with dbh <5 inches).

For the initial simulation year, clearcut yields are calculated using information on standing tree volume and corresponding timber area from the FIA database aggregated to the county level. The thinning yield is 70% of the clearcut yield, assuming a combination of thinning-from-above (Coops et al. 2009; McMahon 2016) when harvesting conventional products and only taking the smaller-diameter trees when harvesting whole trees for biomass.

If stand types in classes 2 and 3 are not harvested, they continue to grow and become class 1 and class 2 stands respectively, depending on the annual increment of quadratic mean diameters for that stand type. If class 2 stands are harvested by thinning, they are not available for additional harvesting until they become class 1 stands. Annual growth yield is based on the net annual growth and the corresponding timber area. For all years beyond the initial year, the yield is assumed to be the initial yield, plus the total growth yield, multiplied by the total numbers of years from the beginning to the current simulation year.

USFPM estimates five timber products including softwood sawlogs, softwood pulpwood, hardwood sawlogs, hardwood pulpwood, and other industrial roundwood. The demands for hardwood sawlogs and other industrial roundwood are aggregated to hardwood sawlogs in ForSEAM. The roundwood harvested for fuel is disaggregated to softwood and hardwood fuel wood, using a ratio calculated with data from Howard, Quevedo, and Kramp (2009). In ForSEAM, sawlogs originate from class 1-size trees. Pulpwood originates from trees in size classes 1 and 2. Whole-tree biomass feedstocks are from trees in classes 2 and 3. The volume of hardwoods (lowland and upland) and 37.5% of mixed wood stands are used in the model for hardwood timber products. The volume of softwood (natural and planted) and 62.5% of mixed wood stand species is used for softwood timber products.

Whole-Tree Harvest: There are four combinations of harvest methods and intensity for whole trees: 1) full-tree clearcut, 2) full-tree thinning, 3) cut-to-length clearcut, and 4) cut-to-length thinning. The full-tree method can use the entire tree, including branches and tops. The cut-to-length method harvests logs only, leaving logging residue behind. For both methods, the intensity can be either clearcut or thinning. Clearcutting removes all of the standing trees in a selected area. Thinning removes part of the standing trees in a selected area.

Annual harvesting intensity is limited to 5% of the amount of timberland area within a ForSEAM region. Also, the harvest intensity is restricted at the state level to ensure that growth exceeds harvest removals. Together, these two factors prevent the model from harvesting more wood in a region than can be grown based on the corresponding state's growth rate. The value of 5% is estimated by taking the potential production compared with the 2010 projected demand estimated by the USFPM. This value was found to be sufficient to meet the future conventional wood demand.

Only class 2 stands may be harvested by clearcutting or thinning. Cut-to-length is used only for softwood timber in the North Central and Inland West regions for class 1 and class 2 stands. No harvesting is allowed on lands with a slope >40% in the Northeast, South, North Central, and Inland West regions since it is assumed that cable harvesting systems are not available in these regions. ForSEAM assumes that only in the Pacific Northwest trees can be harvested for conventional products on timberlands in both slope classes (≤40% and >40%).

A constraint for clearcut and thinning areas was applied in the West, South, and North (see chapter 3 in *BT16* volume 1) to ensure that a certain amount of production was excluded from thinning. This constraint is included because the benefits of thinning, such as increased yields and revenue, are hard to measure and capture at the scale of the current model. In the model, the clearcut portion is 42%, 28%, and 10% for the West, South, and North, respectively.

The timberland constraints built into ForSEAM limit harvested timberland for conventional wood to the maximum percentage of the existing volume of class 1 land that can be harvested in any one period. Other constraints limit the harvest intensity to the existing volume of classes 2 and 3. The third timberland constraint requires cut-to-length harvest acres to equal full-tree harvesting acres in the North Central region and Inland West region. A major timberland constraint restricts logging residue removal to those lands that provide traditional products; growth is also restricted. The volume of trees removed must be less than the 2014 base-year harvest plus the annual growth that occurs within the state on the remaining stands to ensure that harvest never exceeds growth.

Logging Residue Removal: Not all available logging residues are harvested for biomass feedstock use. A retention rate of 30% is applied to residues from clearcut, full-tree harvesting on timberland with a slope of ≤40%. If the available logging residues are from stands located on timberland with a slope of >40%, all of the logging residues are left on the site. If the timberland is thinned (partially cut), 30% of the residues are retained on-site, (i.e., a 30% retention rate) if the slope is >40%. All logging residues from thinned stands are available for harvesting as biomass feedstocks in the model if the slope is ≤40%. The underlying assumption is that residues will still be left on-site because of tree breakage and losses from harvesting trees and that the remaining trees will provide sufficient protection from soil erosion and loss of soil organic carbon.

Land Base and Transition Constraints: To be consistent with the agriculture assessment, only production in the conterminous United States is estimated. Total forestland in the conterminous United States is 623 million acres. Timberland is defined as forestland that produces more than 20 ft^3 per acre of industrial wood annually where harvesting is not prohibited. There is 475 million acres of timberland in the conterminous United States.

The land base for ForSEAM modeling only includes timberland that is classified as nonreserved federally or privately owned and is no more than 0.5 mile from an existing road system. Data from the FIA program database indicate that about 300 million acres of privately owned timberland and approximately 87 million acres of federal lands meet this definition (387 million acres total). The available land base is also categorized into two ground slope classes: 1) slope ≤40% and 2) slope >40% based on the FIA database.

After timberland is clearcut, replanting occurs if the stand was originally classified as planted softwood, and natural regeneration occurs if the stand is one of the other four types. All stands are assumed to replant or regenerate in the same stand type (e.g., natural hardwoods regenerate back to natural hardwood forests).

Equipment, Material and Cost Budgets: A database and associated computer programs based on information from the Consortium for Research on Renewable Industrial Materials (CORRIM) (Oneil and Lippke 2010; Johnson et al. 2005) are used to estimate the harvest equipment, labor, materials, and costs used in ForSEAM. The database contains individual machine costs and attributes such as engine horsepower, capacity, and operation (e.g., felling). The database specifies how these machines are assembled into systems to determine total budgets. Harvest systems and budgets are estimated for each feasible combination of stand type; stand diameter class; ground slope class; harvest method (full tree or cut-to-length); harvest intensity (clearcut or thinning); and product (merchantable products of sawlogs and pulpwood, logging residues; and whole-tree biomass) in five regions (Northeast, North Central, South, Inland West, and Pacific Northwest). The 2004 CORRIM equipment costs are updated to 2014 prices using the Producer Price Index for construction machinery manufacturing (Bureau of Labor Statistics 2015).

Stumpage prices are based on the RISI (2008) international wood fiber report data. The pulpwood price is used as the stumpage price for hardwoods and softwoods stands in class 2. For mixed wood, the price is calculated as 37.5% of the hardwood stumpage price plus 62.5% of the softwood stumpage price. For each stand species, the stumpage price of a class 1 stand is twice that of a class 2 stand. The class 3 stand stumpage price is 50% of the class 2 stand price. If logging residues are collected from the harvested site, their stumpage price is the fraction of the whole-tree stumpage price. The price is based on the ratio of the residue yield to the whole-tree yield, using the FIA database to calculate that value. Price data for hardwood, pulpwood, and roundwood in the West region are not available. In these cases, the 2007 estimate of $23.48 per dry ton for hardwood in the West is used.

2.3.2 Scenarios

Six scenarios are used in *BT16* volume 1 to evaluate U.S. forest-product market outcomes for three levels of national wood-biomass feedstocks demand, two levels of housing recovery, and two levels of southern pine-plantation growth rates (table 2.2). In all scenarios, 1) U.S. demand for solid wood products is driven by projected growth trends in U.S. real gross domestic product (GDP) and single-family housing, and 2) U.S. demand for paper products is driven by real GDP and by recent historical growth rates for advertising expenditures in print media and electronic media (Ince et al. 2011b). Net exports of U.S. forest products are influenced by projections of global demand for forest products and projections of global currency-exchange rates. All scenarios use the 2012 USDA Economic Research Service global projections for GDP and currency exchange rates for all countries to 2030 (USDA-ERS 2015).

The baseline scenario represents moderate housing and low wood energy demand (scenario identifier ML in Table 2.2). It is derived from Ince and Nepal (2012), which assumes a moderate rebound in housing starts. The wood energy demand, which increases by approximately 26% between 2010 and 2040, is estimated by the historical econometric relationship between fuelwood consumption and GDP growth (Simangunsong and Buongiorno 2001). The five alternative scenarios shown in table 2.2 (HL, MM, HM, MH, and HH) vary in housing starts and wood energy demand. Additional information about the assumptions is presented after table 2.2.

For each scenario, ForSEAM was run at specified-biomass demand levels ranging from 1 million dry tons (Mdt) to approximately 185 Mdt in increments of 1 Mdt. Logging residues to meet the specified biomass demand are available only when trees are harvested for conventional timber markets. When those markets are saturated, logging residues are no longer available as a source of biomass. Logging residues are assumed to be harvested as an integrated product, along with the conventional sawlogs and pulpwood, at a relatively low extra cost compared with whole-tree biomass. Therefore, all available logging residues are harvested first in the model to meet the specified biomass-demand level. When the demand is greater, then the model solves for the lowest-cost whole-tree biomass to supplement the demand.

The large volume of data generated by this approach prevented analysis of the results for every simulated demand level. Instead, the highest specified-demand run that had a solution in all years of each scenario was selected to provide a representative estimate of production and harvested acreage. The selected biomass-demand level for each scenario is shown in parentheses in table 2.2.

Table 2.2 | Description of Forestry Scenarios Analyzed in *BT16* volume 1 (Scenarios Used in Volume 2 Are Shown in Bold)

Scenario identifier	Description	Specified biomass demand levels	Housing starts	Wood energy demand
ML (baseline)	**Moderate housing–low wood energy**	1 to 187 Mdt **(116 Mdt)**	**Returns to long-term average by 2025**	**Increases by 26% by 2040**
HL	High housing–low wood energy	1 to 187 Mdt (117 Mdt)	Adds 10% to baseline in 2025 and beyond	Increases by 26% by 2040
MM	Moderate housing–moderate wood energy	1 to 184 Mdt (93 Mdt)	Returns to long-term average by 2025	Increases by 86% by 2040
HM	High housing– moder-ate wood energy	1 to 184 Mdt (94 Mdt)	Adds 10% to baseline in 2025 and beyond	Increases by 86% by 2040
MH	Moderate housing–high wood energy	1 to 184 Mdt (82 Mdt)	Returns to long-term average by 2025	Increases by 150% by 2040
HH	**High housing– high wood energy**	1 to 184 Mdt **(83 Mdt)**	**Adds 10% to baseline in 2025 and beyond**	**Increases by 150% by 2040**

Housing Starts: Moderate housing starts assume a rebound in housing, with average single-family housing starts increasing to the long-run historical trend of 1.09 million per year by 2020 and following a slowly increasing trend thereafter. The high housing option assumes starts would be 10% higher by 2025 and would stay 10% higher throughout the projection. The top quartile of housing starts from 1959 to 2011 is at least 10% above the long-term average, indicating that the higher rate is feasible.

Wood Energy Demand: As discussed above, low wood energy demand is estimated by the historical econometric relationship between fuel wood consumption and GDP growth (Simangunsong and Buongiorno 2001). The moderate and high wood-energy demand scenarios represent increases in domestic and/or pellet export wood-energy demands that are not captured in the historical relationship between fuel wood use and GDP (Abt et al. 2014). The moderate wood-energy demand scenario is estimated as a quadratic demand function that incorporates the announced production facilities in the Forisk Consulting wood energy database through 2020 (Forisk Consulting 2014) and an increase based on continued pellet exports. The high wood-energy demand scenario assumes that production in 2020 will be twice as high as in the moderate scenario.

2.4 Environmental Effects Assessment

2.4.1 Farmgate and Landing Supplies

All of the *BT16* volume 2 environmental effects assessments use farmgate or forest landing estimates of agricultural and forestry supplies, respectively. Only a subset of the agricultural and forestry assessment scenarios and projection years are selected for use in the *BT16* volume 2 analyses. The scenarios are selected to represent a near-term base case (2017), a long-term base case (2040) and a long-term high-yield projection (2040). The $60/dt price runs from the BC1 and HH3 scenarios (table 2.1) were chosen from the agricultural assessment for the base-case and high-yield projections. Thus, the three agricultural scenarios addressed in this volume are BC1 2017, BC1 2040 and HH3 2040. The 3% annual yield increase scenario was selected over the 4% annual yield increase scenario because the former was considered more conservative. Annual county-level data sets containing simulation results for planted area, harvested area, production, and yield for conventional crops, residues, and dedicated energy crops were created for the selected scenarios. Of these scenarios, adjustments were made to exclude wastes and add conventional biofuels (see table 1.1 of *BT16* volume 1).

From the forestry scenarios in *BT16* volume 1, the baseline (ML) and high housing-high wood energy (HH) scenarios were selected for analysis in volume 2 (table 2.2). Thus, the three forestry scenarios addressed in this volume are ML 2017, ML 2040, and HH 2040. Annual county-level data sets of harvested area and production by stand type, material type (residue or whole-tree), size class, harvesting method, slope class, and land ownership for conventional wood products and bioenergy usage were created for the selected scenarios. As mentioned above, only results for the selected demand level were included (table 2.2).

In addition to the area and production data from the select scenarios, selected data from the agricultural budget databases were provided to some of the *BT16* volume 2 investigators. The data included equipment characteristics (e.g., horsepower, fuel usage) and quantities of fertilizers and chemicals applied to establish, maintain, and harvest the conventional crops, energy crops, and residues. Harvest equipment characteristics were provided from the forestry budget database.

2.4.2 Attribution

In the case of agricultural and forest residues, attribution of environmental effects can theoretically by applied to the primary crop (e.g., corn grain, sawtimber), the residue (e.g., corn stover, logging residues) or a combination of the two. In this volume, decisions on attribution of residues vary by chapter, and are specified below in table 2.3.

Table 2.3 | Specification of Attribution of Environmental Effects Between Residue Removals and Primary Biomass Broducts (Effects Are Attributed Entirely to the Biomass Removal for Energy Crops and Whole-Tree Harvests)

Indicator	Chapter	Attribution
Greenhouse gas emissions (agricultural and forest residues)	4	Agricultural residues burdened with emissions from harvest and supplemental fertilizer. Forest residues burdened with 10% of emissions per *BT16* volume 1 approach to costing.
Water quality (agricultural residues)	5	Loadings attributable to primary crop, residues, and energy crops on areas harvested for biomass.
Water quality (forest residues)	6	Loadings attributable to biomass harvest where whole-tree biomass harvests occur. Assumed that there would be negligible incremental impacts from removing residue after harvests, therefore they were not considered in the analysis.
Water yield (forests)	7	Yield attributable to biomass harvest.
Water consumption footprint (agricultural and forest residues)	8	Consumption attributable to primary crop and biomass harvest.
Air emissions (agricultural and forest residues)	9	Emissions from production attributable to primary product; emissions from harvest activities allocated between crop and residue; additional chemical and nutrient applications to replace nutrient removal attributable to the residue.
Biodiversity (agricultural residues)	10	Not applicable. Residue removal not considered.
Biodiversity (forest residues)	11	Changes attributable to residue removal.

2.4.3 Inter-Annual Crop Transition Estimates

Some of the *BT16* volume 2 analyses using the agricultural scenarios also required estimates of inter-annual and cumulative crop transitions. POLYSYS generates files that contain county-level estimates of inter-annual changes of crop-planted areas that correspond to the county-level production estimates. Using these data, we generated interannual county-transition proportions (e.g., 2020–2021) by dividing the changes in county crop-planted area by the total planted area in each county. Expressing the changes as proportions allows for the calculation of multi-year transitions by multiplying the corresponding inter-annual proportions (e.g., multiply 2020–2021 proportions by 2021– 2022 proportions to obtain 2020–2022 proportions). These results provide estimates of cumulative changes in crop-planted areas for each county.

2.4.4 Supplies Delivered to Biorefineries

Some *BT16* volume 2 analyses include a subset of the results from the delivered supply[1] analysis described in chapter 6 of *BT16* volume 1. To summarize, this analysis used a geographically based modeling system to allocate feedstock supplies to potential utilization facilities and calculate the delivered price and

[1] Supply is delivered to the throat of the biorefinery. Simulations are made for biochemical and thermochemical conversion platforms, so future products and conversion processes are not considered in this analysis.

quantity of the supplies (Webb et al. 2014). Costs of unit operations (storage, size reduction, and handling) and dockage (additional charges incurred for disposal of feedstocks that do not meet quality specifications) are derived from previous studies (Cafferty et al. 2014; Kenney et al. 2014). Locations of utilization facilities are based on minimizing the average total delivered feedstock cost. Facility locations are selected iteratively, in order of increasing total delivered cost, until all of the available supply is used.

For each feedstock, five logistics costs are estimated: (1) production costs; (2) other logistics costs (storage, handling, and preprocessing); 3) time transportation cost; (4) distance transportation cost (loaded), and (5) distance backhaul cost (empty). Production costs include operations on the farm (agricultural feedstocks), at the roadside (forestry feedstocks), or at the sorting facility (wastes), along with the grower payment (agricultural feedstocks) or stumpage price (forestry feedstocks). For agricultural biomass, a cost curve was generated from the $60 simulation for the base case and 3% high-yield scenario to represent the production of biomass at varying prices (see chapter 6 of *BT16* volume 1). The farmgate agricultural biomass cost includes production, maintenance, harvesting, and an assumed 10% profit per ton of biomass. Roadside forestry biomass cost includes stumpage and harvesting. Transportation cost is divided into time- and distance-based components. The distance component of transportation cost, namely fuel, varies by the distance traveled. The time cost accounts for the capital cost of the truck and labor cost. Fuel economy is known to change with payload, so distance transportation costs are estimated for fully loaded trucks going to the facility and for empty trucks on the backhaul. The other logistics cost parameter includes the costs of all other operations, such as storage, handling, and preprocessing. The final delivered supply is characterized as the quantity and combined weighted average cost by feedstock at the county of origin for the specified scenarios. The county estimates of feedstocks transported and the associated transport distances are provided to the *BT16* volume 2 investigators requiring such data.

Biomass delivered at prices up to $100 per dry ton was considered to be economically feasible given the uncertainty in simulation results and the potential for reducing logistics costs with technology improvements. Thus, energy consumption and emissions for biomass logistics were considered only for biomass with delivered costs up to $100 per dry ton.

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3 | Land Allocation and Management: Understanding Land-Use Change (LUC) Implications under *BT16* Scenarios

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3.1 Introduction

3.1.1 Objectives

The objective of this chapter is to help readers interpret results from the *2016 U.S. Billion-Ton Report* (*BT16*) volume 1 related to the phenomena generally called "land-use change" (LUC) and "indirect land-use change" (ILUC). LUC can be described as a "change in the use or management of land by humans" (ISO 2015; IPCC 2000). However, definitions of LUC have varied widely in the literature (see appendix 3-A). In this chapter, unless specified otherwise, LUC refers to the effects on land that are caused or implied by the biomass production systems simulated in *BT16*. We describe where, how much, and what type of LUC is associated with the simulations.

The following questions and responses illustrate chapter goals and content:

- Why is analysis of LUC included in the *BT16* volume 2?
    - LUC is an important concern that can determine the acceptability of bioenergy, and current U.S. policies call for monitoring and reporting on environmental effects of biofuel pathways inclusive of LUC.
    - LUC effects are far-reaching and can be measured across all environmental indicators (see chapter 1).
- What are the LUC implications of *BT16*?
    - LUC effects associated with any simulation are determined by model input parameters and assumptions, and are distinctive for each scenario.
    - *BT16* scenarios apply constraints that prohibit net change in the total area of major land classes so that the total area and extent of forestland and agricultural land are held constant throughout all simulations and time periods.
    - Because total forest and agriculture land areas remain fixed, the most significant LUC effects relevant for environmental assessment under *BT16* scenarios involve changes in land management practices.
    - Building on continued trends of yield improvement and cropland area reduction, a principal manifestation of LUC is the net reduction in annual crops, which are replaced by idle land and perennial cover within the fixed agricultural area.
    - Under *BT16* scenarios at $60 per dry ton or less, by 2040, the area in perennial cover increases compared to the agricultural baseline in 2015 by
        - 24 million acres under the base case (BC1)
        - 45 million acres under the 3%-yield annual growth case (HH3).
    - Under the same scenarios, the area in annual crops falls compared to the agricultural baseline in 2015 by
        - 34 million acres under the base case (BC1)
        - 55 million acres under the 3%-yield annual growth case (HH3).
    - Approximately 10 million acres allocated to annual crops in the agricultural baseline in 2015, transitions to idle land under the *BT16* scenarios.

- What other LUC issues are relevant to *BT16*?
  - It is essential to understand the differences between studies designed to estimate policy-driven LUC and resource assessments such as *BT16* that examine potential biomass supplies under specified conditions.
  - The assumptions and constraints used in *BT16* illustrate spatially explicit biomass supplies while excluding most potential LUC concerns by design.
  - Estimates of change always depend on the reference case, and in this chapter we consider the BC1 simulation in 2017, and the agricultural baseline (described in volume 1) in 2015, 2017, and 2040, as references.
  - *BT16* does not simulate other references or define a "business as usual" case for 2040. However, other possible reference case considerations are discussed in appendix 3-A.
  - Replicable methods to measure land-related effects are essential for science-based analysis of biomass production systems.
  - Further research is required to clarify LUC effects of U.S. biomass production systems under different supply, demand, and policy scenarios.

3.1.2 The Importance of LUC and Related Indicators

LUC is important because all other environmental indicators, many of which are addressed in this report, as well as social and economic indicators, can be impacted by LUC (McBride et al. 2011; Dale et al. 2013). Under the Renewable Fuel Standard, LUC and indirect effects caused by U.S. biofuel policy must be considered. Since 2008, the effects of LUC have dominated discussion of environmental impacts of bioenergy because of their implications for greenhouse gas (GHG) emissions, biodiversity, food security, and other aspects of the environment.

The scientific literature identifies two LUC-related issues of high concern: (1) potential loss of areas of high conservation value, such as forests, peatland, wetlands, and native prairies; and (2) potential loss of agricultural output or displacement of cropland. The first type of potential LUC has implications for biodiversity, GHG emissions, carbon stocks and sequestration rates, and other environmental indicators, as discussed in this volume. The second type of potential impact has implications for food security, as discussed in the literature (e.g., GFMG 2010; Durham, Davies, and Bhattacharyya 2012; IFPRI 2015; Kline et al. 2016), as well as indirect effects. Chapter 2 discusses how *BT16* applies modeling assumptions and constraints designed to estimate potential U.S. biomass supplies while controlling for and mitigating these two specific concerns. In this chapter, we focus on LUC implications of the land management practices assumed in association with *BT16* scenarios. As discussed in other chapters, changes in crop type and management are expected to affect most environmental indicators and especially those for soil carbon, GHG and air emissions, water quality, and biodiversity.

3.2 Research Goals Guide Choices for Model Parameters, Assumptions, and Definitions

Different land input parameters and assumptions are applied to answer different questions about land and bioenergy (Dale and Kline 2013a). Many studies have aimed to address questions about the potential effects of a defined biofuel policy on land use (e.g., Fritsche and Wiegmann 2011; Fritsche, Sims, and Monti 2010; Oladosu et al. 2012; Oladosu and Kline 2013; Plevin et al. 2015; Valin et al. 2015; Taheripour and Tyner 2013; Tyner et al. 2010). LUC estimates under specified scenarios require assumptions about relationships among productivity, prices, different commodity markets, and land (characterized by types, costs, locations, ownership, markets, etc.). LUC modeling studies are based on the assumption that biomass production will displace other production or other specific land uses.

3.2.1 The Differences between *BT16* and Analyses that Focus on LUC

BT16 is not an LUC study. Rather, *BT16* describes domestic biomass resource potential with specific limitations on displacing other production (see detailed discussion in section 3.4 below). *BT16* addresses questions about the locations and types of potential biomass within fixed agricultural and forestland areas and under scenarios that provide supplies not only for biomass, but for other projected agricultural and forestry market demands. *BT16* scenarios are neutral about end use (i.e., the potential biomass supplies could be used for any purpose) and biofuel or other policies. While existing policies are implicitly reflected in the USDA baseline projections (USDA 2015a), the U.S. Forest Products Module of the Global Forest Products Model (see chapter 2), and the *BT16* agricultural baselines developed for *BT16* scenarios (see chapter 2), *BT16* supply simulations aim to illustrate prospective sources of biomass independent of any particular bioenergy policy.

BT16 aims to estimate how much biomass could be supplied from current agriculture and forestland in the conterminous United States under supply constraints that limit typical LUC concerns, such as the loss of forests due to cropland expansion. U.S. forestland area and U.S. cropland area are held constant in all scenarios. No land is allowed to transition from forestland to cropland under the simulations. Furthermore, all USDA Conservation Reserve Program (CRP) lands are excluded from biomass production (see *BT16* volume 1, chapter 4). Assumptions and constraints applied in *BT16* scenarios mitigate potential market-mediated, global LUC effects, such as potential impacts on forests outside the United States (see chapter 2), and determine land allocation among crops and land cover. Understanding how these model specifications influence land allocation is relevant for LUC estimates and for the interpretation of environmental effects. In summary, the *BT16* scenarios illustrate future biomass potential from the agricultural and forestland bases as of 2015 and hold those areas constant for each simulation through 2040.

3.2.2 Concepts and Definitions Relevant to LUC

The state of the art for LUC analysis reflects both operational and conceptual limitations associated with terms, definitions, and associated land classifications used for analysis. Operationally, key terms used widely in the LUC and ILUC literature are often poorly defined, as many have acknowledged in the literature (e.g., Dale and Kline 2013a; ISO 2015; Kline, Oladosu, et al. 2011; Valin et al. 2015; Warner et al. 2014). Conceptually, LUC estimates from models are limited by reliance on assumptions ranging from initial land classifications and attributes (including exclusivity of “use”) to the assumed causal drivers for transitions between classes (Efroymson et al. 2016). Large uncertainties in basic land cover classifications are well documented (e.g., Congalton et al. 2014; Kline, Parish, et al. 2011; Feddema et al. 2005; Emery et al. 2017). The classification uncertainties increase when land “use” is inferred from land cover classes (Lambin, Geist, and Lepers 2003), and uncertainties are inherently far greater still whenever an analysis attempts to quantify “change” (O’Hare et al. 2010; Dale and Kline 2013a; Dunn et al. 2017). Even more controversial are assumptions about causal drivers of LUC, such as the interaction of temporary price changes in commodity markets with many other known causal factors of deforestation (Efroymson et al. 2016; Aoun, Gabrielle, and Gagnepain 2013; Kline et al. 2016).

Text Box 3.1 | *BT16* Land Terms and Major Crops Relevant to LUC

Key terms are defined in the glossary. The terms "biomass" and "potential biomass supply" are used without assumptions about end use. This is in contrast to many biofuel LUC assessments that estimate effects of a policy or production level specified for bioenergy. In this chapter, the term "bioenergy" is used in examples that aim to make the discussion relevant to U.S. Department of Energy Bioenergy Technologies Office stakeholders. Moreover, scenarios in 2040 involve biomass "energy crops," so named because they are likely to be used for energy purposes.

Agricultural land can be classified as annual crops versus perennial cover, or as biomass (energy) crops versus traditional (commodity) crops. For our calculations of change in land cover and management, idle land and Conservation Reserve Program (see glossary) lands are excluded. Traditional crops, such as corn and wheat, can supply stover or straw (biomass); however, these are not energy crops as defined by the U.S. Department of Agriculture (USDA) because their primary end uses are not for bioenergy. Agriculture simulations are based on the Policy Analysis System model (see chapter 2) using the following USDA major crops (parenthetical values next to each crop indicate millions of acres in 2015, the initial simulation year of the agricultural baseline): corn (88), soybeans (84), hay (58), wheat (all types, 56), cotton (10), grain sorghum (7), barley (3), oats (3), and rice (3). Forest-sector simulations are based on the Forest Sustainable and Economic Analysis Model (see chapter 2) to estimate potential supplies based on timberlands in the United States.

Every analysis that attempts to consider LUC is a product of underlying input data and assumptions, including how land classes and land use are defined (Dale and Kline 2013a; Woods et al. 2015). *BT16* is no exception, although the goal of volume 1 was to estimate potential sustainable supplies rather than to perform an LUC analysis. *BT16* focuses on biomass potential within the major land classes—agriculture and forestry—in the United States and builds on the best available USDA data sets for these two sectors. *BT16* biomass potential is estimated under constraints that do not permit net changes in the land base over time for primary uses (e.g., forest to cropland) but rather involve changes in specified management over time on existing agriculture and forest domains. This makes *BT16* distinct from other studies that attempt to define and parameterize land classes and to differentiate the services provided to society over space and time according to the classification system utilized. Models attempting to estimate LUC simplify data out of necessity, for example, by aggregating dynamic, heterogeneous uses into single classes for analysis (e.g., crop, pasture, forest, or urban). Relying on simplified land classes to assess LUC and generalizing characteristics of each class can be misleading and detracts from science-based assessment and communication of verifiable impacts.

3.2.3 LUC and Biomass from Forestland

See chapter 2 for a description of methods and assumptions applied to estimate potential biomass supplies from the forestry sector. The potential for the most significant LUC drivers associated with forestry biomass (e.g., loss of natural forest) is excluded from *BT16* by design because the Forest Sustainable and Economic Analysis Model (1) aims to assure that demands for conventional wood products were met, in addition to those for biomass; (2) assumes no changes in areas for total timberland, plantations, and natural forest management lands; and (3) incorporates supply constraints reflecting considerations, such as no new road building and limits or exclusions for biomass removals depending on terrain slope. As with agriculture lands, if less-restrictive assumptions are applied, larger potential biomass supplies could be simulated, but additional environmental issues would also be expected to arise.

Furthermore, *BT16* does not consider the fact that some historic cropland is in transition to become forest due to afforestation incentives provided under the CRP and similar programs. Because *BT16* scenarios aim for supply potential that reflects some sustainability principles, all CRP lands were reserved and excluded from consideration in scenarios.

Thus, the estimates of biomass from the forestry sector are meant to be conservative and avoid significant LUC concerns. Potential effects of alternative forest management approaches on the existing forestland, (e.g., water quality, habitat for selected species) are discussed in other chapters of *BT16* volume 2. The remainder of this chapter focuses on the changes simulated on agriculture land.

One LUC effect relevant to forest cover is the increasing use of cropland for short-rotation woody crops (SRWCs). For the purposes of this analysis, these are treated as changes in management practices on existing agricultural lands because, after a short rotation, the lands could rotate back into other agricultural uses. For example, as shown in table 3.1, by 2040 in HH3 case, 11 million acres of cropland are planted in SRWCs that can be coppiced (e.g., willow, eucalyptus), and an additional 13 million acres of cropland are planted in other SRWCs (e.g., poplar, pine). These changes in land management are discussed separately as one type of LUC within the agriculture sector.

3.3 Indicators to Capture LUC Effects

To understand environmental effects of biomass production on land, clearly defined indicators and units are required to characterize and measure changes over space and time (McBride et al. 2011). The broad definition of LUC is nearly impossible to apply with consistency because any action or inaction of humans that potentially impacts land could be described as LUC. Furthermore, major changes in land qualities can occur within a forest or agriculture landscape without reaching a specified threshold for a defined change in cover class (a common proxy for LUC in modeling), such as forest/pasture or pasture/cropland. Therefore, specific indicators that permit consistent measurement of pertinent characteristics (i.e., of effects that stakeholders care about) are essential. Examples of indicators relevant to LUC include carbon stocks and net primary productivity or biomass yield. While these are not measures of LUC per se, they are examples of indicators that capture the effects of different land management practices and production systems. Soil carbon is discussed in chapter 4. This chapter reviews how the amount of land managed for annual crops, pasture, and other perennial crops varies under different scenarios.

Two important conclusions about the use of LUC information to estimate environmental effects can be drawn from extensive literature and field work (e.g., Gasparatos et al. 2017): (1) what matters is what really changes rather than general land labels used for land classification, and (2) different management practices within a defined land class can lead to significant changes over time in measured values for environmental indicators (e.g., carbon stocks, biodiversity, water quality). For example, Fargione et al. (2008) illustrate how the estimation of effects of bioenergy on carbon stocks depends on many factors independent of the basic land class used for LUC assessment. Forests range from degraded woodlands in dry environments to old-growth tropical forests. Carbon stocks and accumulation rates can vary by orders of magnitude while the land remains labeled as "forest." The same holds true in agricultural systems where, in addition to soils, weather, and prior use, the carbon stocks and sequestration rates depend on factors such as the type, timing and frequency of site preparation, fertilization, harvest, and soil tillage (e.g., specific equipment used, type and depth of tillage, area disturbed).

Biomass supplies in *BT16* are sourced from the utilization of residues and coproducts from forestry and agriculture (e.g., timber thinning, corn stover), which are recognized in the literature to involve negligible potential for direct or indirect LUC (e.g., Fargione et al. 2008); biomass supplies in *BT16* are also sourced through modifications of agricultural management practices, which influence environmental indicators over time. The incremental increases in biomass production under *BT16* complement rather than displace current production. The assumptions and approach underlying *BT16* reflect historical U.S. trends to improve land management efficiency in response to new and increasing biomass production. From 1984–2011, for example, agricultural output increased by 1.5% per year while total area of land used for agriculture decreased by more than 0.5% per year, on average (Wang et al. 2015).

LUC-related effects that are estimated using indicators are a product of comparing *BT16* scenarios (BC1 in 2017 and 2040 and HH3 in 2040) to each other and to the agricultural baseline in 2015, 2017, and 2040. Estimated effects always depend on the reference case, and many alternative future scenarios are possible (appendix 3-A). While *BT16* scenarios exclude LUC between forestry and agriculture uses by design, and also exclude the use of CRP land for biomass crops, the scenarios involve changes in land management, crop type, and crop acreages within specific portions of the remaining agricultural landscape. The magnitude and implications of these changes are discussed below.

3.4 LUC and Agricultural Land: Cropland and Pasture

The allocation of land among agricultural uses, including conventional crops, energy crops, and perennial cover, is presented in table 3.1.

Table 3.1 | Crop Type, Cover Classification (Annual, Perennial, Idle), and Total Area in the Agricultural Baseline and in the *BT16* Scenarios Considered in Volume 2

		Agricultural Baseline 2015	Agricultural Baseline 2017	BC1 2017	Agricultural Baseline 2040	BC1 2040	HH3 2040
Crop	**Cover Class**	**Millions of Acres**					
Barley	Annual	3.5	3.2	3.2	2.9	2.8	2.7
Corn	Annual	88	90	90	89	85	74
Cotton	Annual	9.8	9.8	9.8	11	8.6	7.7
Oats	Annual	3.0	2.5	2.5	2.4	2.1	1.9
Rice	Annual	2.9	2.9	2.9	3.1	3.0	2.8
Sorghum	Annual	7.5	7.4	7.4	7.0	6.2	5.8
Soybeans	Annual	84	78	78	77	66	60
Wheat	Annual	56	53	53	54	46	42
Total Major Crops		**255**	**246**	**246**	**246**	**219**	**197**
Hay	Perennial	58	57	57	57	56	56
Idle	Idle	13	22	22	23	23	23
Subtotal other cropland (idle, hay)		71	79	79	80	79	79
Total Cropland excl. energy crops		**326**	**326**	**326**	**326**	**298**	**277**
Total Pasture excl. energy crops		**446**	**446**	**446**	**446**	**409**	**407**
Bio-sorghum	Annual					1.7	2.3
Coppice wood	Perennial					5.0	11
Energy cane	Perennial					0.0	0.3
Miscanthus	Perennial					21	37
Non-coppice	Perennial					9.3	13
Switchgrass	Perennial					28	24
Total Energy Crops		**0**	**0**	**0**	**0**	**64**	**88**
	Perennial	504	504	504	504	528	549
	Annual	255	246	246	245	221	200
	Idle	13	22	22	23	23	23
Total Agricultural Land Considered in *BT16*		**772**	**772**	**772**	**772**	**772**	**772**

		Agricultural Baseline 2015	Agricultural Baseline 2017	BC1 2017	Agricultural Baseline 2040	BC1 2040	HH3 2040
Crop	Cover Class	Millions of Acres					
Additional U.S. Agricultural Land:							
Reserved CRP	Idle	*27*	*27*	*27*	*27*	*27*	*27*
Other farmland (woodlands, built up, roads, waste land, other)		*110*	*110*	*110*	*110*	*110*	*110*
Total farmland incl. CRP reserve		**909**	**909**	**909**	**909**	**909**	**909**

Table 3.1 summarizes total land allocation by class for the agricultural baseline in 2015, 2017, and 2040 to allow comparison with allocations under the *BT16* scenarios analyzed in this volume. The land allocation data are consistent with U.S. farmland classifications as defined by the USDA National Agricultural Statistics Service (NASS) (USDA NASS 2014) and as reported in the USDA baseline projections (USDA 2015a), with pasture categories combined in table 3.1. For comparison, note that the most recent Census of Agriculture (USDA NASS 2014) identified 914 million acres of total farmland, with 390 million in cropland (includes irrigated and cropland pasture); 415 million in other permanent pasture and range; 77 million in woodlands and grazed woodlands; and another 33 million acres in built-up areas, wasteland, or other non-productive uses of farmland. The smaller area considered in *BT16* compared to the total USDA census (USDA NASS 2014) reflects reductions in cropland area based on the USDA baseline projections (USDA 2015a) and the exclusion of farmland outside the conterminous United States in *BT16*. The bottom rows of table 3.1 illustrate that 137 million acres were excluded from consideration in *BT16* simulations before the analysis began to apply constraints: 27 million acres of cropland in CRP were excluded, along with 110 million acres in built-up areas, wasteland, or other non-productive uses of farmland.

The differences in land allocation and management observed under different years and scenarios in table 3.1 include (1) increases in idle cropland area in all scenarios compared to the agricultural baseline in 2015; (2) decreases in conventional crop area in all scenarios compared to 2015; (3) decreases in pastureland area in 2040 *BT16* biomass scenarios compared to other scenarios; and (4) net increases in perennial land cover under *BT16* biomass scenarios in 2040.

Idle cropland includes land allowed to go fallow for a period as part of normal rotations with other crops, as well as land available to support crops in response to market signals (see glossary). Because we do not assume idle cropland is managed exclusively as perennial or annual cover, idle remains a separate land class. For *BT16* scenarios, 27 million acres of CRP are held constant and excluded from eligibility for any other use. By USDA's definition, CRP falls into the "idle cropland" class. Thus, including the reserved CRP lands, there would be 50 million acres of idle cropland in the 2040 scenarios. LUC-related issues associated with different types of agricultural land management are discussed below.

3.4.1 Changes in Agricultural Land Management under *BT16* Scenarios

The primary types of LUC associated with *BT16* supply scenarios involve changes in land management practices on land that has been in use for conventional crops and pasture. The most significant net LUC from 2017 to 2040 is the transition from conventional annual crops to perennial land management systems, a transition that accelerates with increasing demand for biomass. The area estimated to be managed as perennial cover in 2040 is 45 million acres greater under the HH3 scenario than the area of perennial cover in the 2015 agricultural baseline or the 2040 agricultural baseline (see chapter 2) without new biomass demand. The geospatial distribution of the net change from annual to perennial cover is illustrated in figure 3.1 for BC1 2040 (reflecting a 24 million–acre expansion) and figure 3.2 for HH3 2040. The darker colors in figures 3.1 and 3.2 represent counties where perennial cover increased by 25%–40%. The light grey shading over most counties in the United States indicates that change was negligible or small (less than +/-5%). No counties have loss of perennial cover greater than 5% in 2040 under *BT16* scenarios. Larger increases in percentage of perennial cover occur on agriculture land in areas where simulated returns from conventional crops are not as competitive with energy crops under the conditions defined in the base case scenario, BC1 2040.

The total land in perennial cover is about the same in the following scenarios: the agricultural baseline in 2015 and in 2017, the BC1 scenario in 2017, and the agricultural baseline in 2040 (table 3.1). However, as with other land categories, while the total area in a class may appear to be constant across the nation over several years, this lack of net change can mask significant shifts in locations of perennial cover as well as net changes in any given county. In general, we observe that perennial cover increases incrementally in response to assumed biomass markets under *BT16* scenarios.

The net expansion of perennial cover is significant in terms of land area (i.e., 24 to 45 million acres) but modest when considered relative to the overall agricultural landscape considered in the scenarios (772 million acres), as shown in figure 3.3. The expansion of idle cropland as a separate category in each scenario relative to the 2015 agricultural baseline is also illustrated in figure 3.3.

Figure 3.3 illustrates how total agricultural area managed as annual crops is estimated to decline and transition to perennial cover when the allocation of land in the agricultural baseline in 2015 is compared to land allocations in 2040 under (1) the agricultural baseline projection to 2040 without biomass demand; (2) BC1; and (3) HH3. Figure 3.3 illustrates the progressively increasing amounts of land that transition on net from annual crops to perennial cover under these scenarios. The figure also illustrates that these shifts are small relative to the total agriculture land area considered in the analyses (772 million acres). Finally, note that in addition to the 27 million acres of CRP land reserved outside the analysis, the simulations include 23 million acres of idle land in each future scenario. The idle land provides a potential cushion, allowing response to unexpected increases in demand for crops or biomass in other sectors.

Figure 3.1 | Geospatial distribution of changes in perennial cover under the base case (BC1) scenario[1]

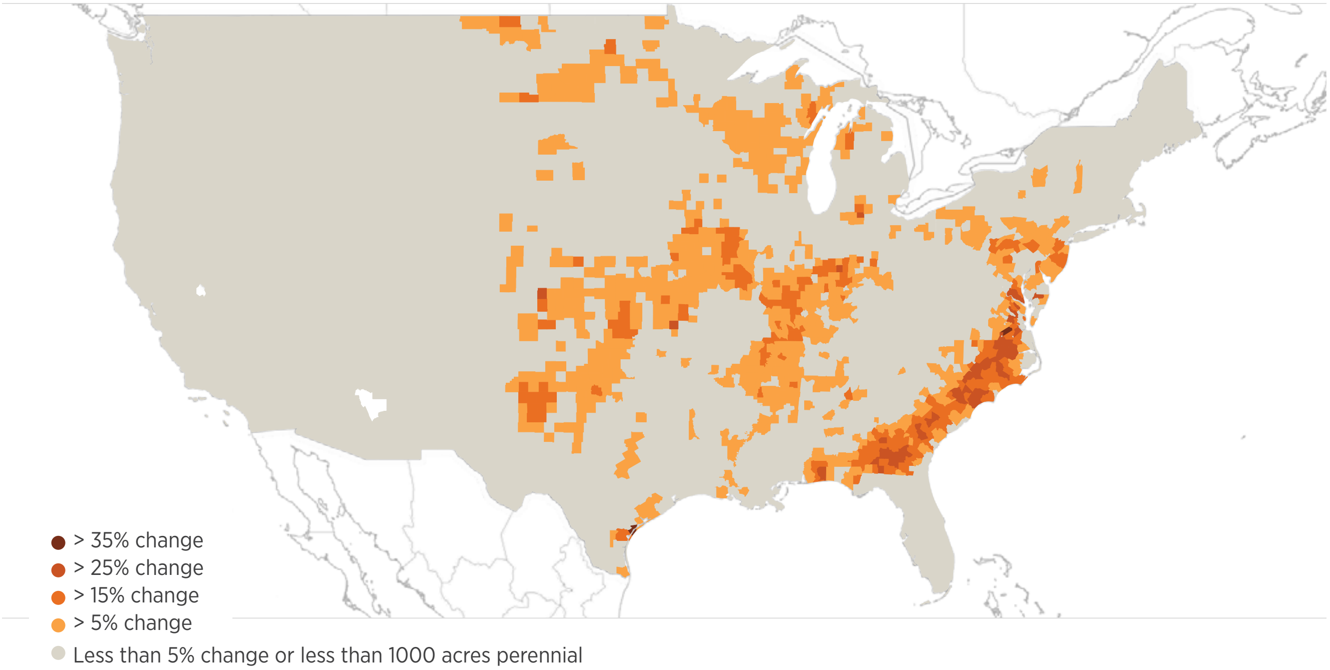
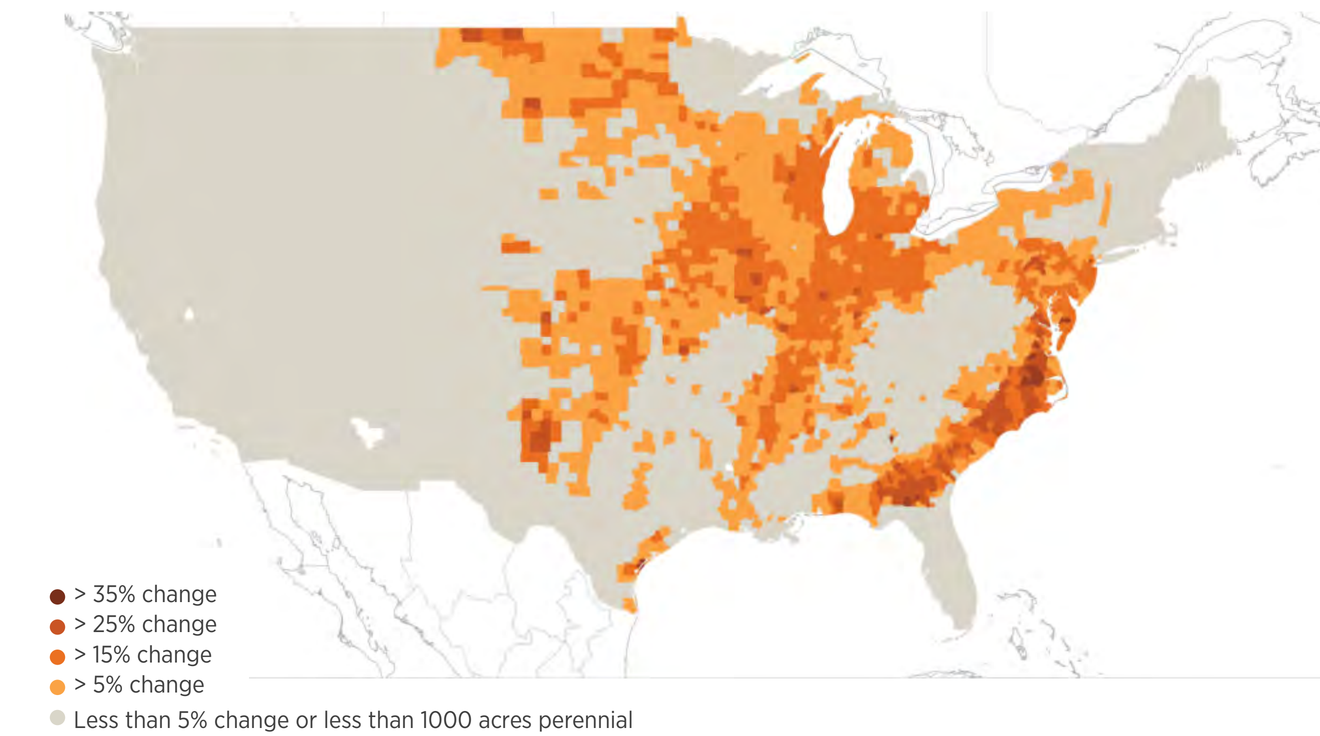
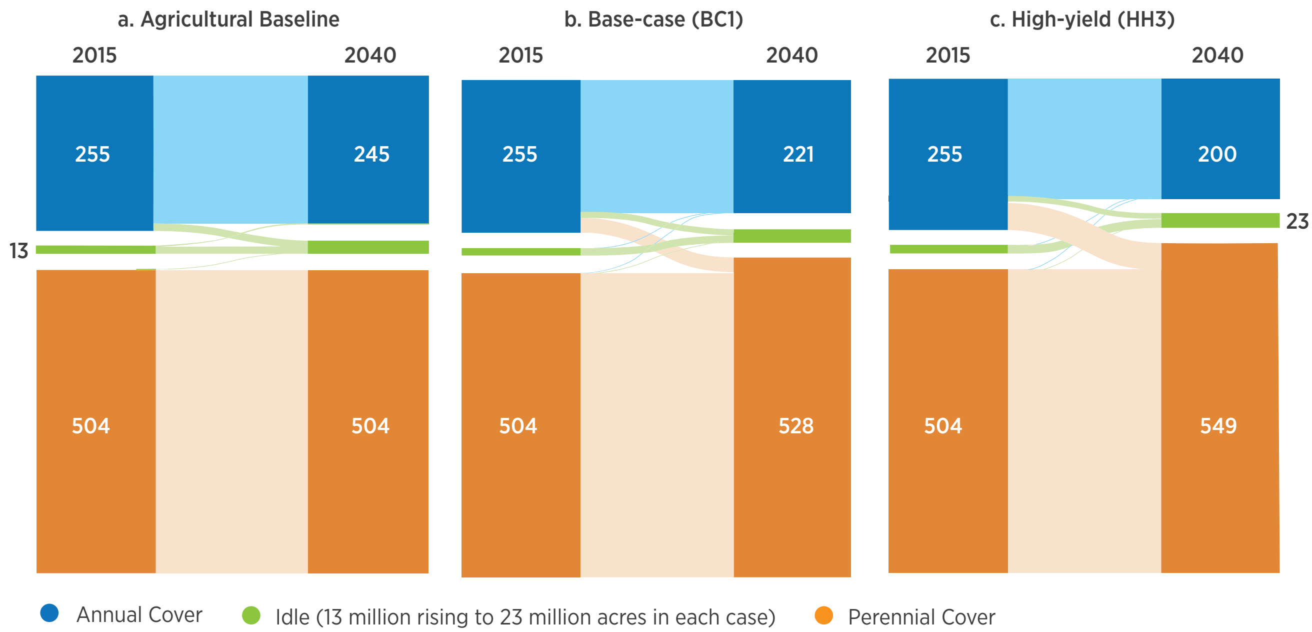


Figure 3.2 | Geospatial distribution of changes in perennial cover under the 3% annual yield increase (HH3) scenario[1]



[1] Change in perennial cover by county is the difference between the percentage of total agricultural acres (cropland + pasture) managed as perennial cover in *BT16* 2040 scenarios (BC1 or HH3) and the percentage managed as perennial cover in the 2040 agricultural baseline without new biomass production.

Figure 3.3 | Agricultural land (millions of acres) managed as annual crops, perennial cover, or idle cropland in 2015 and 2040 as estimated under (a) the agricultural baseline; (b) base case scenario (BC1); and (c) high-yield scenario (HH3)



In addition to the gradual transition from row crops to perennial biomass crops illustrated in figure 3.1, changes in management occur on pasture. By 2040, 37–39 million acres, or about 8% of total pasture area in the 2015 extended agricultural baseline, would undergo changes in management to produce energy crops. This area is not segregated in figures 3.1–3.3, which compare annual crops to perennial cover, because both pastureland and the energy crops illustrated are classified as perennial cover.

The changes from annual to perennial land management affect 3% of total agricultural land under BC1 and about 6% under HH3, and the transitions occur gradually between 2015 and 2040. There are also gradual changes in the management of pastures, with about 8% of total pastureland area in the 2015 agricultural baseline shifting to management for energy crops by 2040. Fencing and pasture rotation are management practices that are assumed to intensify production on another 56–58 million acres of pasture (13% of total pastureland) to maintain forage output in tandem with increasing energy crop production. Percentages here are expressed relative to the total areas of cropland and pastureland in the 2015 agricultural baseline and the projected agricultural baseline in 2040 (table 3.1). As with any model, input parameters and assumptions regarding land classes, land area available for different uses, and productivity influence how land is allocated among traditional and energy crops over time. Assumed increases over time in the productivity of pasture (see *BT16* volume 1, section 4.8.5), yields for conventional and energy crops, and simulated prices of biomass are the drivers for the modeling results allowing for increased biomass feedstock production within the current (2015) agricultural landscape.

3.4.2 Land Input Assumptions Drive LUC Estimates

The input values for land parameters and constraints relevant to LUC are described in chapter 2. Key parameters impacting LUC include the initial land base and land classes considered, and the annual rates of expansion allowed. For example, energy crop acreage in a county is limited to 5% of permanent pasture, 20% of cropland pasture, and 10% of cropland. These percentages reflect an estimate of barriers and opportunities associated with the adoption of new crops.

Before applying any constraints, an initial agricultural land base of 772 million acres was considered for *BT16* biomass supply scenarios modeled in the Policy Analysis System (POLYSYS) (table 3.1). This acreage includes eight major row crops plus cultivated hay on cropland, for a total of 313 million acres of cropland, plus 446 million acres of pasture. For *BT16*, pastureland includes 11 million acres classified as cropland pasture, plus other pasture and rangeland (figure 3.4). The definition of each class is based on the USDA 2012 Census of Agriculture (USDA NASS 2014; see glossary for full definitions), and the acreages in table 3.1 for cropland classes were based on average values reported over 4 years in recent NASS statistics (see appendix C of *BT16* volume 1).

When interpreting any description of LUC, it is essential to understand that "change" is always expressed with respect to the comparison of two selected values. Thus, LUC associated with *BT16* varies depending on whether it is a product of comparing a given simulation (1) to another simulation (e.g., BC1 2040 versus HH3 2040), (2) to the agricultural baseline in 2015 or 2040 (table 3.2), (3) to different years within a given scenario (e.g., BC1 2017 versus BC1 2040), or (4) to some other reference case. Changes occur in the USDA baseline projections (USDA 2015a) and in the projected agricultural baseline simulated in POLYSYS, independent of assumed new biomass demand. For example, in the agricultural baseline scenario, the area planted in major crops is estimated to decrease by about 10 million acres while overall outputs increase through improved productivity.

In *BT16* biomass scenarios, the area of agricultural land managed for annual crops in BC1 2040 is 25 million acres less that the quantity simulated in BC1 2017; and from BC1 2017 to HH3 2040, the decline is 46 million acres. Similar differences are observed if BC1 2040 and HH3 2040 are compared to the agricultural baseline in 2040 (table 3.1). However, the reduction in land area managed for annual crops is different if these scenarios are compared to the 2015 agricultural baseline (table 3.2), due primarily to decreased demand for commodity crops between 2015 and 2017. Most reductions in annual crop acreage over time can be accounted for by increased yields and decreased area planted in conventional crops (primarily soy beans, corn, and wheat; see table 3.1). As the area managed for conventional annual crops declines, the area managed as perennial cover increases along with increasing energy crop production.

Table 3.2 highlights the net changes in land managed as annual crops, idle, and perennial cover when the 2015 agricultural baseline is compared to scenarios for 2040. Table 3.3 shows the allocation of 2015 cropland acres to specific biomass crops in 2040 under the two scenarios (BC1 and HH3). Table 3.4 illustrates the allocation of 2015 pastureland to biomass crops under the two scenarios.

Table 3.2 | Total Agricultural Land Allocation by Scenario and Class: Annual Crop, Perennial Cover, or Idle Cropland (millions of acres). Differences in 2040 Compared to the 2015 Agricultural Baseline Are Noted in Parentheses. (The sum of some columns is affected by rounding.)

Land Type	Agricultural Baseline 2015	Agricultural Baseline 2040	*BT16* BC1 2040	*BT16* HH3 2040
Millions of Acres				
Total land in annual crops	255	245 **(-10)**	221 **(-34)**	200 **(-55)**
Perennial cover	504	504	528 **(+24)**	549 **(+45)**
Idle cropland[a]	13	23 **(+10)**	23 **(+10)**	23 **(+10)**
Total	**772**	**772**	**772**	**772**

[a] Does not include CRP lands.

Table 3.3 | Land Allocation by Crop Type: Energy Crops on Cropland (millions of acres)

Crop Type and Land Cover Classification	BC1 2040	HH3 2040
Biomass sorghum	2	2
Total annually cultivated biomass crops	**2**	**2**
Switchgrass on cropland	7	8
Non-coppice SRWCs on cropland	5	9
Coppice SRWCs on cropland	2	8
Miscanthus on cropland	11	21
Energy cane on cropland	0	0
Total perennial biomass crops	**25**	**47**

Table 3.4 | Land Allocation by Crop Type: Energy Crops on Pasture Including Cropland Pasture (millions of acres)

Crop	BC1 2040	HH3 2040
Switchgrass	21	15
Non-coppice SRWCs	4	4
Coppice SRWCs	3	3
Miscanthus	10	16
Energy cane	0	0
Total	**37**	**39**

3.4.3 Agricultural Land Allocated to Biomass Crops

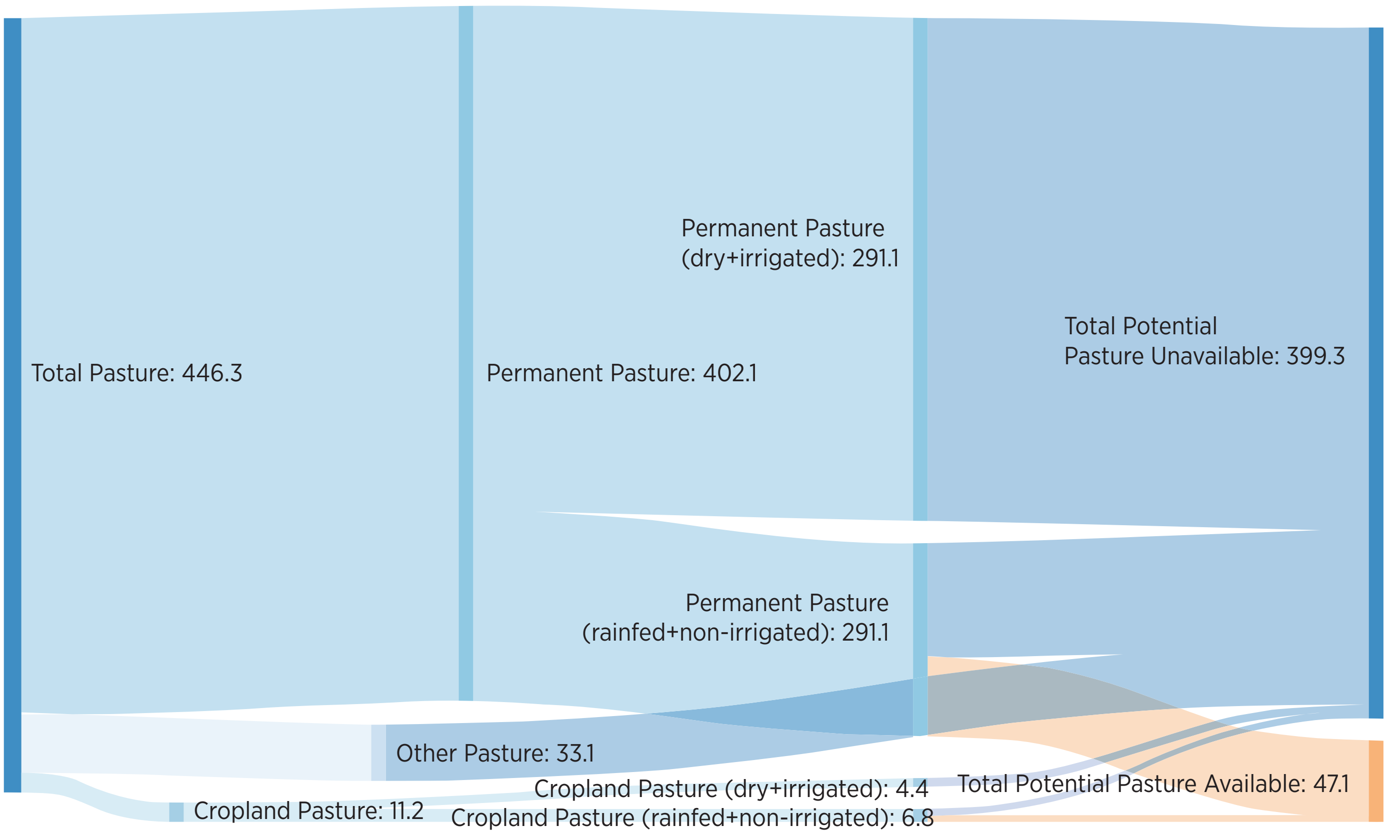
After all constraints used for *BT16* simulations are in place, the total agricultural land area considered within the POLYSYS model runs (e.g., land "eligible" for potential energy crop production) is about 243 million acres (196 million cropland + 47 million pastureland). The POLYSYS simulations considered the competitiveness of energy crops compared to other potential crops on only this subset (31%) of the initial agricultural land base of 772 million acres. Recall that the 772 million–acre initial land base already excluded 137 million acres of farmland, including CRP, from the analysis (table 3.1). Under the biomass scenarios discussed in this volume, 64 million acres (BC1) or 88 million acres (HH3) are allocated to be managed as energy crops by 2040, representing 8% (BC1) or 11% (HH3) of the initial land base, respectively, and about one-third of the area identified as being potentially eligible for energy crops under the constraints and assumptions used for *BT16* simulations.

In conclusion, the energy crop land allocation in 2040 (64 or 88 million acres for BC1 and HH3 scenarios, respectively) represents less than 10% of total private farmland in the conterminous United States (USDA 2014). Under *BT16* scenarios, yield improvements and pasture intensification gradually allow for increasing quantities of biomass production without significantly displacing output required to meet future projected demand in other sectors. These results reflect assumptions for crop yield improvements that meet future demands for food, feed, and fiber on less land, and are consistent with a continuation of historical agricultural land productivity trends (Wang et al. 2015).

3.4.3.1 LUC Implications of BT16 Constraints for Energy Crops on Pastureland

In addition to the limit on annual rates of expansion in *BT16* scenarios, energy crops are not allowed on irrigated pasture, as this is assumed to be retained to supply specialized local markets. Likewise, energy crops are excluded from dry rangelands or pasture with less than 25 inches of precipitation per year. The constraints for rain-fed pastureland reduce the area eligible for planting energy crops in any year to a defined land base of 118 million acres (see *BT16* volume 1, appendix C, figure C-2). Further constraints are applied such that in any one county, energy crops may not exceed 40% of the eligible land for pasture over the simulation period (i.e., 2017–2040) because of the requirement for management-intensive grazing to maintain forage output (*BT16* volume 1, appendix C). When all constraints are applied to the baseline pasture area of 446 million acres, the maximum eligible pastureland for energy crops represents about 47 million acres, or 11% of total pastureland, as shown in figure 3.4. Assumptions regarding pasture management intensification to meet projected future demand for forage (see chapter 2) have implications for modeling results.

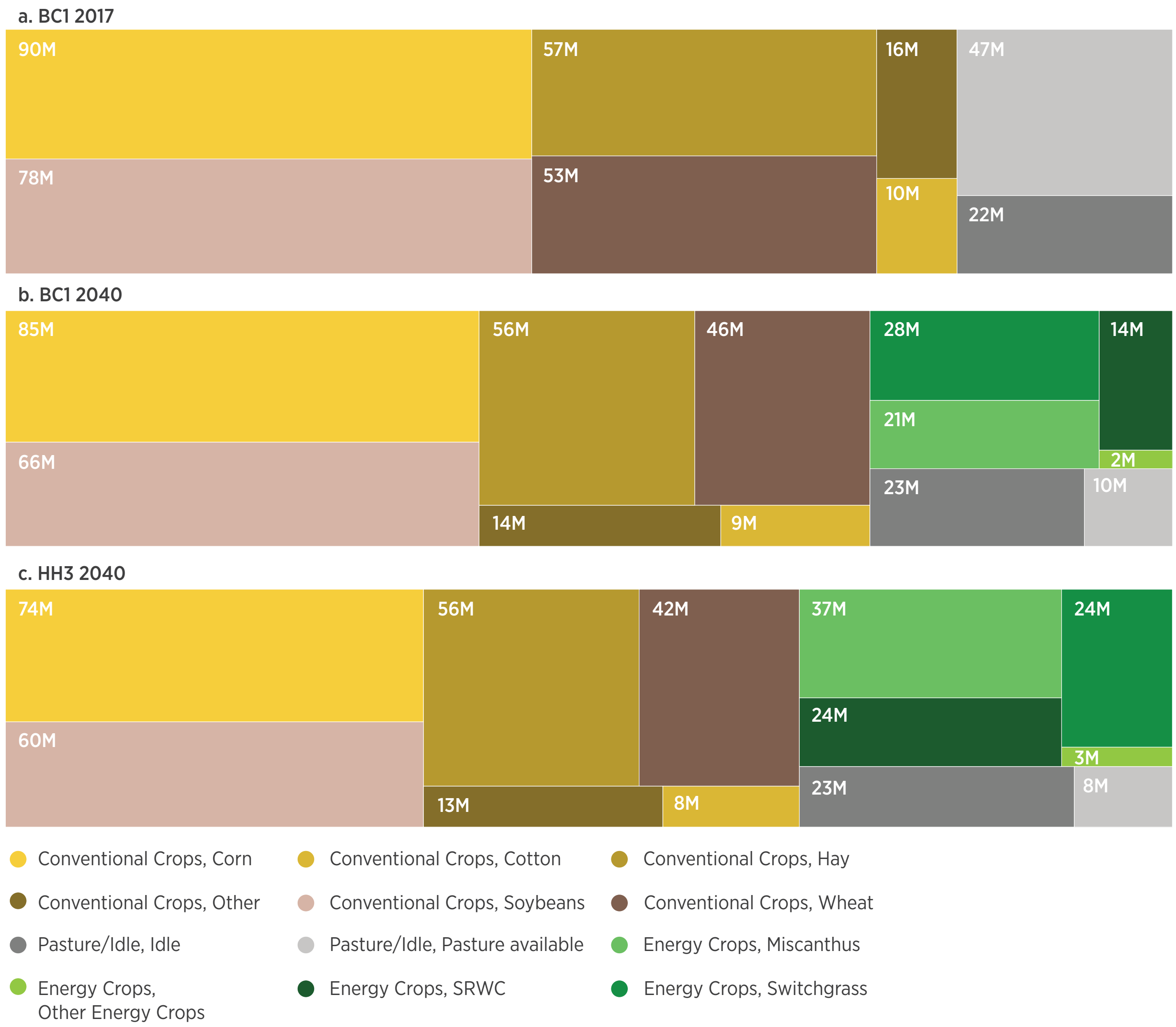
Figure 3.4 | Total U.S. pastureland area and subset eligible for biomass crops (millions of acres). The constraints applied in BT16 reduce the area of pasture eligible for energy crops from a total of 446 million acres to 47 million acres (applicable to all scenarios, in all years).



3.4.3.2 Energy Crops on Cropland

The cumulative effect of the *BT16* constraints for expansion of energy crops is that the maximum amount of cropland potentially eligible for energy crops by 2040 represents about half of the cropland area considered in the 2015 agricultural baseline and in BC1 2017. The cumulative expansion in 2040 of 27 million acres of energy crops in BC1 represents only about 8% of total 2015 cropland area (326 million acres) and 15% of the eligible cropland area under the constraints used in *BT16* (27 million acres out of 181 million eligible). The high-yield scenario (HH3) results in a cumulative planting of energy crops by 2040 on 49 million acres of cropland, or about 15% of the 2015 agricultural baseline cropland area. As illustrated in figure 3.5, the allocation of cropland to row crops declines over time in *BT16* scenarios in association with increasing biomass production. A gradual reduction in U.S. cropland area is consistent with historic trends and with the agricultural baseline projection that simulated a 10 million–acre reduction in cropland area from 2015 to 2017 (table 3.2). In part, the reduced area of cropland reflects the fact that total factor productivity of U.S. agriculture has been increasing while land as an input has been declining (Wang et al. 2015).

Figure 3.5 | Allocation of cropland (326 million acres) and pastureland potentially eligible for energy crops (47 million acres) under BT16 simulations (millions of acres): (a) BC1 2017; (b) BC1 2040; and (c) HH3 2040. Each Figure (a–c) represents allocations across 373 million acres.



3.5 LUC Modeling

Models are used to estimate LUC by comparing areas for a defined land class (e.g., forest or agriculture) under two simulations. If assessing effects of bioenergy policy, LUC studies typically involve one simulation in which biofuel production increases and a reference case in which it does not. The differences in the area of each land class that are generated by these two scenarios are presented as LUC.

In most studies, the model outputs do not distinguish between direct and indirect LUC (Valin et al. 2015; Dale and Kline 2013b), but these labels are sometimes applied. Differentiation of ILUC from direct LUC is typically based on assumptions about the baseline or reference land use and system boundaries.

For example, in the case of economic models examining U.S. biofuel policies, LUC that occurs outside the United States is commonly labeled "indirect." However, a study focusing on biomass production in a single U.S. state may consider LUC projected outside of that particular state to be indirect. Other studies attempt to allocate land areas based on an assumed initial land cover compared with a simulated land cover, wherein any land used for biomass production that is modeled to occur on non-agricultural land is considered a "direct LUC," and the sum of all other changes in land use is assumed to be indirect.

The potential global impacts of an expansion of biomass production in the United States depend on many factors not analyzed under *BT16* scenarios. Reasonable assumptions about increasing biomass production could generate estimates that vary widely not only in terms of magnitude, but also in terms of direction of the effects—particularly in terms of whether forestland is expected to expand or contract in response to policies associated with biomass production (see appendix 3-A; Kline et al. 2009).

3.5.1 How *BT16* Relates to Concerns about ILUC

BT16 is not designed to address questions about LUC, but understanding how bioenergy policies actually interact with other policies, markets, and disturbances (such as fire) is critical for more accurate LUC assessment (Kline and Dale 2008). A review of the conceptual basis for LUC modeling can illustrate how common concerns about indirect effects are managed in *BT16* with a focus on ILUC modeling. The two main forces assumed to drive ILUC are (1) price mechanisms and (2) crop displacement:

- Under the price mechanism, ILUC can occur if (1) biomass production causes higher prices for other commodities; (2) these higher prices are transmitted to markets in other countries; and (3) the response in those countries to the higher prices is to clear more land for growing those crops than would have been cleared otherwise.
- Under the displacement mechanism, ILUC can occur if (1) biomass production displaces local output of a crop; (2) the reduced output of the crop is replaced by growing more of the crop elsewhere; and (3) growing more of the crop elsewhere requires the clearing of new agricultural land.

Both of these mechanisms require causal pathways (a➔b➔c…) in which the absence of any one step would block the effect (Efroymson et al. 2016). For example, under the first mechanism, if higher prices are not transmitted to other nations, or if higher prices cause intensification rather than new land clearing, then the pathway is interrupted and the assumed effect would be blocked. Empirical evidence suggests that such conditions may create breaks in the causal chain assumed for the price mechanism. Rather than testing for the existence of these mechanisms, economic models for ILUC typically begin by assuming the mechanisms are in place and then seek to assess effects of a "shock" in biofuel demand to generate ILUC estimates.

Regarding the two basic mechanisms above, *BT16* constraints were applied to minimize these "market-mediated" effects. For the price mechanism, it is estimated under the *BT16* supply scenarios that commodity prices could be higher or lower depending on the rate of yield growth assumed (see *BT16* volume 1, appendix C). Regardless of whether prices are projected to decline or rise, the price changes associated with biomass production are small relative to other drivers of change in food commodity prices (Kline et al. 2016). More detailed analysis of the impacts of potentially higher or lower prices (depending on the *BT16* scenario) on global markets and land use is beyond the scope of the analysis for this study.

Regarding potential crop displacement mechanisms, *BT16* simulations were based on scenarios that allowed conventional commodity outputs to increase over time and fulfill increasing demand. The assumed incremental expansion of energy crops in tandem with increasing productivity reduces potential mechanisms theorized to cause ILUC. Under the HH3 scenario, in which energy crops occupy the greatest area (88 million acres) by 2040, the land in row crops is simulated to decline by 56 million acres while total output continues to grow each year to meet or exceed demands projected under the 2040 agricultural baseline (see *BT16* volume 1, appendix C). While corn stover is an important source of biomass in BC1 and HH3 simulations, acreage in corn and conventional crops overall decline in biomass production scenarios. These *BT16* results are consistent with decadal trends, which show a small but steady reduction in conventional crop acreage over time. This study focuses on potential new cellulosic biomass supplies building from a 2015 agricultural baseline; it does not consider changes to current conventional biofuel programs (e.g., corn starch ethanol and soy-based biodiesel production).

The *BT16* constraints aim to avoid biomass production locations, management practices, and economic competitions that would represent likely environmental concerns (see chapters 1 and 2). This approach is consistent with other studies that investigate options to produce biomass while preventing or mitigating LUC and other environmental impacts (e.g., Brinkman et al. 2015; RSB 2015; Gerssen-Gondelach et al. 2016; Gerssen-Gondelach, Wicke, and Faaij 2015; Beringer, Lucht, and Schaphoff 2011; Schubert et al. 2009; Wicke et al. 2015). The assumptions applied to estimate potential biomass supplies that could be produced from current agricultural and forestlands without changing the areas now used for those broad categories are likely to be as accurate as (if not better than) alternative assumptions that attempt to predict how these land areas will change in the future (e.g., see Buchholz et al. 2014). Even though no one expects all current forestland acres to remain the same over the next 25 years, the *BT16* assumption that net area does not change is defensible given the purpose of the assessment and historical trends (discussed below). No net change in area is a common *ceteris paribus* (all else held constant) modeling assumption that facilitates simulations by avoiding additional complications. Furthermore, the U.S. Renewable Fuel Standard (H.R. 6 2007) only considers biomass used for fuels to be renewable if it is derived from land cleared or cultivated for agriculture or managed forests prior to 2007. For more discussion of the assumptions underlying the agricultural baseline and how *BT16* scenarios address projected future demand, see chapter 2 of *BT16* volume 2 and appendix C of *BT16* volume 1.

3.5.2 *BT16* Results in Context of Other LUC Studies

It is difficult to compare the *BT16* resource assessment to other studies designed to estimate LUC, as the questions asked and approaches applied are distinct. However, it can be enlightening to carefully review the input parameters and assumptions underlying each approach to determine what is driving the results of a given simulation of future biomass production. Assumptions and details behind *BT16* are carefully documented to support transparent analysis (see chapter 2).

Input data sets and assumptions are critical factors that determine LUC assessment results. The land class ontology, land areas and uses considered, and land rents assumed in a baseline are key factors, along with how spatially explicit land units are defined and how they are segregated or aggregated for analysis. These input specifications vary widely from study to study and are one of many sources of divergent LUC estimates. Further, the criteria and data used to differentiate land cover from land use, and to specify past productivity and potential future productivities at high resolution (not to mention current carbon stocks and rates of net sequestration or emissions), are rarely documented but are also critical to many LUC effects assessments.

Modelers acknowledge that ILUC estimates cannot be validated (NRC 2011; Valin et al. 2015; Babcock 2009). Calibrating estimates of ILUC attributed to biomass production is challenging because (1) the LUC is not defined in practical, consistent, and verifiable terms; (2) other confounding factors determine if and when observable changes, such as deforestation, occur around the world; (3) the processes involved are not singular events but rather reflect constant and ongoing incremental changes and dynamic cycles; and (4) to calibrate and validate models would require extensive and costly field analysis to support statistical analyses of all potential factors and support a defensible allocation of observed changes among countless causal agents (Efroymson et al. 2016; Valin et al. 2015; Kline et al. 2009). Even if all the data and statistical analyses could be completed, a reference case must be simulated in order to estimate "change," and therefore, modeling assumptions are a necessity (NRC 2011).

Given high uncertainty and limitations of LUC models (Plevin et al. 2015; Verburg, Neumann, and Nol 2011; Aoun, Gabrielle, and Gagnepain 2013; Souza et al. 2015; Hertel et al. 2010; NRC 2014), it is important to examine underlying assumptions and input variables that drive LUC results for any study in order to understand and interpret results. Indeed, many assumptions used in past LUC modeling for bioenergy have been found to be invalid (e.g., Babcock 2009; Kim and Dale 2011; Kline, Oladosu, et al. 2011; Dale and Kline 2013a), and there is little empirical evidence to support the types and magnitudes of LUC that have been projected (Langeveld et al. 2014; Babcock and Iqbal 2014; Oladosu et al. 2011). Recent research suggests that the state of science is inadequate to include ILUC in international standards (Zilberman et. al., 2010; ISO 2015; ASTM 2016). As stated in a policy analysis report by the National Research Council, the "range of estimates for GHG emissions from indirect land-use changes is wide…," but "GHG emissions from land-use changes cannot be ignored…results by definition carry the assumptions and inherent uncertainties in these models"; the report concludes that "[a]dditional research is needed to better understand the socioeconomic processes of land-use change and to integrate that process understanding into models" (NRC 2011). The caveat to carefully examine input specifications and assumptions is applicable to any analysis attempting to estimate impacts of future or alternative land management, including *BT16*. Comparing input data and assumptions helps put land allocations from *BT16* scenarios into a broader context of LUC analysis. See appendix 3-A for further discussion.

Estimating future LUC is difficult in part because of the controversies that surround analysis of past LUC. For example, some analyses begin by assuming that land in cropland subcategories—such as idle, hay, and cropland-pasture—are "non-agricultural" grassland in the baseline. It is then not surprising that these analyses identify large amounts of "grassland conversion" (e.g., Mladenoff et al. 2016; Wright and Wimberly 2013). However, based on USDA definitions, acres that such studies flag as "converted" are more accurately described as forming part of ongoing management and rotations on cropland because these lands were previously used and classified as cultivated cropland (USDA NASS 2014; Kline, Singh, and Dale 2013; Qin et al. 2016; Johnston 2014). Further, managing idle, hay, and cropland-pasture land subcategories in rotation with row crops may be a preferable strategy to achieve ecosystem benefits (such as soil conservation, reductions in pests and the need for herbicides and pesticides, and soil moisture conservation) and to efficiently achieve other goals within constraints dictated by local circumstances. Regardless, under *BT16*, idle and hay are considered part of the cropland class, which is consistent with USDA definitions.

There is often not a clear line to separate grassland from pasture, or pasture from other cropland (see appendix 3-A). Alternating or coproducing row crops with perennial crops over long rotations is one of the many management complexities that makes analysis of LUC difficult or misleading. Therefore, we recommend focusing on actual management practices and the specific effects of those practices on environmental indicators, rather than vague LUC labels for temporary changes in land management.

3.6 Discussion

In this section, we review how *BT16* simulations compare to historic LUC trends, discuss limitations and uncertainties inherent in LUC analysis, and propose some directions for future research. Because some type of LUC is constantly occurring practically everywhere that humans are present and because LUC involves multiple ongoing interactions rather than a singular event, modeling LUC is a challenge. Therefore, LUC assessments must begin by clearly defining the question to be addressed, the type of LUC of concern, and the data to be used, and then applying an approach appropriate for the situation.

In the case of *BT16*, given the constraints that prohibit net changes in total areas for forest and agriculture (and the reservation of 27 million acres for CRP within the agriculture land base), the LUC issues relate to estimated land management changes and how the management practices and locations associated with biomass production compare to historic land management and alternative future scenarios. Above, we reviewed the *BT16* scenarios compared to projected future baseline scenarios. Below, we consider historical data and trends.

3.6.1 Land for Biomass Crops in Context of Historical Trends

To place the *BT16* land allocations in 2040 scenarios into perspective, consider that over the 30-year period of 1982–2012, U.S. agricultural output increased persistently at an average rate of 1.5% growth per year, while over the same time period, the land used as an input for agricultural production fell at an average rate of 2.7 million acres per year—resulting in an 82 million–acre net reduction (summing cropland, pasture, and range), as shown in figure 3.6 (USDA 2015b). Focusing on the area of cultivated cropland, USDA analysis found that this input to production fell by 66 million acres, from 376.2 million in 1982 to 310.3 million in 2012, as illustrated in figure 3.7.

The ability to increase agricultural output while using less land over the past two decades is largely attributed to "total factor productivity" improvements (figure 3.8; USDA ERS 2016a; Wang et al. 2015). System efficiency can improve by increasing coproducts and reducing wastes. Risks and costs are reduced by diversifying market options and increasing flexibility for substitution.

Future agricultural land-use trends will be influenced by many factors, including the impact of climate change on crop yields (chapter 13), commodity prices, and agricultural policies. Under the *BT16* BC1 scenario, 64 million acres could be dedicated to biomass by 2040. This is similar to the acreage that could shift to non-agricultural uses if historical trends were to continue throughout the simulation period. However, future land-use trends may not follow past trends and are always uncertain. If new technologies and markets create incentives for cover crops, double crops, or higher yields, or if other mechanisms increase land-factor productivity, then less land will be required to meet future demand projections and more land would be available for other uses, including biomass. However, if yields do not grow as assumed in the *BT16* scenarios, or if weather or markets disrupt production, or demands for commodity crops are higher than anticipated, then less land would be available. Thus, while *BT16* simulations appear reasonable and are consistent with long-term historical trends for agricultural land management, actual future land use will be dependent on many factors.

Figure 3.6 | Net change in land cover/use between 1982 and 2012 (thousands of acres) (USDA 2015b)

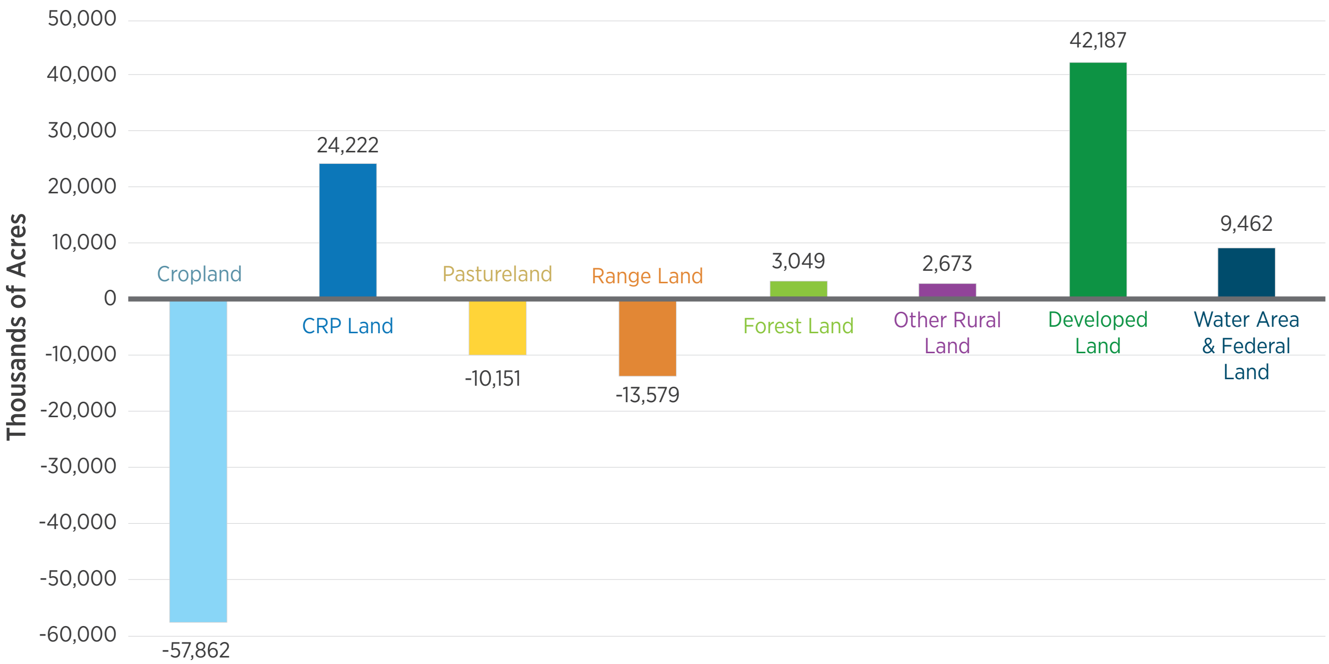


Figure 3.7 | U.S. cropland cultivated and uncultivated, 1982–2012 (USDA 2015b)

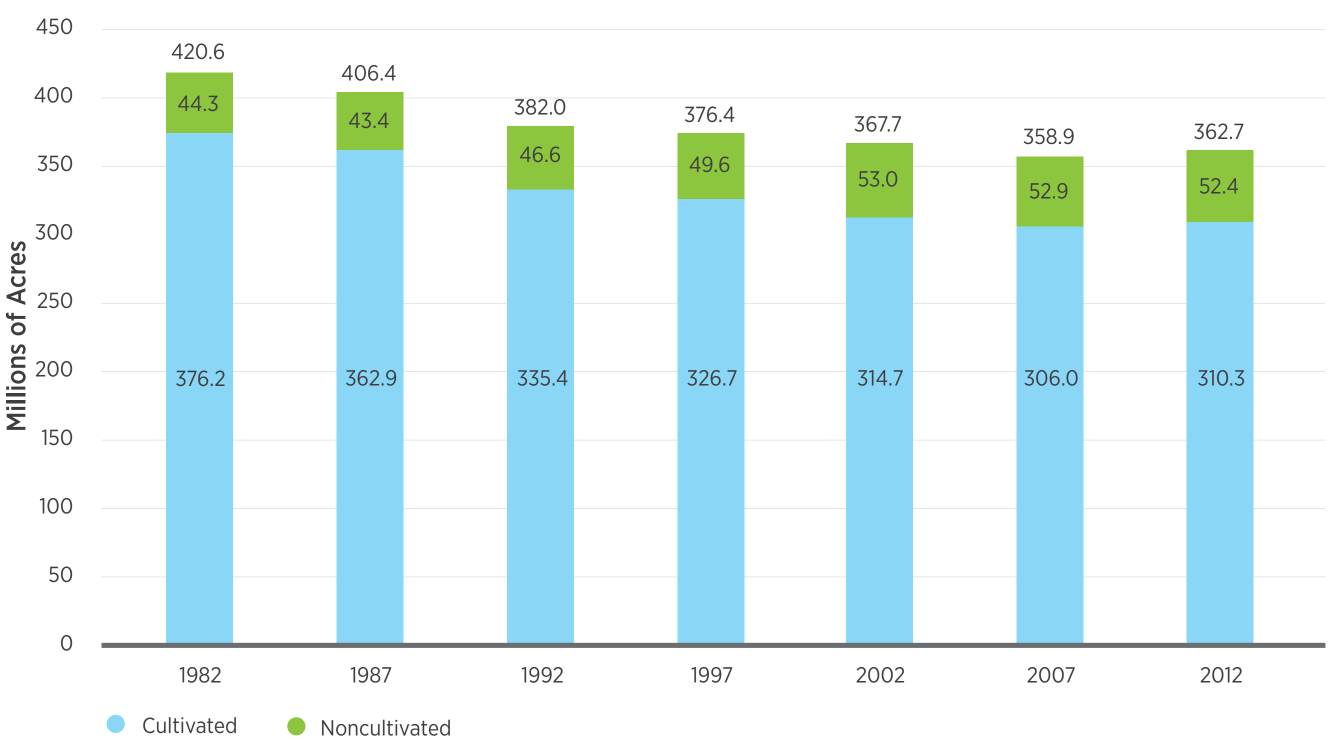
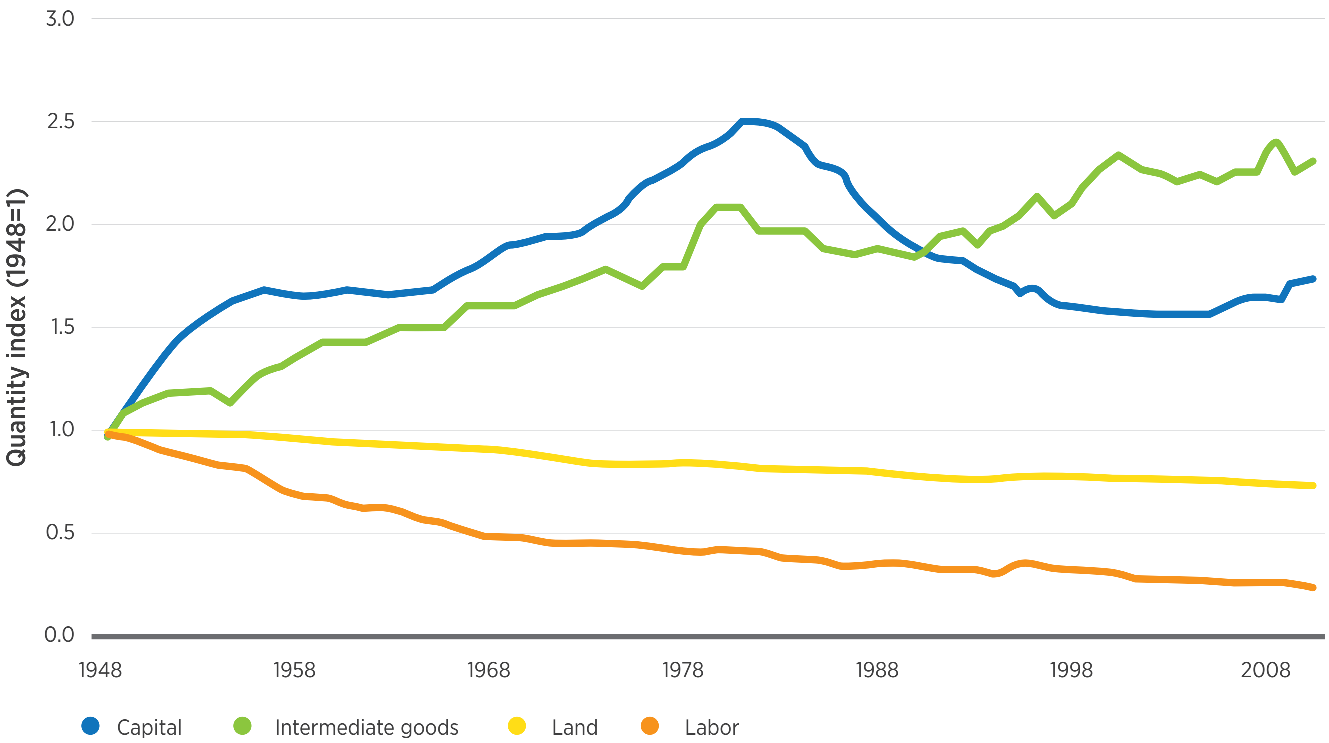


Figure 3.8 | Total factor productivity in U.S. agriculture steadily increased from 1948-2010 while the value of land as an input to production decreased (Figure reproduced from Wang et al., 2015).



3.6.2 Implications and Potential Benefits of *BT16* LUC

Desirable improvements in measured values for environmental indicators—such as air quality, soil carbon, and GHG emissions—are expected when management practices change from input-intensive annual crops to low-input perennial cover crops, SRWCs, and idle land (e.g., Robertson et al. 2008; Dale et al. 2014). Under *BT16* BC1 2040 and HH3 2040 scenarios, these transitions in land management (or LUC) from annual to perennial cover occur on 34 or 45 million acres, respectively. This is the most important type of LUC associated with *BT16* scenarios.

Despite data limitations and uncertainties, evidence from other chapters in this volume and biomass case studies shows that significant environmental improvements can be achieved when agricultural lands are managed for native perennial cover crops rather than annual crops (Dale et al. 2011; Robertson et al. 2008). Perennial crops require lower quantities of pesticides, herbicides, and fertilizers, as well as less mechanized field work, such as spraying, cultivating, and tillage passes (frequency and types of tillage), and less tillage depth (intensities) over time.

The measurement and interpretation of environmental indicators are highly dependent on contextual conditions (Efroymson et al. 2013). The benefits of native perennial cover crops depend largely on two variables: (1) the length of time perennial cover is sustained before soils are again cultivated or disturbed and (2) the alternative land management system in the absence of the perennials. However, net benefits of perennials also depend on additional contextual factors (e.g., soil types, slope, orientation, historical soil management, and crop rotations), management, and weather. Similarly, the effects of biomass crops on lands that were formerly pasture will depend on the types of cover or crops, how land is managed, and what the alternative land cover and management scenario would be in the absence of biomass markets.

To understand the magnitude of benefits that could be derived if 45 million acres of U.S. cropland that were previously managed for row crops were instead managed as perennial cover, consider experiences documented from CRP. The environmental benefits of CRP have been widely acknowledged (e.g., Cowan 2010; Dale et al. 2010; Dale et al. 2014; Herkert 2002; Herkert 2007; Robertson et al. 2008). The extent of CRP enrollment is currently capped by congressional legislation not to exceed 24 million acres (Agricultural Act of 2014, Pub. L. No. 113-79). That is less than the 27 million acres reserved for CRP under all *BT16* simulations. More importantly, it is less than half as large as the net reduction in annual crops simulated in *BT16* HH3 scenario (55 million acres). Therefore, some types of environmental benefits from the biomass production simulated under *BT16* could be estimated to be of similar magnitudes as, cover larger areas than, and be more widely distributed than current CRP, which is assumed to be maintained or allowed to expand somewhat under all scenarios. When land that was previously managed for annual crops becomes managed for perennial energy crops, the expected net effects on the environment depend on several factors, including the prior land conditions, prior land management, the energy crop that is planted, and the management of the energy crop system. Some research suggests that native grasses, such as switchgrass, can increase the abundance of bird species that are conservation priorities (Murray et al. 2003).

Outcomes are more uncertain on pastureland. Beneficial or adverse effects may occur when energy crops are grown on land formerly managed as pasture, depending on many contextual conditions. For example, if the baseline pasture is assumed to be a healthy mixed grassland that is subsequently cultivated and planted with a non-native (exotic), monoculture species such as miscanthus, declines in grassland bird species could occur (see chapter 10). On the other hand, if the baseline pasture is assumed to be poorly managed, over-grazed, or eroded, and subsequent management restores perennial cover with native grasses or SRWCs, there could be significant improvements in soil, water quality, wildlife habitat, and biodiversity. While many potential beneficial environmental effects can be estimated based on the results of *BT16* simulations, the uncertainties and limitations associated with any LUC analysis remain significant.

3.6.3 Uncertainties and Limitations in LUC Assessments

In developing and interpreting LUC assessments, one must gauge what questions are reasonable and useful to ask, balancing research objectives with available data and models. *BT16* was designed to estimate the quantity of economically-viable biomass that could be produced under a set of constraints that are meant to avoid or mitigate many of the potential negative impacts associated with LUC. The analysis is not a prediction of the future, but rather, a spatially explicit illustration of a specific biomass production case.

One advantage to the *BT16* approach is that it reduces some large uncertainties inherent in economic modeling of the LUC effects of energy crops (e.g., Plevin et al. 2015). Some researchers consider the uncertainty in LUC modeling to become unbounded and unknowable when indirect effects are included (O'Hare et al. 2010).

Nonetheless, several areas of uncertainty remain in *BT16* volume 1, and uncertainty is inevitable whenever future events are modeled. Uncertainties in LUC estimates arise from crop management assumptions, reference cases, and land classifications, all of which are discussed in more detail below.

BTI6 assumptions relevant to LUC include the specifications assumed for managing each crop system. The timing and type of land management are critical in determining changes in soil organic carbon, GHG emissions, and other environmental variables over time. Yet, spatially explicit data for management, such as type, depth, and timing of tillage activities, are limited. Agricultural scenario analyses tend to assume simple, single-step transitions from one crop to another crop, rather than the complexity involved in the use of cover crops and long-term rotations (Brankatschk and Finkbeiner 2015) or the highly variable tillage intensities and timing, which necessarily respond to weather conditions. Commodity market fluctuations are normal and also influence management in any given growing season. The uncertainty surrounding these variables increases exponentially as they are projected further into the future. Researchers are still learning about the extensive range of crop rotations and management practices used in U.S. agriculture today (Porter et al. 2016; Porter et al. 2015; James 2016). In the real world, land uses are not exclusive, as is assumed in models. For example, livestock are pastured on cropland after crop harvest. Similar practices can be applied to land managed for biomass. Any single field can provide a mix of products ranging from timber and biomass to fruit, grains, and pasture. When multiple crops and multiple uses are simplified into classes for analysis, LUC estimates may have little relationship to the actual changes in soil and water management on the ground.

Uncertainties are also associated with adoption rates for new crops and technologies. Swinton et al. (2016) found low willingness to bring marginal lands into production for bioenergy crops but generally found a greater willingness to use existing agricultural lands—a finding that is aligned with the assumptions applied in *BTI6* (Swinton et al. 2016; Swinton et al. 2011). However, analysis of these and other socioeconomic factors that influence adoption rates and LUC were not within the scope of this *BTI6* assessment.

Among many challenges associated with the *BTI6* analysis—and, indeed, most analyses that consider U.S. biomass production and LUC—is the lack of data to clearly characterize past land-use history. It is for this reason that the soil organic carbon change analysis (chapter 4) relies on assumptions about land-use history regarding how much time land had spent in cropland and pasture. Historical data for tillage and crop rotations have significant bearing on actual environmental conditions and future outcomes.

3.6.3.1 Reference Case

The reference case is the point of comparison used to estimate change. Reference cases may be called the business-as-usual, extrapolated future, extended baseline, or counterfactual case. Whenever a change is calculated, the point of comparison becomes the reference case. The reference case for most analyses in this volume is the BC1 2017 scenario. However, reviewers concerned about LUC recommended that this chapter consider the agricultural baseline as a reference case, as discussed earlier. Appendix 3-A reviews how different potential reference case assumptions can generate wildly divergent conclusions about the expected LUC associated with a set of well-defined *BTI6* scenarios.

History suggests that changes in the area classified as agriculture or cropland in the future will depend on a mixture of local and national factors, ranging from how ownership changes over time to stock market returns, policies impacting land taxation, farm programs and subsidies, and, particularly, the programs defined under the federal farm bill (i.e., the current 2014 farm bill [Agriculture Act of 2014, Pub. L. No. 113-79]). Farm bill provisions, such as crop insurance, CRP funding, and crop subsidies, have an influence on the U.S. agricultural landscape that appears to be more important than short-term price signals and biofuel markets (Babcock 2009; Kline et al. 2016). For example, despite price spikes in farm commodity prices that began in 2006, USDA acknowledged that "in 2007, total cropland area—which includes cropland used for crops, idled cropland, and cropland used for pasture—reached its lowest level since the Major Land Use series began in 1945" (USDA ERS 2016b).

Similarly, there are uncertainties in assumptions necessary to estimate future pastureland productivity and intensification options under reference scenarios. As with most aspects of modern agricultural production, the relationships between forage yield, stocking rates, management intensification practices, and other markets are far more complex in the real world than in model simulations. Historical trends show increasing livestock production from a decreasing land area, and the majority of U.S. meat now comes primarily from confined animal operations. As grain yields increase and prices stagnate, livestock producers may find it advantageous to continue shifting to supplemental feed as a substitute for grazing. For more details on the uncertainties surrounding pastureland in *BT16*, see volume 1.

In *BT16*, the reference system for agriculture is represented by the agricultural baseline (*BT16* volume 1, appendix C). Because there is a 10 million–acre difference between 2015 and 2017 agricultural baseline scenarios, the net reduction in annual crop acreage under *BT16* scenarios will depend on which reference case is used. This difference illustrates the importance of clearly specifying the reference case.

Assumptions are necessary to simulate future conditions as a reference point to estimate LUC. If a model assumes that, on the margin, land not required for agriculture returns to forest, that model's results are distinct from a model that assumes those lands would end up being managed for urban or other developed uses. Thus, the assumptions behind the reference case used in determining LUC are at least equally as important as those governing the biomass case. Yet, there is no agreement on how to best define a reference case for comparison (Soimakallio et al. 2015; Zamagni et al. 2012; Kline, Oladosu, et al. 2011).

There is also little agreement on how the timing of measurements should occur to define "change" and whether change should be simplified to be a single, irreversible event (as is often assumed in models) or to be represented by multiple events, cycles, and transitions that can be reversed (Dale and Kline 2013a). Partly due to these complications, the reference system is not clearly specified in most studies purporting to conduct LUC analysis (Soimakallio et al. 2015; Matthews et al. 2014).

3.6.3.2 Definitions and Data Sources

Differences in LUC estimates and their interpretation also arise when studies rely on different definitions or data sources for basic inputs, such as available agricultural land. For example, confusion is often generated from overlapping land classifications at the cropland-pastureland interface and the USDA definitions associated with pasture and grazing lands that have changed over time (see appendix 3-A). USDA sources for total pasture/rangeland on private property in 2007 ranged from 409 million acres to 529 million acres—a 120 million acre (30%) difference, depending on which source and definitions are used (USDA 2016c; also see table 2 in appendix 3-A). This is one of many reasons why there are large uncertainties when attempting to measure LUC involving cropland and pastureland.

Consider, for example, a 2016 article on LUC associated with biomass in the conterminous United States (i.e., the same area considered in *BT16*), which began by assuming an agricultural land base of 366 million acres, including both cropland and pasture (Hudiburg et el. 2016). This is less than half of the USDA-defined agricultural land base considered in *BT16* and helps illustrate how seemingly similar studies can generate divergent results. Different baseline land bases and different assumed land productivities will generate starkly different estimates of LUC associated with the same level of biomass production. Many published analyses of LUC for bioenergy lack a clear exposition of detailed baseline data and specifications for land classes and productivities, making it difficult to interpret and compare the results (Soimakallio et al. 2015).

3.6.3.3 Crop Rotations and Indistinct Lines among Land Classes

Crop rotations matter for LUC estimates because they imply changes in inputs, emissions, soil carbon, water quality, and other variables that depend not only on what is grown in a given year, but also on what was grown in prior years and what will be grown in subsequent years. For convenience, models of LUC omit most complexity of crop rotations. Some models, as in *BTI6*, choose a few representative rotations, such as corn-soy, for the analysis. Ideally, historical crop rotations over a 25-year period should be considered when developing scenarios 25 years into the future. Lacking such data adds uncertainty to LUC assessment and the corresponding estimates of soil organic carbon, GHG, and other factors. When assumptions omit or ignore past practices and crop rotations, the estimates of environmental impacts associated with land management for biomass production can be skewed, misrepresented, or misinterpreted (e.g., see Dunn et al. 2017; Dunn, Mueller, and Eaton 2015; Kline, Singh, and Dale 2013).

When the USDA National Laboratory for Agriculture and Environment (James 2016; Porter et al. 2015) assessed rotations in fields 15 acres or larger in size over a 6-year period (2010–2015) in the Corn Belt, 36,098 unique rotation strings were identified. While most rotations in the Corn Belt involve corn and soy beans, the next most common rotation observed was surprising: 5 years of pasture with 1 year of corn. Indeed, following the different variations of corn-soy rotations, the next six most common unique rotations identified by USDA in the Corn Belt all involved pasture in rotation with other crops. This suggests that a significant share of land classified as pasture is managed in rotation with annual crops. And conversely, a share of annual cropland likely includes forage or pasture rotations. Most LUC studies assume distinct boundaries and inherent differences in soil quality and productivity between pastureland and cropland in the United States. Available data sets such as the USDA "cropland data layer" have limitations when they are used to estimate LUC (Reitsma et al., 2016).

Complex and constantly evolving crop rotations are one of the challenges to conducting meaningful LUC analysis (Brankatschk and Finkbeiner 2015), especially where existing models allocate land among simple crop groups based on assumed average generic classes, such as pasture versus row crops. Monitoring to gather relevant measures of site-specific environmental indicators (e.g., soil and water qualities, productivity) that are associated with long-term management regimes (such as crop rotations) and then potentially incorporating the field data into models (Kröbel et al. 2016) will be important for improved future analysis. Given the history of U.S. agriculture and its shrinking footprint on the overall landscape, as well as the increasing complexity of observed crop rotations, these assumptions merit review and adjustment to align with empirical evidence.

3.7 Future Research

The large variability in results from previous LUC analyses associated with increased U.S. biomass production underscores the need for more consistent and transparent approaches to LUC assessment. One key area of future work could be to integrate the *BTI6* assumptions and outputs from BC1 and HH3 scenarios with global models to estimate potential ILUC effects. The following areas also merit further research—in collaboration with other agencies and stakeholders—because of their implications for the potential land management change and LUC modeling related to biomass production supply chains:

- The implementation of double cropping and the extent to which it is reflected in yield estimates

- The implementation of crop rotations and the extent to which they are reflected or not reflected in land-use and land-cover data
- The characterization of management practices, idle land, pasture, cropland-pasture, and CRP in agriculture models and how the evolving use of these lands can influence measurement of change (i.e., perceived LUC) in land characteristics and environmental indicators over time
- Effects on other markets that could be induced by changes in relative prices of biomass feedstocks
- Historical changes in U.S. land management, primary drivers of change, and the ways that biomass production interacts with those drivers
- The accuracy of assumptions about pasture intensification, based on an analysis of the scientific literature
- Inter-model comparisons for LUC effects of U.S. biomass production scenarios
- Updated empirical studies (such as indexed decomposition analysis) of effects of historical biomass production changes over time and correlation with environmental, social, and economic sustainability indicators
- The role of extreme events, environmental thresholds, and potential buffering effects associated with biomass supply chains
- Definition of a consistent and systematic hierarchy to characterize soil disturbance and management intensities for agriculture and forestry.

To better understand the effects on land cover and land management attributable to particular biomass production or to any specific intervention, monitoring needs to provide data on both the effects over time and the human behaviors that drive those effects. Considering how observed indicators evolve over time (before and after a policy is implemented or before and after management practices are modified, for example), while applying clear and consistent definitions for the effects of concern, can support causal analysis and attribution among multiple drivers of an observed LUC effect (Efroymson et al. 2016).

To understand how *BT16* compares to other studies requires investigation of the underlying data sets and input parameters (land classification, productivity, elasticity factors, etc.). This research could include the documentation of how different input parameters and specifications influence results. Such inter-model comparison efforts can help to pinpoint the items that require additional research to reduce uncertainty. In the near term, the specifications used for *BT16* could be compared to another well-documented analysis of LUC associated with a similar level of future U.S. biomass production (e.g., Hudiburg et al. 2016).

3.8 Conclusions

The objective of this chapter was to help readers interpret LUC associated with biomass supply changes from *BT16* volume 1, with an emphasis on energy and other agricultural crops. As described in this chapter, LUC can refer to land management change or land cover change or both. LUC scenarios are modeled and are therefore uncertain, but they are predictably dependent on model assumptions and input data. The purpose of *BT16*—to estimate biomass that could potentially be available at particular prices, given a market—necessitated that economic models would be used to estimate changes in land management. Moreover, land management change determines environmental effects that are estimated elsewhere in this report.

The constraints and assumptions applied in models determine the range of results that are possible. *BT16* simulations are constrained so that no net changes occur in forest and agriculture land areas. Input assumptions regarding land classes and productivity have a major influence on how land is allocated among traditional and energy crops over time within the agricultural sector. The implementation of constraints in *BT16* effectively reduces potential adverse environmental effects and also reduces the potential biomass supply itself compared to volumes of biomass that could be estimated in the absence of the constraints.

In *BTI6*, the total land area estimated to be managed for energy crops on agricultural land in 2040 is 64 million and 88 million acres under BC1 and HH3 cases, respectively. In both cases, 97% of the total estimated energy crop acreage is managed as perennial crops, such as switchgrass or SRWCs. The remaining 3% of the biomass crop area is composed of biomass sorghum, an annual crop—1.7 million acres in BC1 and 2.3 million acres in HH3. Also, note that the net area in idle land remains constant in all three 2040 scenarios, but amounts and locations of idle cropland vary in each scenario, as idle land rotates with other crops.

The primary type of LUC associated with *BTI6* biomass supply scenarios involves land management practices to transition up to 45 million acres of annual crops to perennial cover by 2040. The environmental effects that are discussed in the following chapters are largely outcomes of this LUC.

The environmental benefits derived from shifting land from annual crops to native perennial cover can be expected to resemble the benefits that have been documented for the CRP program. However, effects associated with monoculture and exotic crops such as miscanthus replacing mixed vegetation on pastureland could be negative.

Under *BTI6* biomass demand scenarios in 2040, 37 million (BC1) or 39 million (HH3) acres of pastureland (approximately 8% of total pasture area in the 2015 agricultural baseline and the BC1 2017 scenario) are managed for the perennial energy crops shown in table 3.4. As described in chapter 2, a proportional share of remaining pasture (56–58 million acres) undergoes improved management (fencing, rotation) to accommodate the biomass crops while meeting other market demands. The assumptions regarding intensification of pasture are required to produce biomass feedstocks within the constraints established that aim to meet other market demands without changing the total area dedicated to agriculture and forestry.

The land management changes described above reflect the purpose of *BTI6*: to identify where and how much biomass is potentially available at particular prices, assuming a growing U.S. bioeconomy. While the scenarios and results can be useful for policy analysis, they are not meant to reflect anticipated policies or predictions. Other LUC studies ask different questions and use different approaches and assumptions. Few LUC models specify all the implications of their assumptions and modeling parameters with respect to land management changes. This is a key component of the *BTI6* analysis. In all LUC studies, the approaches and assumptions should reflect clearly defined research goals.

The ambiguity in overlapping land-use labels leads us to call for science-based indicators and monitoring to test hypotheses related to any environmental effects (e.g., measured changes in soil organic carbon, biodiversity, GHG emissions, etc.) that occur in response to the changes in management required for biomass crops, rather than assuming effects based on perceived "changes" from pasture to cropland. At a minimum, consistent definitions for land cover and land management are required to support a consistent analysis of change over time. This is not easy given that even within the U.S. government, definitions, classifications, and measuring methods vary over time and among agencies. *BTI6* mitigates some of these problems and uncertainties by clearly documenting assumptions and sources and applying a single model to represent sectoral activities (e.g., POLYSYS for agriculture, Forest Sustainable and Economic Analysis for forestry).

The challenges faced when trying to measure LUC associated with biomass production are large. Consistent and transparent use of terms and definitions for land cover classes, crop types and rotations, and characterization of land management are essential elements for improved LUC analysis. The land class definitions and initial acreages applied in *BT16* are based on USDA sources, and the simulation assumptions are consistent with current land uses, laws, and regulations.

Our review of LUC modeling concludes that different approaches attempt to answer different questions, and each approach will generate results that are driven by model specifications, definitions, data sets, and assumptions. Empirical data are not available to support definitive analysis when simulating the future. Therefore, assumed values are applied in models, and the assumptions have a large influence on estimates of LUC and corresponding environmental effects.

Improved monitoring of changes in land cover, crop type, and land management practices (all of which represent different aspects of LUC) is recommended as a basis for reducing uncertainty. Monitoring, including monitoring of changes in clearly defined land attributes, is essential to guide continual improvement in environmental indicators and in the models that simulate them. A U.S. bioeconomy should provide a reliable source of renewable biomass for materials and energy while promoting beneficial LUC, defined as continual improvement in land management practices over the long term, to provide multiple services and benefits to society.

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Appendix 3-A: Terminology, Definitions, and Sources

Science-based sustainability metrics apply methods for consistent measurements. Metrics with relevance to *BT16* and LUC include: crop type (along with the type, carbon stocks [density], evolution, and duration of specific characteristics of vegetative land cover), soil management practices (type, intensity and frequency of tillage, and other activities that disturb or impact soil, water, and vegetation), productivity (above and below ground, both in terms of material harvested and in terms of total NPP [McBride et al. 2011]), disturbance regimes, and environmental indicators analyzed in other chapters of *BT16* volume 2 (e.g., soil carbon, GHG emissions, biodiversity, etc.). Additional metrics are applicable to forest management and LUC (structure, age class, above and below ground carbon, NPP, etc.).

Indirect LUC is not a science-based metric. There is no agreement on clearly defined units, replicable measurement procedures, or published standards for assessing and distinguishing between direct and indirect LUC. As

Table 3A.1 | Published Definitions and Descriptions of ILUC with Respect to Bioenergy Vary Widely and Allow for Subjective Interpretations

Definition	Reference
"When existing cropland is used for biofuel feedstock production, forcing food, feed, and materials to be produced on new cropland elsewhere. This expansion is called indirect land-use change, or ILUC...Because ILUC occurs through global market mechanisms with many direct and indirect effects, it can only be modelled, not measured."	Valin et al. 2015
"Market-mediated or policy-driven shifts in land use that cannot be directly attributed to land-use management decisions of individuals or groups," where land use refers to "the total of arrangements, activities, and inputs undertaken in a certain land cover type."	Verbruggen, Moomaw, and Nyboer 2011
"Whereby mechanized agriculture encroaches on existing pastures, displacing them to the frontier," "takes place when agricultural activities displaced from one region are reconstituted in another one...In such a situation, deforestation at particular locations occurs partly due to events far away," "occurs as loss of land dedicated to a given crop (or production strategy) in one region triggers its expansion in another region."	Arima et al. 2011
"The hypothesis is that the planting of biofuel crops on pastures or croplands in consolidated agricultural regions induces increased expansion of agricultural land in frontier regions to compensate for the lost food production capacity."	Barretto et al. 2013
"Land-use change that occurs outside the system boundary because of the loss of a service that the land provided before the application of the bioenergy activity."	Bird et al. 2010
"If the area (where the cultivation of the biofuel crop is taking place) was previously utilized for other purposes, that activity might be displaced to other areas. This...may occur in the same country where the feedstock is produced, but due to the international trading of crops it is possible that they are displaced to other parts of the world competing with local production of food, feed, and with nature conservation."	Di Lucia, Ahlgren, and Ericsson 2012

Definition	Reference
"Displacing previous production to other land" following "the production of biofuels feedstocks on arable and pasture land."	Fritsche and Wiegmann 2011
"Occurs outside the system boundary because of the displacement of services (usually food production) provided by the land before the change."	Bird et al. 2011
"results in displacement effects, including price-induced changes in global commodity markets, that, in turn, also lead to land being altered from one state to another, with resulting changes in GHG emissions and carbon stocks on that land"	Sanchez et al. 2012
"when pressure on agriculture due to the displacement of previous activity or use of the biomass induces land-use changes on other lands in order to maintain previous level of (e.g., food) production"	Van Stappen, Brose, and Schenkel 2011

illustrated in table 3A.1, LUC and ILUC are ambiguous and subjective terms that have been defined and interpreted inconsistently.

Science-based analysis begins with clear terms and definitions (Dale et al. 2013). The lack of agreement on clear definitions has been noted as an underlying factor confounding analysis of LUC and ILUC (Kline et al. 2011; Warner et al. 2014; ISO 2015). Agreement on definitions, and consistent use aligned with those definitions, are prerequisites for understanding and communicating the effects of bioenergy production on land and for the allocation of causal burden to different factors in the case of a defined land disturbance, such as deforestation (Efroymson et al. 2016).

One common example of LUC cited in the literature is deforestation, a change in land cover typically defined by remote sensing analysis. The change in classification from forest to some other use is easier to observe and measure than most other LUCs, yet presents many challenges. The threshold point at which classification of a land unit changes is independent of actual land use before or after deforestation was identified. Deforestation typically results following decades of changes and incremental degradations prior to the point when a threshold (e.g., 10% canopy cover) is no longer met and land unit classification changes. Another example of LUC found in the literature is when production from cropland (e.g., a field in conventional corn/soy rotation) is used for bioenergy rather than animal feed. In this case, all aspects of land cover and management could remain unchanged while the use of one part of the harvested grain changes. Another example could be when the corn/soy rotation field switches from conventional tillage to reduced-till (a change in management practice). Another LUC could be when the legal status of a parcel changes (even if nothing else changes).

3A.1 Issues of Initial Land Cover Classification are Complex and have Huge Influence on LUC Analysis

For assessing LUC in the United States, USDA (Allison A. Borchers, personal communication) recommends that the National Resources Inventory (NRI) be used. If considering effects on an indicator associated with changes in land cover, please note that the USDA NRI (USDA ERS 2015, USDA ERS 2016) is the government data product designed to provide wall-to-wall consistency in land cover and use. NRI is explicitly designed "to provide legitimate trends and estimates of change across multiple points in time." The NRI classification system uses a different set of definitions than those used by *BT16* and USDA Agricultural Census to distinguish between cropland (363 million acres), pasture (121 million acres), and rangeland (406 million acres) (see table 3A.2). The constraints applied in POLYSYS simulations effectively limit the modeled supply of energy crops to a subset of the cropland and pastureland as defined in the NRI. The 2015 NRI is the only U.S. government source designed to provide nationally consistent data for U.S. land cover and land use over the 30-year period of 1982–2012 (USDA 2015), using the following classes for all non-federal land:

- Cropland including tilled and untilled (cropland pasture) and CRP
- Pasture (seeded and managed for forage crops with periodic inputs, complications can arise as the definition overlaps with some cropland-pasture and some permanent pastures)
- Rangeland (these lands may be managed and seeded but are less intensively managed for grazing than pastureland)
- Forestland (based on USDA Forest Inventory Analysis)
- Water
- Developed, barren and "other rural land" (homesteads, roadways).

USDA explains that there are many different sets of data for land area in a given class, depending on year, data source, and definitions applied. Table 3A.2 illustrates some of the differences. The potential for misinterpretation when doing LUC analysis is high when users re-arrange classes or make assumptions about subcategories such as idle cropland and CRP. For example, by reclassifying those cropland subcategories as grassland, and then declaring a LUC whenever those parcels are put back into production, an analysis can generate large quantities of LUC. And by ignoring the total landscape dynamic of cropland-pasture/grassland rotations, the LUC can be further exaggerated (Kline, Singh, and Dale 2013).

While differences in reported area for a given land cover or use are sometimes purely jurisdictional (e.g., the Bureau of Land Management manages 158 million acres of public pasture/range lands) or depend on whether federal lands are included or excluded (e.g., forest), the choice of data set has huge implications for any LUC analysis. The areas by class cited in the table below vary from 311 million to 408 million acres (over 30%) for cropland; from 409 to 751 million acres (80%) for forest, and 409 to 995 million acres for permanent pasture/range (140%). Even when only private lands are considered, the values vary significantly. For example, Nickerson et al. (2015) show that in 2007, private pasture/rangeland area could range from 409 million acres under NASS surveys to 529 million under NRCS surveys, a 30% difference depending on which source and definitions are used.

When LUC analyses use data from multiple sources and classification schemes, or selectively use data without accounting for "wall-to-wall" land cover in a landscape, it becomes impossible to verify a baseline and undermines credibility of the simulations. These LUC analyses become "shell games" where changes are calculated for selected parts of a landscape without accounting for all the corresponding changes in the remainder of the landscape (Kline, Singh, and Dale 2013). The USDA Economic Research Service provides guidelines for use of data and recommends that the NRI data set be used for LUC analysis involving major land classes.

Table 3A.2 | Land Use and Land Cover Estimates for the United States, by Source (Nickerson et al. 2015)

	Land Use					Hybrid (LU/LC)	Land Cover	
	USFS	BLM	NASS	Census Bureau	ERS	NRCS	USGS	BLM
Scope of Coverage	All forestland	Area managed by BLM	Land in farms	Urban areas	All land uses	All non-federal land	All land and water cover	Area managed by BLM
Category	*Millions of acres*							
Forest/ woodland	751	11	75	-	671	409	600	69
Forest in timber use		11	46	-	544			
Forest in grazed use			29	-	127			
Permanent pasture/range	-	158	409	-	614	529	995	174
Cropland	-	-	406	-	408	390	311	-
Urban areas	-	-	-	68	61	112*	102	-
Rural parks, wilderness areas	-	2	-	-	252	-	-	-
Rural trans-portation	-	-	-	-	26	*	-	-
Other	-	85	32	-	232	504	373	13
Total area included in estimates	751	256	922	68	2,264	1,944	2,381	256
Total U.S. land area: 2,264 million acres[a]								
Total U.S. land and water area: 2,381 million acres[b]								
Year estimates were derived	2007	2007	2007	2010	2007	2007	2006**	2007

[a] Source: Census Bureau

[b] Source: U.S. Geological Survey

* NRCS combines Urban areas and Rural transportation into a Developed land category. NRCS estimates exclude Alaska.

** USGS data are from 2006, except Alaska and Hawaii estimates are from 2001.

3A.2 Reference Case Considerations for LUC Modeling

Interpretations of outputs from any prospective model should reflect the assumptions and constraints imposed on the model, recognizing that the outputs are not a prediction of the future. The inherent uncertainties of future projections are compounded if results are then used to estimate a "change" compared to some other simulated future or reference system. Effects of LUC are manifested in the differences identified when the biomass scenario is compared to the reference case scenario (Koponen et al. 2016). Projecting management details and effects into the future inevitably involves significant judgment and guesswork for both the biomass and reference scenarios. Independent of the constraints applied in *BT16* and the agricultural reference cases illustrated in this chapter, a range of other plausible reference cases for *BT16* can be considered. Consider the following possibilities for what could occur on the landscape in the absence of bioenergy markets:

- The agricultural land used for energy crops in *BT16* scenarios could return to forest. This possibility is supported by the historical transitions observed in different parts of the United States from the 1800s to the 1980s. However, little evidence supports this hypothesis in more recent decades, given current trends in U.S. land cover (USDA 2015).
- The agricultural land used for energy crops in *BT16* could transition to urban and developed uses, since this has been the predominant type of expanded land use leading to net loss of agriculture land over the past 40 years and continuing to present. However, the rate of loss to developed uses has declined in recent years.
- The agricultural land used for energy crops in *BT16* could transition into cropland pasture and forage crop rotations, as acreages for these land covers tend to expand when row crop prices fall and shrink when row crop prices rise, and because rotations between cropland and pasture represent the largest gross LUC over the past 40 years (Lubowski et al. 2006).
- The agricultural land used for energy crops in *BT16* could simply be left in agriculture and managed for lower yields and/or lower-risk crops. This has been observed in the past, for example, when low corn prices led to fewer acres in high yield (densely seeded) corn, and more acres in lower-yield corn, sorghum, and soy beans. Aspects of this scenario are reflected in the agricultural baseline as total agricultural area remains unchanged but the land in rotation as "idle cropland" increases and other crop and pasture land areas hold mostly constant through 2040.

Historical evidence suggests that at least a bit of all of these reference case alternatives will emerge with or without bioenergy markets. How much transition occurs, where, and which types of transition predominate, will depend on many factors, with bioenergy markets playing a minor role relative to the many more significant policy, environmental, and economic factors that determine crop prices, productivity, access to markets, and sector growth.

The rate of increase in productivity assumed in the agricultural baseline as projected to 2040 is less than the historic average rate of 1.5% per year documented over the prior 3 decades (Wang et al. 2015), a period when total agricultural land area decreased by 82 million acres (USDA 2015) as cropland outputs rose. However, while historic trends on a national basis point to improved productivity and reduced overall cropland area (USDA 2015), studies examining selected areas in the Midwest over short time frames found the opposite trend (e.g., Lark, Salmon, and Gibbs 2015 examined four years [2008–2012]; also see comments on methods and results: Dunn, Mueller, and Eaton 2015; Kline, Singh, and Dale 2013). These contradictions underscore the need for better monitoring and accurate assessment of land management and effects on well-defined, verifiable qualities for soil, water, and vegetation.

3A.3 *BT16* LUC Constraints and Land Allocation Scenarios

BT16 biomass supplies are estimated under assumptions that prohibit net cropland expansion into forestland (or vice versa) and biomass crop harvest on sensitive lands (see chapter 2). One rationale for such constraints is to reduce the number and types of assumptions required for modeling. Another reason is that it avoids many complications involved when intermingling large data sets from different sources, a necessity whenever a model attempts to couple forestry and agricultural models, or attempts to expand beyond the temporal or spatial boundaries of available census and land (remote sensing) data products. Further, in order to model LUC between sectors, value judgments and assumptions are required to define what is expected to cause or deter future exchanges between forestland and agriculture. For example, some studies have attempted to estimate the potential impacts of bioenergy markets on CRP lands (e.g., Walsh et al. 2003; Secchi et al. 2009; Huang, Khanna, and Yang 2011), but the economic model projections for large-scale CRP contract cancellations and non-renewals in response to high corn prices proved to be wrong. Demand for CRP contracts consistently outstripped the funding available for the program and CRP contract area peaked in conjunction with some of the highest corn prices on record.

BT16 scenarios identify sustainable supply potential and therefore prioritize CRP as a land use (27 million acres of CRP were excluded from the scenarios [see table 3.1]). Furthermore, the past four decades of U.S. experience reflect significant swings in commodity prices without notable response in the relative size of the agricultural and forestland areas. This is due in part to a large latent productive potential in U.S. agriculture. U.S. farmers have demonstrated an ability to respond to rising price signals, over-produce and drive prices back down, while consistently using less total agricultural land (USDA 2015; USDA ERS 2015; USDA ERS 2016; Lubowski et al. 2006).

BT16 constrained biomass production to land already in productive agricultural and forestry uses in 2015. The scenarios analyzed in volume 2 also excluded irrigated land. These constraints limit potential impacts in sensitive and special-use lands to previously existing conditions. By definition, no LUC occurs on sensitive lands. The simulations are also designed (see chapter 2) to reduce potential for international indirect effects by prioritizing estimated future demands for food, feed, fiber, and exports through adjustments using price elasticities (see volume 1:360). Additional assumptions and constraints are applied to limit the rate, scale, and types of simulated transitions from conventional crop management and pasture to management for energy crops.

If *BT16* had not incorporated assumptions that limit biomass potential from less sustainable sources, the projected biomass supply at any given price point would be larger. There are several reasons to support the assumed LUC constraints. First, changes in agriculture and forestry production systems take time and the incremental nature of change is reflected by the constraints applied. Second, current U.S. energy and land policies protect water, soils, and other ecologically sensitive lands (e.g., see EPA 2016; NRC 1993) and explicitly exclude biomass from federal forests and from land that was not already in agricultural production in 2007 (EPA 2010). Third, the constraints are consistent with historic land-use trends as discussed below (USDA 2015). Fourth, such constraints are consistent with the U.S. strategic plan for decarbonizing the economy (White House 2016) and nationally determined contributions to the Paris Climate Accords, and the U.S. Bioeconomy Vision (BRDI 2016). Finally, eliminating these constraints would be inconsistent with the *BT16* aim to estimate sustainable biomass supply.

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04 | Fossil Energy Consumption and Greenhouse Gas Emissions, Including Soil Carbon Effects, of Producing Agriculture and Forestry Feedstocks

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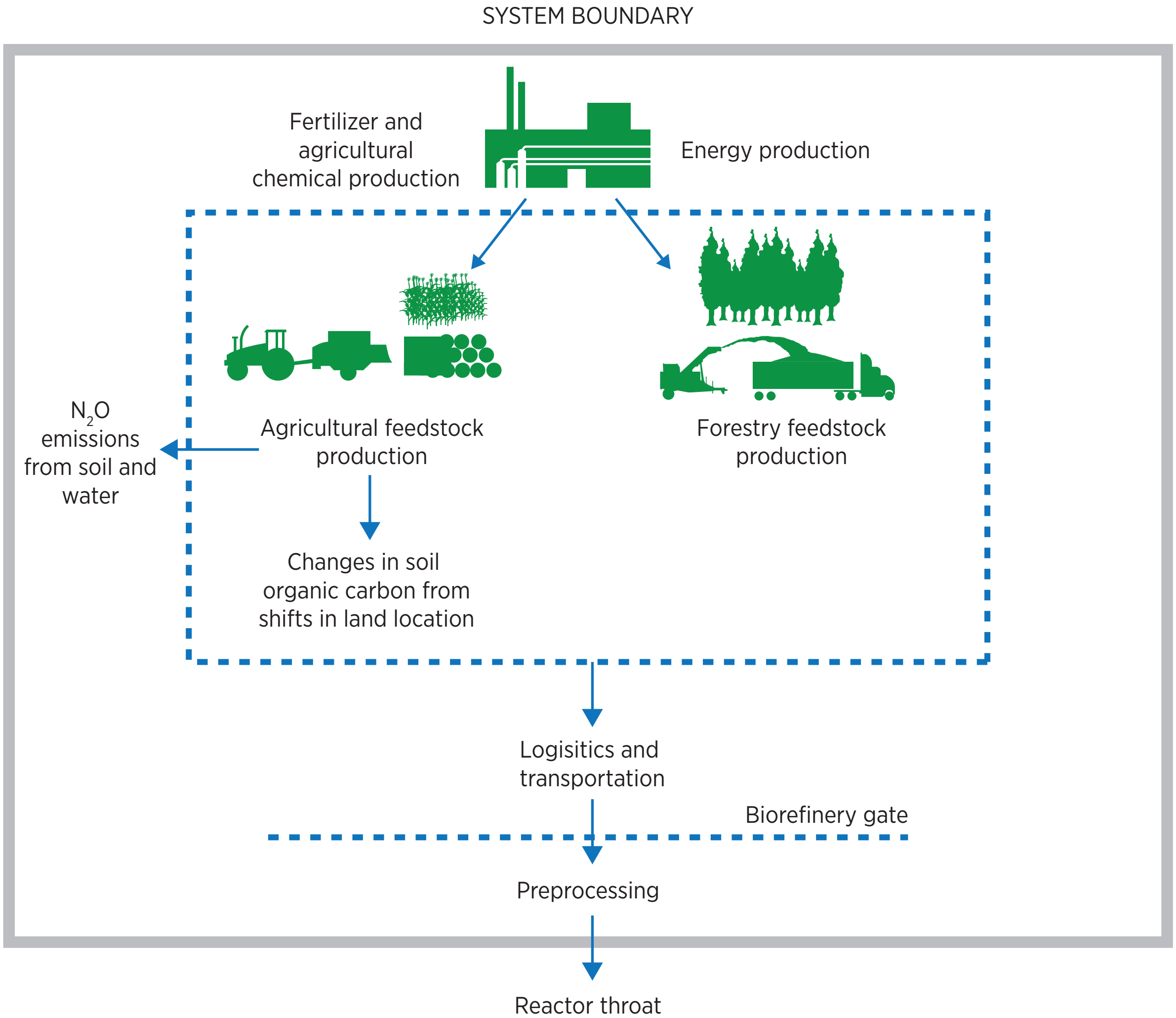
4.1 Introduction

One key measure of the environmental effects of producing biomass is the associated greenhouse gas (GHG) emissions. In this chapter, GHG emissions refers to the carbon dioxide equivalent (CO_2e) of CO_2, methane (CH_4), and nitrous oxide (N_2O) emissions combined with their 100-year global-warming potentials in the Intergovernmental Panel on Climate Change's *Fifth Assessment Report* (IPCC 2013). Furthermore, an objective of expanding the domestic biomass supply is to reduce fossil energy and petroleum consumption through application of biomass toward different processes and products that currently use fossil energy sources as feedstocks. In this chapter of the *2016 Billion-Ton Report (BT16)* volume 2, fossil energy consumption and GHG emissions associated with producing biomass—including the upstream energy consumed and emissions released from fertilizer production, agricultural chemicals, and fuel used in farming—are estimated. In addition, we consider the contribution of changes in soil carbon to net GHG emissions as a result of producing feedstock on land that was previously in other land covers or under different management practices prior to production of biomass estimated to be grown under *BT16* volume 1 scenarios. This analysis was carried out with the Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET®) model as released by Argonne National Laboratory (ANL) in 2015.

The results presented in this chapter include the GHG emissions and fossil energy consumption associated with select scenarios defined in the first volume of *BT16*. These scenarios are the base case for 2017 (agricultural base case and forestry baseline combined; BC1&ML 2017)[1] and base and high-yielding 2040 cases (BC1&ML 2040, HH3&HH 2040). *BT16* volume 1 analyses did not include a business as usual case for forestry and agriculture and analysis of associated GHG emissions does not either. Results are presented at the county level and include calculated GHG emissions and energy consumption per dry ton of feedstock for each feedstock type. The results reflect the GHG and energy intensity of producing only agricultural and forest-derived biomass in each *BT16* scenario, not the emissions and energy associated with the entire agricultural and forestry sectors. National-level results for GHG emissions and fossil energy consumption are also presented. The system boundary for the analysis of agricultural and forestry feedstocks is shown in figure 4.1. The system boundary for both types of feedstocks is similar and includes direct energy use during feedstock production, transportation, and preprocessing; energy required for fertilizer and chemical production; and N_2O emissions from fertilizer application and biomass decomposition. However, changes in soil organic carbon (SOC) are only evaluated for agricultural feedstocks. Furthermore, because forested area was held constant (agricultural land did not expand into forested land), and therefore forested areas were not cleared in Volume 1 scenarios, changes in above-ground carbon were not considered. Please see Chapter 3 for a discussion of land use change in *BT16* scenarios and section 4.2.3 for additional discussion of above ground carbon. Materials and energy consumed in the manufacture of farming/forestry equipment and trucks used for biomass transportation are excluded from this analysis. Indirect GHG emissions from growing biomass—for example, from indirect land-use change brought about by market factors—are outside of the system boundary.

Furthermore, we incorporate cases from Rogers et al. (2016) in which the biomass produced per *Volume 1* is converted to biofuel, bioproducts, and biopower that can then displace petroleum-derived fuels, products, and power. This exercise permits an estimation of GHG emissions and fossil energy consumption reductions as compared to business as usual (BAU) scenarios and expands the system boundary beyond that of the *BT16* analysis (fig. 4.1). However, the expansion of boundaries to include reduction of emissions from fossil energy consumption does not account for changes in costs in fossil fuel-based and bio-derived fuels and products over time.

Figure 4.1 | System boundary of this chapter's analysis. All steps within the gray box are included.



4.2 Methods

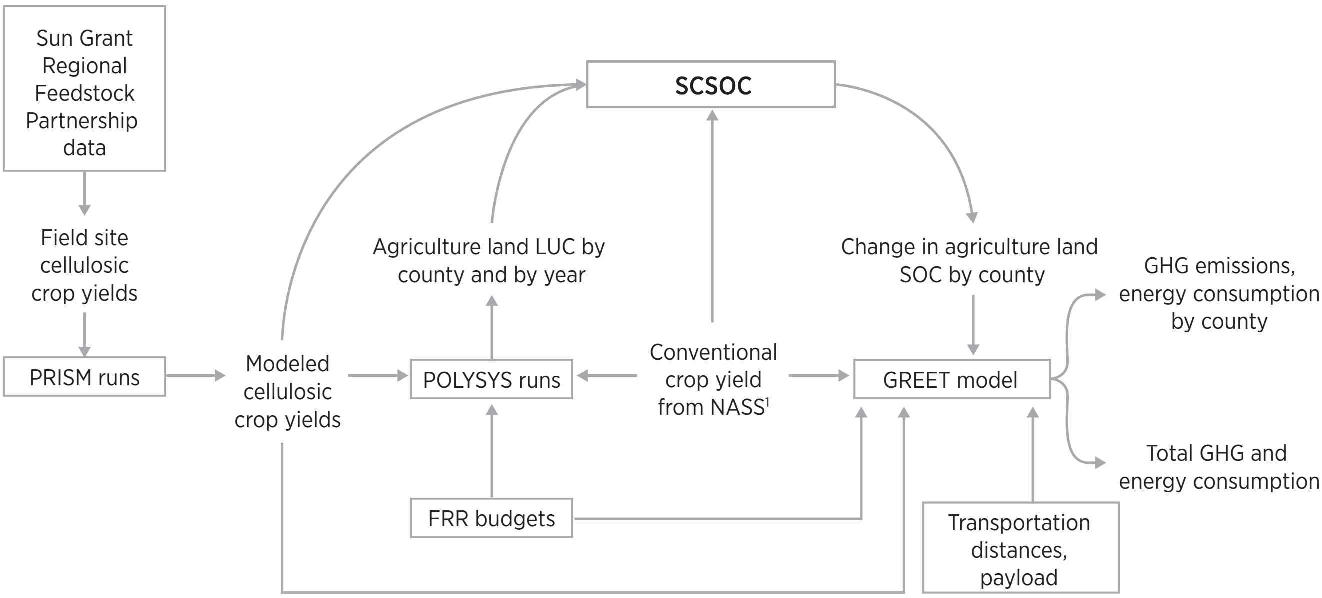
This section provides an overview of the methodology for estimating fossil energy consumption and GHG emissions in *BT16* for base-case (BC1 2017 and BC1 2040) and high-yield (HH3 2040) scenarios for agriculture, and moderate growth in housing/low growth in wood energy (ML 2017 and ML 2040) and high growth in housing/high growth in wood energy (HH 2040) scenarios for forestry (see chapter 2 for details regarding each scenario). Figures 4.2 and 4.3 present the data and calculation flow used to estimate GHG emissions associated with biomass production in the agricultural and forestry sectors. The following subsections describe each step of this methodology including data sources and assumptions.

4.2.1 Material and Energy Consumption during Feedstock Production

To estimate fossil energy consumption and GHG emissions associated with the production of biomass, the first phase shown within the system boundary (fig 4.1), energy, fertilizer, and chemicals consumption

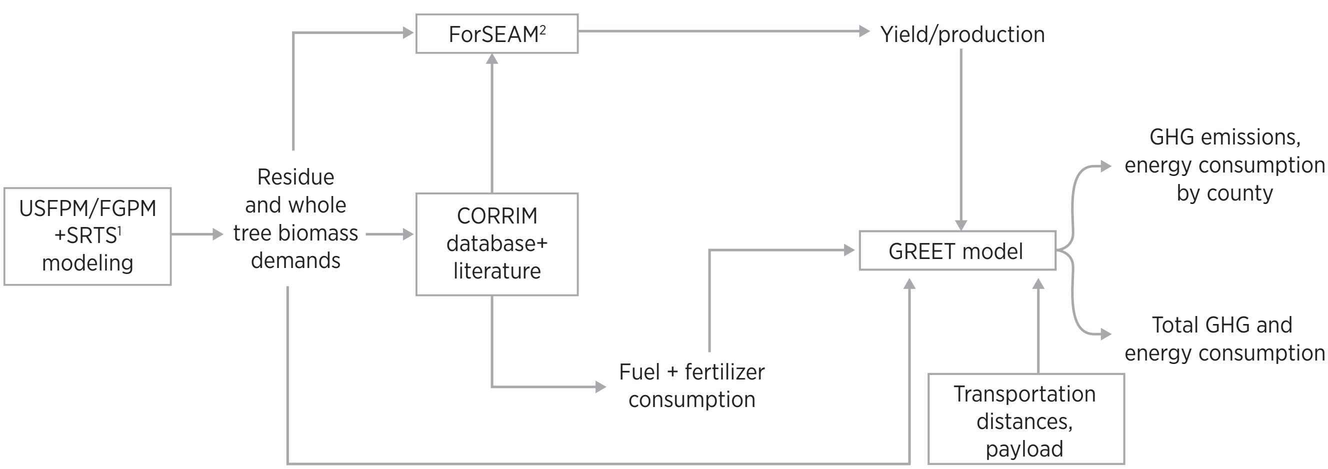
[1] This chapter uses combinations of agricultural and forestry scenarios to provide a projection of possible environmental effects from both types of biomass. Therefore, the convention of the "&" sign is used to represent a combination of two scenarios.

Figure 4.2 | Schematic of methodology applied to estimate GHG emissions associated with producing agricultural biomass. The fossil energy consumption estimation methodology is analogous but does not incorporate input from SCSOC (Surrogate CENTURY Soil Organic Carbon model, based on CENTURY, which is available from Colorado State University).[1] NASS: represents the National Agricultural Statistics Service from the U.S. Department of Agriculture (USDA). FRR Budgets: Farm Resource Regions as defined by the USDA are depicted in figure 4.10. For each FRR, there is one budget containing fuel, fertilizer, and agricultural chemical consumption per feedstock. Transportation distances, payload, and pre-processing fuel consumption are based on results in chapter 6 of *BT16* volume 1.



[1.] NASS: represents the National Agricultural Statistics Service from the U.S. Department of Agriculture

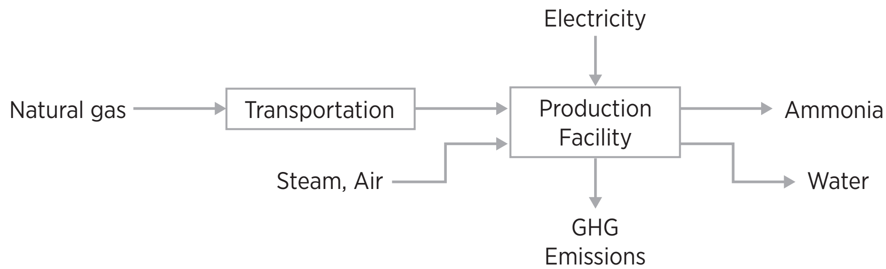
Figure 4.3 | Schematic of methodology applied to forestry-derived feedstocks to estimate GHG emissions from biomass production. Fossil energy consumption estimation methodology is analogous. Transportation distances, payload, and pre-processing fuel consumption are based on BT16 volume 1, chapter 6 results. (CORRIM – Consortium for Research on Renewable Industrial Materials)



1. U.S. Forest Products Module/Global Forest Products Model (USFPM/GFPM) with the Subregional Timber Supply (SRTS) to determine wood energy demands

2. ForSEAM is a version of POLYSYS developed for forestry

Figure 4.4 | Process to produce ammonia fertilizer, the emissions from which are included in the total emissions associated with biomass produced with ammonia as a fertilizer. (Upstream burdens to produce and deliver natural gas and electricity to the fertilizer plant, not shown in the figure, are also included.)



per unit area of land for each crop and the yield of each crop are required. For analyses of the agricultural sector, Farm Resource Region (FRR) budgets are the source of fuel, fertilizer, and chemical consumption. In the case of forest-derived feedstocks, fertilizer application and energy consumed in harvesting and site prep derives from the literature and from a Consortium for Research on Renewable Industrial Materials (CORRIM) database. On-site fuel, fertilizer, and chemical consumption could be called "purchased energy" or "on-site materials consumption." GREET estimates the upstream burdens (i.e., consumption of materials, energy, and emissions) associated with producing these fuels, fertilizers, and chemicals to yield a "full fuel-cycle" result for material and energy inputs to farms at the county level. Figure 4.4 provides an example of how full life-cycle GHG emissions associated with ammonia production are calculated in GREET (Johnson, Palou-Rivera, and Frank 2013). The calculation accounts for natural gas and electricity production to the point of use at the ammonia facility. At the facility, methane reforming, a water-gas shift reaction, and methanation occur. Carbon dioxide is produced by the water-gas shift reaction and is emitted to the atmosphere, which is accounted for in GREET. Additionally, emissions from transporting ammonia-plant inputs to the production facility and produced ammonia to farms are included. The total of these upstream emissions from ammonia production is assigned to biomass produced with ammonia as a fertilizer. Similarly, upstream emissions associated with all inputs to the production of agriculture and forestry biomass are included in this analysis. We note that feedstock production emissions for any given crop reflect those incurred in the year the feedstock is harvested. A full description of the calculation of energy and GHG intensity of agricultural and forest-derived biomass is contained in appendix 4-A.

4.2.2 Estimation of SOC Changes

An in-depth analysis of SOC changes upon bioenergy-crop-relevant land transitions using the surrogate CENTURY soil organic carbon (SCSOC) model at both state (Kwon et al. 2013) and county levels (Qin et al. 2016a) was conducted in previous work. SCSOC uses calculations and parameters from CENTURY, but it has been modified to permit simulation of bioenergy crop production (Qin et al. 2016a). Important inputs to this model include crop yield, the root-to-shoot ratio, soil type, and weather data (Kwon et al. 2013; Qin et al. 2016a). SCSOC-estimated changes in SOC are treated as emission factors (EFs) in units of carbon dioxide mass per area per year. These EFs can be combined with estimates of changes in land allocation (e.g., a change in the crop planted on a land parcel or a change in land cover from pasture to cropland) from an economic model like POLYSYS (the agricultural economic model used to generate biomass supply estimates for in *BT16* volume 1) for different biomass-production scenarios to yield the GHG implications of large-scale feedstock-production increases.

Figure 4.2 and figure 4.5 illustrate how we have adopted this approach to estimate SOC EFs for application in *BT16* for agricultural crops. In particular, the SOC EFs (denoting SOC changes) can be estimated for each specific land-allocation change determined by POLYSYS (fig 4.5). These EFs were then combined with actual land-area change associated with each land-allocation change to calculate total SOC change for a specific scenario of land that transitioned from one type to another (fig 4.5). SOC changes associated with forestry systems are not quantified in this analysis. These species-dependent changes are influenced by many silvicultural management factors such as nutrient management and harvest method (Lal 2005). Moreover, the extent and composition of litter influence these changes. At present, there are not sufficient data and modeling capability to address SOC changes of forestry systems in the *BT16* although some considerations for the evaluation of soil organic carbon changes in forestry systems are provided in appendix 4-B. In future analyses, SOC changes in forestry systems for biomass production may be examined. In the following subsections, we first explain how we conducted SOC modeling with SCSOC. The next subsection describes how the SOC EFs are paired with output from POLYSYS that describes how land moved into and out of production of crops in the *BT16* scenarios.

4.2.1.1 Application of Soil Carbon Modeling to BT16 Agricultural Scenarios

Important inputs to the SCSOC model include bioenergy and other crop yields at the county level. Yield is a major factor determining above- and belowground biomass production which influence soil organic matter inputs. These inputs contribute to the accumulation of SOC. In SCSOC, historical conventional crop yields (e.g., corn, wheat, and soybean) are based on USDA-NASS statistical data (USDA 2015). The reference yields in the start year (2015) of the POLYSYS-modeled production period for energy crops such as switchgrass, miscanthus, poplar, and willow are based on the Climate Group's Parameter-elevation Relationships on Independent Slopes Model (PRISM). SOC and POLYSYS economic modeling to estimate land-allocation changes consistently use these yield inputs—this is important because yields drive results of both models, which are being used together. For the land-use change (LUC) period (2015–2040), biomass yield is determined by scenarios, with a 1% annual yield increase rate in BC1 and 3% in HH3, which are consistent with POLYSYS yield assumptions. In SOC modeling for the GHG emissions analysis, all conventional crops were grown with conventional tillage while most energy crops are modeled as being produced with no tillage. (*BT16* volume 1 modeling did include different tillage scenarios, and future work may refine treatment of tillage in SOC modeling.) SOC simulations also consider the potential impacts of erosion by applying the erosion rates for croplands and pasture, hay, and grasslands obtained from National Resources Inventory erosion estimates (Natural Resources Conservation Service), which are based on the Universal Soil Loss Equation (USLE) and the Wind Erosion Equation (WEQ) (Dunn et al. 2014). Climate-related inputs to SCSOC are based on county-level monthly temperature and precipitation data from weather stations between 1960 and 2010. Soil texture classes (e.g., sand, clay, and loam) within each county are determined from the Harmonized World Soil Database (Qin et al. 2016a).

Figure 4.5 | Schematic of data sources and estimations of SOC changes for agricultural feedstocks. POLYSYS estimates both land area change and allocation changes for each allocation change. (CCLUB – Carbon Calculator for Land Use Change from Biofuels Production)

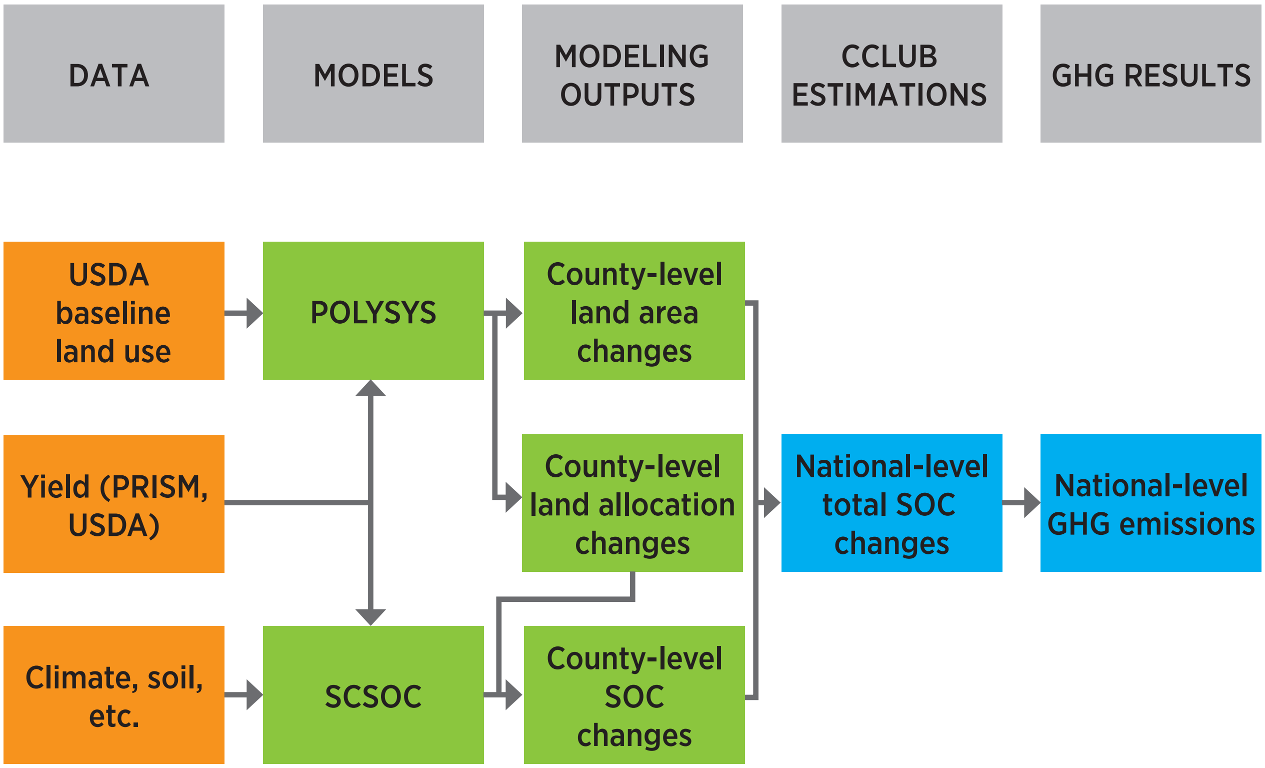


Table 4.1 lists the crops simulated in POLYSYS and how SOC changes associated with them are modeled with SCSOC. Crops that fall into the same crop cohort (e.g., barley, oats, or wheat) are simulated with comparable SOC-modeling settings with specific parameters describing biomass production and return (e.g., harvest index or residue return rate). Rice, eucalyptus, pine, and energy cane are not specifically modeled for SOC change since these crops are associated with less than 1% of the land area that underwent a land-management or land-cover change per POLYSYS outputs in both BC1 and HH3 scenarios.

The SOC model was run at a county level prior to 1881 until 2040 for each potential land transition from one use or land cover to another (e.g., pasture to miscanthus) to calculate the SOC change over the biomass feedstock production period (25 years).

The SOC change rate (SOCr) (Mg C ha^{-1} yr^{-1}), also referred to as SOC *EF*, indicates the average annual SOC change over time *(T)* (fig 4.6). A positive *SOCr* indicates a SOC loss while a negative value indicates a SOC gain.

For county *i* undergoing a given land transition (e.g., pasture to miscanthus) *j* over a number of years T (starting from time *0* to *T*):

Equation 4.1:

$$SOCr_{ij} = \frac{SOC_{0,ij} - SOC_{T,ij}}{T}$$

These county-level EFs were matched with associated amounts and types of changes in land allocation from POLYSYS. For example, the emission factor for a pasture-to-miscanthus production in Lyon County, Kansas, was applied to the 39,000 hectares that

Table 4.1 | Simulation of Crops in SOC Modeling

POLYSYS crops*	SOC modeling approach	Notes
Land use history (prior to 2015)		
- Barley, corn, cotton, oats, rice, sorghum, soybeans, and wheat	Cropland†	Cropland and pasture are assumed to represent historical land patterns according to the 2015 crop types in POLYSYS
- Hay, pasture	Pasture‡	
Land allocation (2015–2040)		
- Corn, soybeans, and wheat	Corn, soybeans, wheat †	Used existing SCSOC parameters
- Switchgrass and miscanthus	Switchgrass and miscanthus‡	Used existing SCSOC parameters
- Willow and poplar	Willow and poplar‡	Used existing SCSOC parameters
- Barley, cotton, oats, sorghum, and biomass sorghum	Barley (wheat), cotton (grass), oats (wheat), sorghum (corn), and biomass sorghum(corn) †	Crops are simulated under similar crop cohorts (in parentheses) with specific parameters (e.g., harvest index or residue return rate)
- Rice, eucalyptus, pine, and energy cane	N/A	Crops existed in POLYSYS but are not included in SOC modeling because of their insignificant contribution to overall shifts in land allocation
- Idle land	N/A	Land moving into and out of the POLYSYS idle land category was assumed to experience no SOC change because the idle land category has no specified characteristics or classification regarding vegetation growth or residue management

* Included only crops associated with land use change. NA, not applicable. †Crops under conventional tillage and ‡no tillage as assumed for analysis for this chapter only.

experienced this transition between 2015 and 2040. Application of POLYSYS output for this purpose is described further in the next section. The total SOC change (Mg C) in county i associated with biomass production until t_x (which is 2040) is calculated as equation 4.2.

Equation 4.2:

$$\Delta SOC_{Total,i} = \sum_{j=1} SOCr_{i,j} \cdot A_{i,j} \cdot P_{i,j} \cdot (t_x - 2015)$$

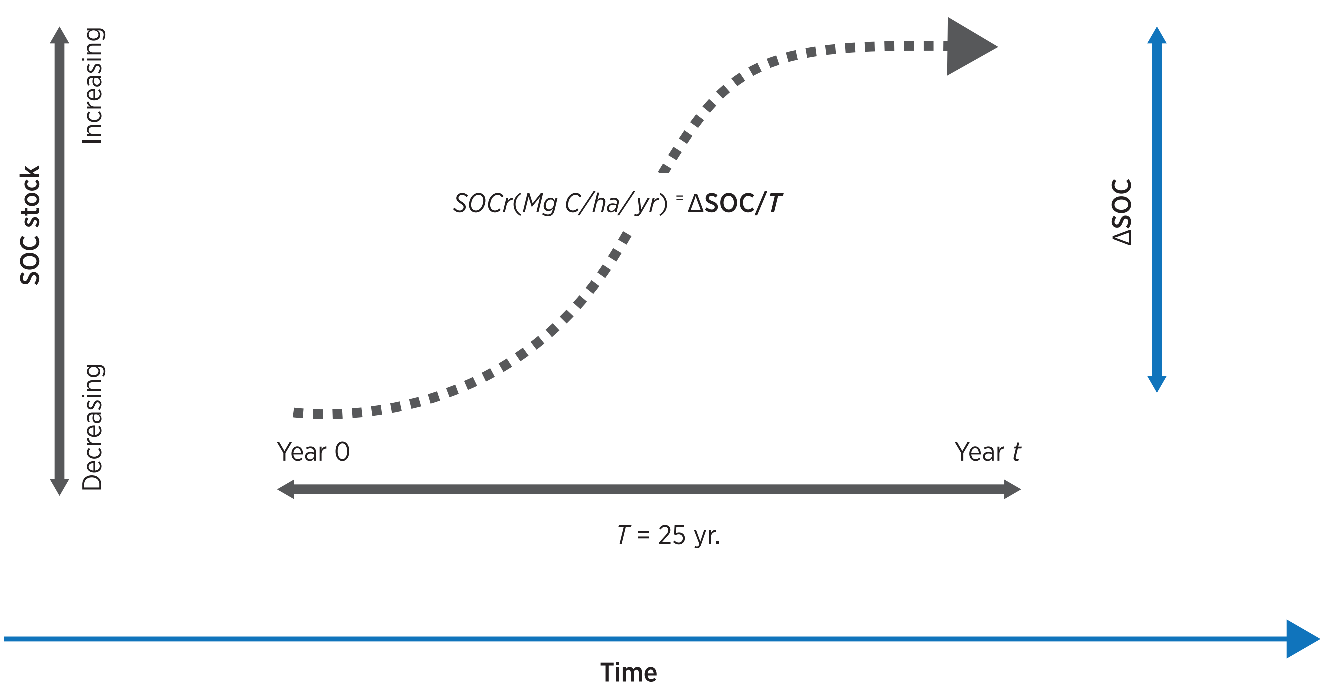
In this equation, A is the land area, P is the probability of a certain land transition (e.g., pasture to miscanthus) (ΣP_i=1, see next section) and t_x is the target biomass production year (here 2040). This calculation produces the total SOC change over the 25-year production period, which is divided by total agricultural biomass production in the county over the same period. All biomass produced in a given county then, regardless of type, is assigned the same SOC change intensity (SOC per unit biomass basis). A positive SOC change value indicates net carbon loss, and a negative value indicates net carbon gain. The SOC change, in terms of carbon, is converted to GHG emissions by a factor of carbon content in carbon dioxide (44/12). For detailed SOC model descriptions, please refer to our earlier publications (Kwon et al. 2013; Dunn et al. 2014; Qin et al. 2016a).

4.2.1.2 Applying POLYSYS Outputs to Estimating County-Level SOC Changes in BT16 Scenarios

There are two key challenges in using land allocation outputs from POLYSYS to model SOC changes relevant to *BT16* scenarios. These challenges and the techniques devised to overcome them are conceptualized in figure 4.7. The first challenge is that the SCSOC model relies on information about land-use history going back more than 100 years (fig. 4.7*A*). POLYSYS does not consider land-use history, but only begins tracking areas of land in a given county planted with certain crops at the start of the simulation (i.e., 2015). The second challenge is that POLYSYS does not keep track of the changes in the land allocation or cover of any given parcel of land at a sub-county level over time after 2015 (fig. 4.7*B*). Rather, the model output contains the area of land in one county planted in any given crop each year. If land area planted with corn decreases, that decrease may represent land newly planted with soy or with switchgrass, for example. This feature of POLYSYS output presents a challenge for SOC modeling that relies on information about the change in land use for a single parcel of land over time.

Regarding the first challenge pertaining to land-use history, SCSOC needs to adopt a historical land-use pattern without complete information from POLYSYS. In previous analyses (e.g., Qin 2016a), the land-use history, which strongly influences results, was originally constructed for simulating historical SOC dynamics by dividing the entire simulation of the land's history prior to the year the land undergoes a change in allocation into three periods: pristine prior

Figure 4.6 | SOC stock change and change rate



to 1881, 1881-1950, and 1951 to present (e.g., 2010) (fig. 4.7). Pristine land use is either grassland for native grassland and permanent pasture or forest for all forest cover. These land-history patterns are designed to represent major land uses over time as well as to capture SOC changes over a relatively long time period—SOC pools are not stable under short, frequent changes in land use. In the BT16 analysis, to overcome the first challenge, two major land-use types are assumed to represent historical patterns according to the land allocation in 2015 (fig. 4.7C). Based on earlier simulations of land-use history (Kwon et al. 2013; Qin et al. 2016a), the first, historical cropland includes all conventional crops in POLYSYS (e.g., corn, soybeans, wheat, and oats) and the second, pasture, is used for pasture and hay. Sensitivity of results to land-use history can be explored in future work.

Regarding the second challenge, POLYSYS outputs are used to generate probability matrices for feedstock production between 2015 (the year in which POLYSYS simulations begin) and 2040. The probability describes the distribution of designated

Figure 4.7 | Conceptualization of land-use/land-allocation change in simulations in different modeling systems A) LUC modeling framework in previous studies with land use history included (Kwon et al. 2013; Qin et al. 2016a); B) POLYSYS output in the form of annual county-level land-use matrices; and C) the land patterns used in this analysis to capture both land-use history and longer-term (25-year) land-use matrices from POLYSYS. Each row represents one unit of land experiencing changes of land use through time. The pixel color indicates a specific land use during different time periods.

Table 4.2 | Land-Area and Land-Use Allocation Pattern Outputs in POLYSYS

County	Land area	Initial (2015) allocation	Final allocation	Probability
1	A1	Corn	Switchgrass	P_1
1	A1	Corn	Miscanthus	P_2
1	A2	Soybeans	Switchgrass	P_3
-	-	-	-	-

land allocations (e.g., agricultural land planted with switchgrass or willow) originating from each of the 2015 land allocations at the county level (Table 4.2). This approach does not take into account the many potential changes in land allocation in a county over the 25-year time horizon of this analysis, but allows SOC to approach a relatively stable state so that a reasonable emission factor can be modeled for lands that changed initially from cropland or pastureland. While this analysis adopts a 25-year time horizon given the parameters of the *BT16* study, in previous analyses (e.g., Qin et al. 2016a), researchers chose a 30-year time horizon for biofuel feedstock production to match the time horizon that the U.S. Environmental Protection Agency (EPA) uses in its modeling for the Renewable Fuel Standard analysis (EPA 2010). A 30-year time horizon was also chosen because SOC typically returns to equilibrium within 30 years following a land transition (Qin et al. 2016b). An exception is if forested land is cleared and planted in corn; in that case, SOC can take many decades to stabilize. Forest land, however, is restricted from transition to agricultural land in *BT16* as described in volume 1.

Readers should keep in mind these two key limitations involved in estimating SOC changes associated with *BT16* scenarios. The SOC changes reported here should be viewed as estimates that indicate the directionality and estimated magnitude of SOC changes associated with the specific *BT16* scenarios rather than as a prediction of SOC that would exactly occur at the county level or would occur as compared to a business as usual scenario. Future work may investigate sensitivity of results to the key assumptions including land history prior to allocation change and tillage practice. Alternative techniques in using POLYSYS output to generate estimates of SOC changes may also be examined.

4.2.3 Changes in Aboveground Carbon

Potential aboveground carbon changes of the select scenarios are not considered in this chapter. Of all potential land transitions, clearing forested land to grow crops incurs the most significant amount of aboveground carbon change. The carbon stock in the trees is lost, and then every year, some amount of carbon that would have been sequestered and added to the existing carbon stock is not sequestered (Dunn et al. 2013). This latter missed opportunity to capture atmospheric carbon is called foregone sequestration. However, this type of land-allocation change would not occur under the *BT16* scenarios because of modeling constraints placed upon POLYSYS and ForSEAM as described in volume 1 that preclude the exchange of land between forestry and agriculture.

The types of land allocation changes that are simulated in the *BT16* scenarios – land use shifts within the agricultural sector - are not likely to cause significant changes in aboveground carbon. The primary land cover types in the POLYSYS 2015 start year prior to transition are cropland and pastureland. In the case of agricultural land, crops are harvested annually, and there is not a significant carbon stock on the land to be lost. Similarly, pastureland undergoes an annual cycle in the amount of biomass because significant portions of aboveground biomass are lost due to grazing, fire, and natural death. There is no foregone sequestration in either case because there is little stable, existing carbon stock on the land to continue to build as is the case in forests. Therefore, the only significant change in aboveground carbon stock is the loss of any initial carbon stock, which could be amortized over the period of study, which in this case is 25 years. One interesting case is the conversion of pastureland or cropland to the production of short-rotation woody crops. In this situation, the aboveground carbon stock is likely built over time as the woody crop sequesters carbon, but this sequestration is much shorter-lived than it would be for tree species with longer rotation lengths.

It is important to note that the two challenges that impact the estimation of SOC changes would affect the estimation of aboveground carbon changes if it were undertaken in this analysis. The first challenge of not knowing the land-use history prior to 2015 precludes knowledge of the aboveground carbon stock at the time of the change in land allocation. Furthermore, the absence of dynamic POLYSYS output regarding the progression of what is planted on any given sub-county parcel of land over time translates into a lack of information regarding how carbon stocks change on that parcel of land.

4.2.4 Representative Bioeconomy Cases

The analysis presented in this chapter is limited to the system boundary in figure 4.1, which ends after feedstock logistics and transportation. This system boundary does not enable analysis of the extent to which using biomass feedstocks for fuel, power, and chemicals offers a potential GHG benefit on a life-cycle basis relative to using fossil-derived feedstocks. Investigating this question requires evaluation of cases that specify the end uses of the biomass produced. To address this question, this chapter references an analysis undertaken by Rogers et al. (2016) to assess the size and benefits of a Billion-Ton Bioeconomy (BTB). The intent of assessing the GHG and energy impacts of the BTB cases is to provide the full life-cycle GHG and energy impacts of using the amount of biomass produced in the *BT16* scenarios.

For the purposes of the BTB analysis, the "bioeconomy" describes the integral role of abundant, sustainable, and domestically produced biomass (agriculture and forestry-derived) in producing biofuels, generating bioheat and power, and producing renewable chemicals and other bio-based compounds to grow the U.S. economy. It is important to note that although Rogers et al. (2016) considered several cases for biomass end uses including a base case, and cases in which ethanol, jet fuel, biopower, and bioproducts were prioritized for biomass use, the biomass produced could be used for any number of purposes. Furthermore, the analysis did not consider indirect effects or overall demand associated with price changes of biomass and fossil based feedstocks. The BTB analysis adopted two levels of biomass availability in the year 2030 based on the low- and high-yield *BT16* scenarios for that year.

To estimate GHG emissions and energy benefits associated with the BTB cases, a tool called Bioeconomy Air and Greenhouse Gas Emissions (AGE) was developed to estimate the energy, air quality, and GHG impacts of the bioeconomy cases as compared to an "all fossil" baseline case (Rogers et al. 2016). AGE includes *BT16* and BAU parameters for biofuels, conventional fuels, biopower, and biochemicals. The AGE tool allocates biomass by feedstock type to production of different types of biofuels, bioproducts, biopower, and steam. The AGE tool estimates the production amounts of biofuels, bioproducts, biopower, and steam using conversion factors or yield assumptions from specific biomass feedstock types to end products and calculates the amounts of conventional fuels, products, power, and steam that are displaced. AGE calculates the total energy consumption, GHG emissions, and air pollutant emissions in each scenario and its respective "all fossil" scenario on the basis of life-cycle GHG emissions and energy consumption for conventional fuels, biofuels, biopower, conventional power, biochemicals, and conventional chemicals generated by GREET. Using GREET as an AGE parameter source ensures a consistent basis for analysis of biofuels, conventional fuels, and other end products with the rest of the *BT16* GHG analysis. In this application of Bioeconomy AGE to the *BT16* analysis, the feedstock GHG emissions as presented in the above section are used to override GREET default values in Bioeconomy AGE to estimate the life-cycle GHG emissions of biofuels, bioproducts, biopower, and steam derived from various feedstocks in *BT16*. The total biomass tonnage by biomass type used for these scenarios is only the biomass delivered to the reactor throat at less than $100 per dry ton. However, the GHG emissions for the production of all biomass available from *BT16* volume 1 is used, even if their logistics is too cost prohibitive for delivery to the reactor throat.

4.3 Results

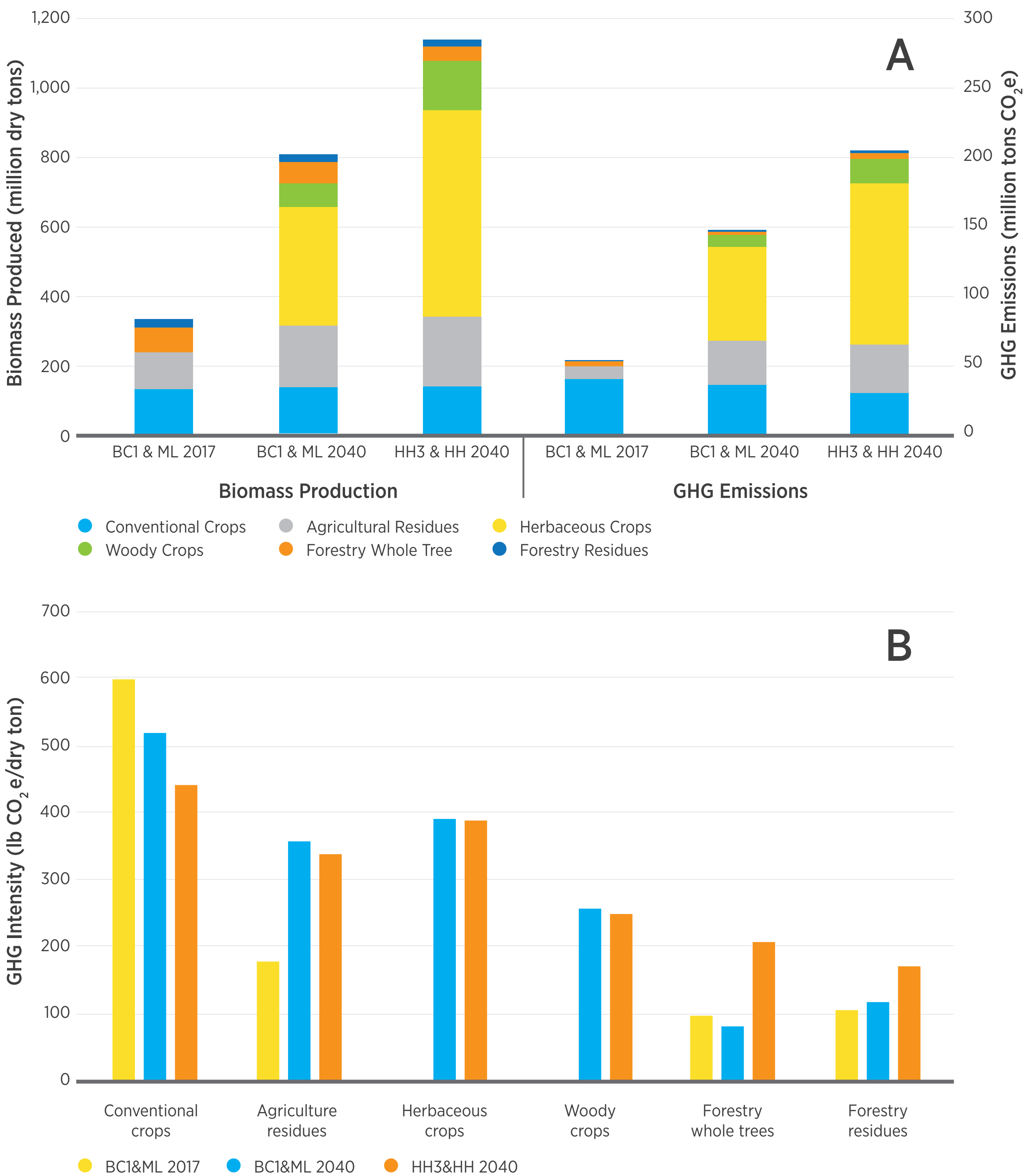
Results are presented in three sections. First, energy consumption and GHG emissions associated with forestry and agricultural operations are described. Next, section 4.3.2 describes SOC changes at a county level associated with changes in land allocation in the agricultural sector based on the POLYSYS modeling in volume 1. Finally, section 4.3.3 combines operational and SOC change-related GHG emissions to describe at a county level the net GHG emissions associated with the 2040 base-case and high-yield scenarios developed in volume 1.

4.3.1 Energy Consumption and GHG Emissions Associated with Forestry and Agricultural Operations and Logistics

Figure 4.8A displays the breakdown of biomass produced nationwide and associated GHG emissions under the BC1&ML 2017, BC1&ML 2040, and HH3&HH 2040 scenarios. On the national scale, the GHG intensity (GHG emissions divided by the total produced biomass) is 331, 364, and 359 lb CO_2e per dry ton of total biomass, for the BC1&ML 2017, BC1&ML 2040, and HH3&HH 2040 scenarios, respectively. The GHG intensity is lower under the high-yield scenario in 2040 compared to the base-case scenario in 2040 because feedstock yields in the HH3&HH 2040 scenario are higher while some of the agricultural inputs per acre stay constant (e.g., fertilizer application rate or diesel consumption in harvesting).

Figure 4.8*B* provides the GHG intensity for producing each feedstock type for all three scenarios. Conventional crops have a higher GHG emissions intensity than all other feedstocks, which decreases between the 2017 and 2040 scenarios because yields increase. The herbaceous crops' GHG emissions intensities only slightly decrease between the base-case and high-yield 2040 scenarios, (392 compared with 390 lb CO_2e per dry ton) because most of the inputs for these feedstocks are applied on a per dry ton of biomass harvested and are not affected by higher yields in the HH3 2040 scenario. Woody crops see the same trend, with intensities of 258 lb and 250 lb CO_2e per dry ton for the base-case and high-yield 2040 scenarios, respectively. The contribution of agricultural residues to potential biomass produced in 2017 is higher than the contribution of this feedstock type to GHG emissions (fig 4.8A). In the 2040 scenarios, however, shares of total biomass tonnage and GHG emissions contributed by agricultural residues are roughly equal. This increased intensity is caused by a shift from conventional logistics in 2017 to advanced logistics in 2040. Advanced logistics, used to pelletize biomass at a regional depot, consume more energy than conventional logistics per ton of biomass. See *BT16* volume 1, chapter 6, for a full discussion of logistics operations in 2017 as compared to 2040. A summary of logistics modeling assumptions is provided in chapter 2 of this volume.

Figure 4.8 | Potential biomass production and GHG emissions (all production emissions and only feasible logistics emissions) (A) and GHG intensity (B) by crop type. Conventional crops (e.g., corn and soybeans), agricultural residues (e.g., corn stover, wheat straw, oat straw, sorghum stubble, and barley straw), herbaceous crops (e.g., switchgrass, miscanthus, energy cane, and biomass sorghum), woody crops (e.g., poplar, willow, loblolly pine, and eucalyptus), and forest biomass (e.g., hardwoods, softwoods, mixed woods) are included.



Forestry whole tree biomass has a lower share of GHG emissions than does agricultural biomass. The GHG intensities for the production of forestry biomass are lower than other crops because not all forestry plots are subject to site preparation, which consumes diesel fuel, and because fertilizers are either not used or are used more sparingly than they are for agricultural crops. On a per-dry-ton basis, logistics operations and corresponding emissions are roughly equivalent between whole-tree feedstocks and non-crop agricultural feedstocks and are not a reason for differing GHG intensities of forest-derived and agricultural feedstocks. One difference regarding forestry whole-tree biomass is that the GHG-emission intensity is lower for the BC1&ML 2040 scenario compared to the BC1&ML 2017 scenario, which is due to the logistics stipulation leaving any biomass with a delivered cost of more than $100 per dry ton on the field. There are more instances of biomass left on the field for the 2040 base-case scenario compared with the 2017, and as a result, the energy-intensive GHG emissions of advanced logistics are not included for this biomass. Overall, this analysis finds that forest residues are a minor contributor both to biomass tonnage and GHG emissions.

Figure 4.9 displays the breakdown of total potential biomass production and GHG emissions from producing the biomass associated with *BT16* scenarios BC1&ML 2017 by FRR (regions depicted in fig 4.10) in the BC1&ML 2017 scenario. In nearly every FRR, GHG emissions are dominated by conventional crops. FRR 7, the heart of corn and soy production in the United States, could potentially have the highest level of GHG emissions GHGs compared to other FRRs. The FRR that exhibits the second-highest modeled GHG emissions is in the North Central United States (FRR 9). FRRs 4 and 10–13, which have whole trees as the dominant feedstock type, would not be significant contributors to national GHG emissions from biomass production. In 2017 scenarios, herbaceous crops and forestry crops do not contribute to GHG emissions in any FRR, as they are not yet produced.

In the BC1&ML 2040 scenario (fig 4.11), the contribution of herbaceous crops and residues rise, compared to the 2017 scenario, especially in the Central Plains, including a large part of Texas (FRRs 7 and 8). FRR 5—which includes Tennessee, Kentucky, and West Virginia—also exhibits notable GHG emissions from herbaceous crop production. On the other hand, the western United States sees little biomass production and, correspondingly, low GHG emissions associated with biomass production in the BC1&ML 2040 scenario.

In the HH3&HH 2040 scenario (fig 4.12), the main FRRs contributing to GHG emissions do not change, but emissions associated with producing herbaceous crops and agricultural residues experience the most significant increases in FRRs 7 and 8. It is increased production of these energy grasses that, in fact, drive increased emissions between BC1&ML and HH3&HH scenarios for 2040.

The estimated GHG intensity of producing each feedstock for all three scenarios (not including transportation emissions) is presented in figure 4.13. Annual crops, corn and especially soybeans, would have much higher GHG emissions per dry ton than the crop residues regardless of yield scenarios. This is mainly a result of agriculture diesel and fertilizer consumption. For conventional crops, diesel and fertilizers are needed for soil preparation, planting, and harvesting, while using agriculture residues as biomass results in limited fuel consumption for residue collection and fertilizer consumption only to replace the nutrients lost due to residue removal.

For herbaceous and woody crops, estimated GHG intensities fall mostly below 200,000 g-CO_2e per dry ton, although willow and poplar in the BC1 2040 scenario (fig 4.13B) have larger variations in GHG intensity than other biomass types. For these two feedstocks, the fertilizer and diesel inputs are based

Figure 4.9 | Estimated total GHG emissions (A) and biomass production (B) in each FRR by crop type for BC1&ML 2017. Conventional crops (e.g., corn and soybeans), agricultural residues (e.g., corn stover, wheat straw, oat straw, sorghum stubble, and barley straw), herbaceous crops (e.g., switchgrass, miscanthus, energy cane, and biomass sorghum), woody crops (e.g., poplar, willow, loblolly pine, and eucalyptus), and forest biomass (e.g., hardwoods, softwoods, mixed woods) are included.

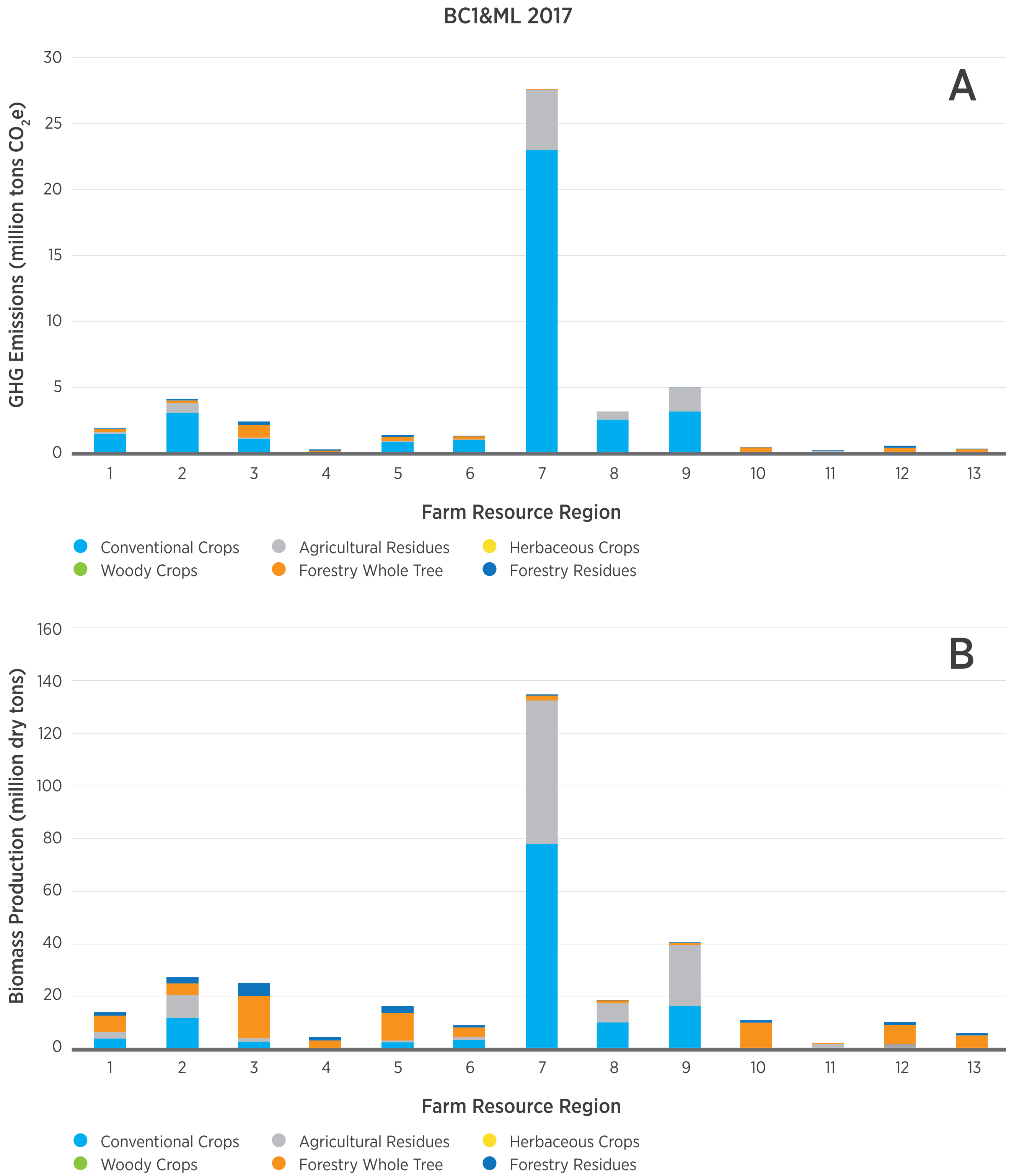
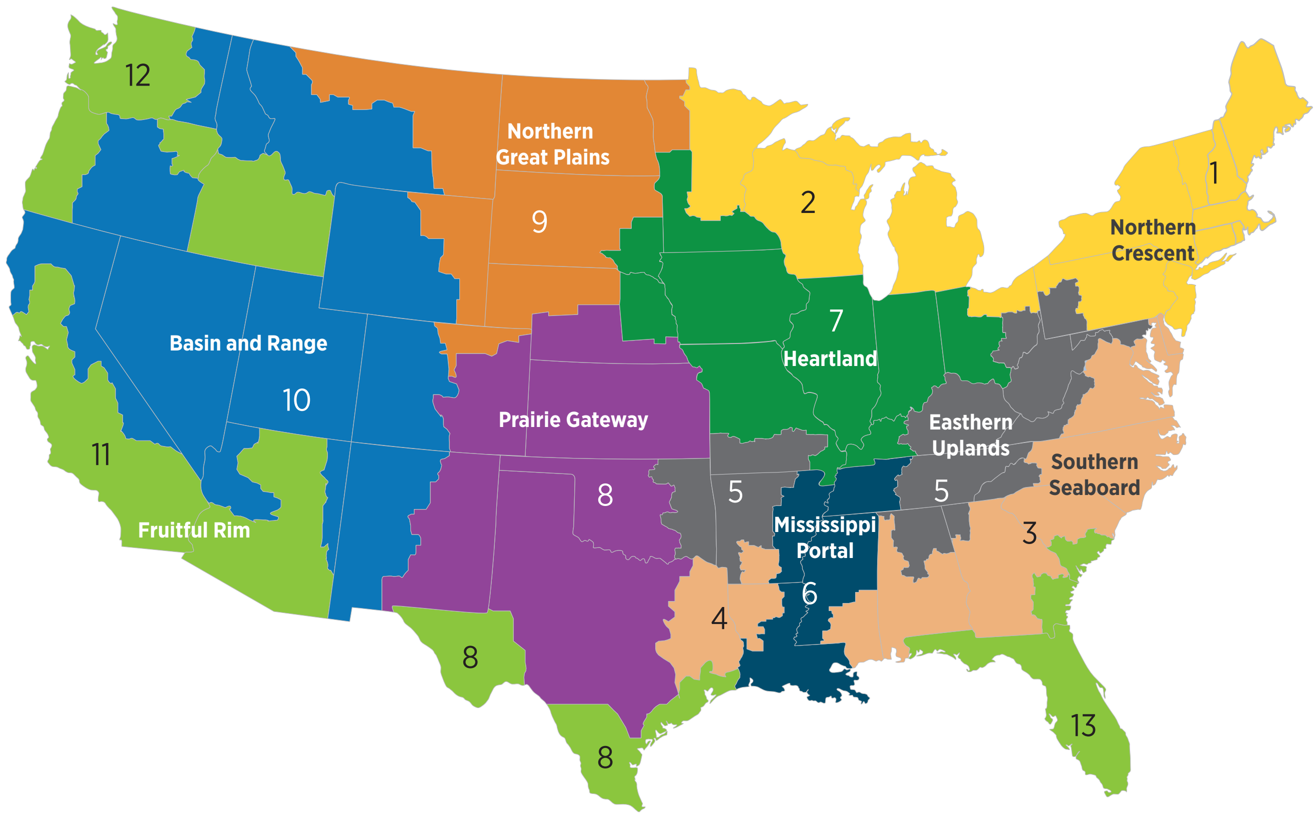


Figure 4.10 | USDA Farm Resource Regions



on planted acres which, in some instances, greatly exceed the harvested acres. For example, in Lincoln County, Colorado, more than 23,000 acres would be planted in poplar, but only 2,300 of those acres are harvested because not all acres had reached the end of the rotation. GHG emissions reported herein include diesel and fertilizer consumption for planted acres. In counties such as Lincoln County, Colorado, with a low harvested-to-planted acres ratio, GHG intensity would therefore be high. Another factor influencing GHG intensity is biomass yield, in large part because, as described, some FRR budgets report fertilizer and fuel inputs on a per-acre basis. When yields are high, GHG intensities are lower compared to counties with lower yields. For poplar in the BC1 2040 scenario (fig 4.13*B*), the county-level harvested yields range from 17–67 dry tons per acre in that year, while in the HH3 2040 scenario (fig 4.13*C*) the harvested yields range from 19–89 dry tons per acre. The states with the highest harvested poplar yields include Georgia, Indiana, and Kentucky. In both the BC1 and HH3 2040 scenarios, Harlan County, Kentucky, has the highest harvested poplar yield at 67 and 89 dry tons per acre, respectively. However, in the BC1 2040 and HH3 2040 scenarios, respectively, Harlan County, Kentucky, contributes only 390 and 510 dry tons of poplar biomass. As a result, the relatively low GHG intensity of potentially producing poplar in these counties is not a major driver of GHG results. In fact, the bulk of poplar production in the 2040 scenarios comes from counties with GHG intensities for poplar production that fall toward median GHG intensities for producing this type of biomass.

Figure 4.11 | Estimated total GHG emissions (A) and biomass production (B) in each FRR by crop type for the BC1&ML 2040 scenario. Conventional crops (e.g., corn and soybeans), agricultural residues (e.g., corn stover, wheat straw, oat straw, sorghum stubble, and barley straw), herbaceous crops (e.g., switchgrass, miscanthus, energy cane, biomass sorghum), woody crops (e.g., poplar, willow, loblolly pine, and eucalyptus), and forest biomass (e.g., hardwoods, softwoods, and mixed woods) are included.

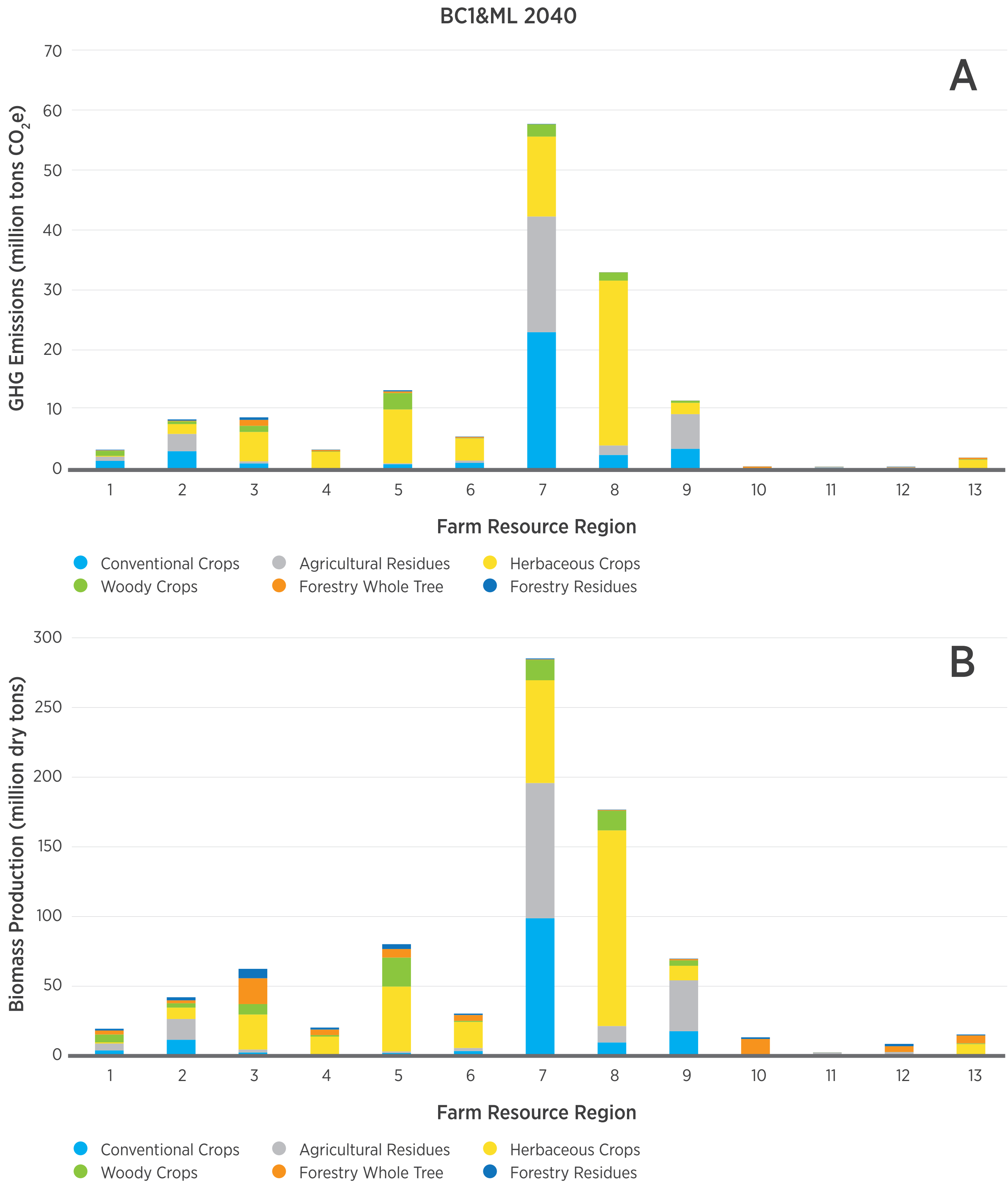


Figure 4.12 | Estimated total GHG emissions (A) and biomass production (B) in each FRR by crop type for the HH3&HH 2040 scenario. Conventional crops (e.g., corn and soybeans), agricultural residues (e.g., corn stover, wheat straw, oat straw, sorghum stubble, and barley straw), herbaceous crops (e.g., switchgrass, miscanthus, energy cane, and biomass sorghum), woody crops (e.g., poplar, willow, loblolly pine, and eucalyptus), and forest biomass (e.g., hardwoods, softwoods, and mixed woods) are included.

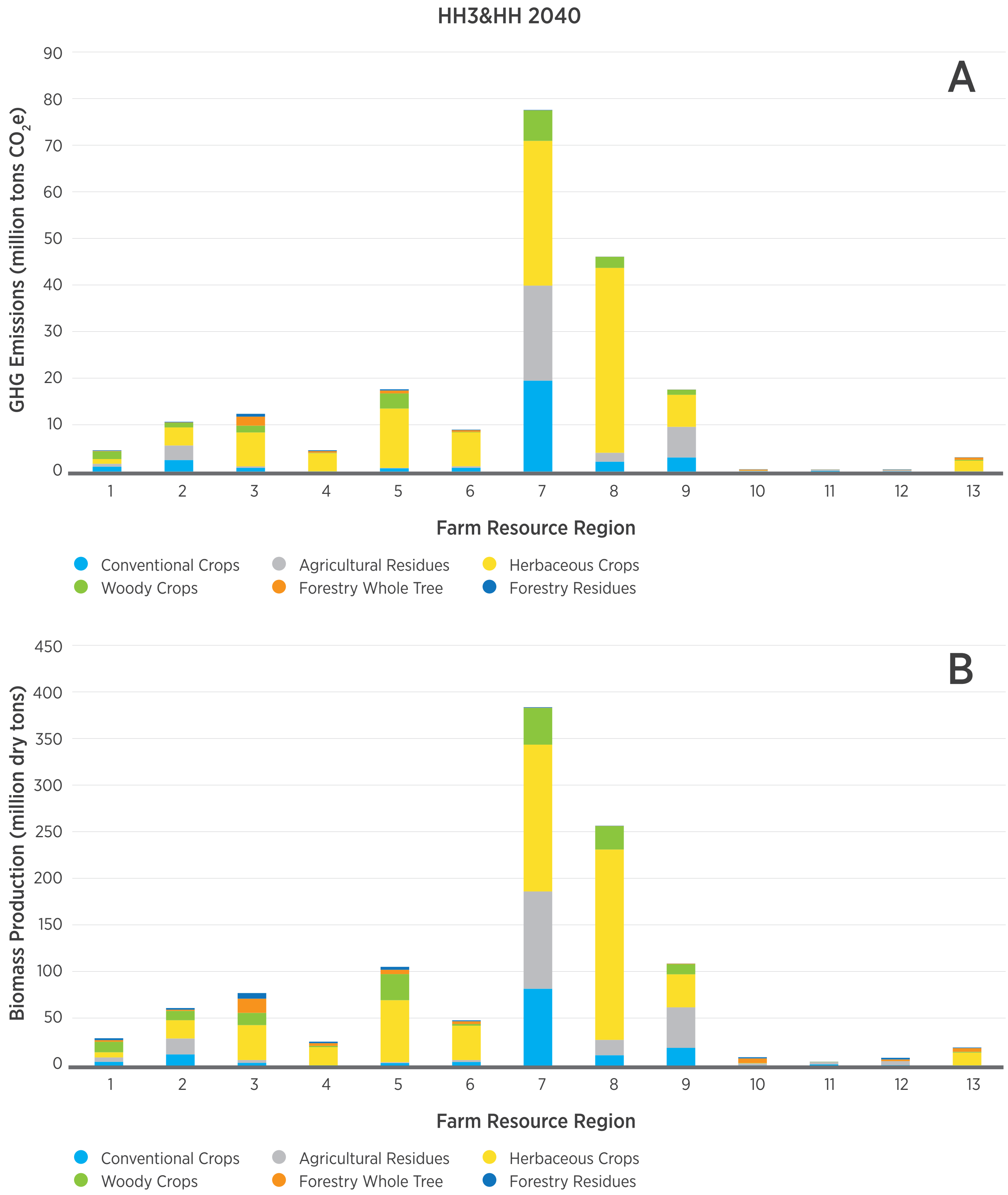


Figure 4.13 | Estimated intensity of GHG emissions-associated agricultural activities, including operations and logistics, under three scenarios. (A) BC1 2017, (B) BC1 2040, and (C) HH3 2040. Herbaceous and woody energy crops are not available in 2017 (a). In the boxplot: the box limits indicate the 25th and 75th percentiles, center line shows the median, whiskers are 1.5 times the interquartile range, and the box width is proportional to square-root of the number of observations. The number "1" denotes crop grain (for annual crops) or tree (for wood), and "2" denotes crop or tree residues.

Acronyms: COR – corn; SOY – soybeans; BAR – barley; OAT – oat; SOR – sorghum; WHE – wheat; BIO – biomass sorghum; ENE – energy cane; MIS – miscanthus; SWI – switchgrass; EUC – eucalyptus; PIN – pine; POP – poplar; WIL – willow; HLO – hardwood lowland; HUP – hardwood upland; MIX – mixed wood; SNA – softwood natural; and SPL – softwood planted.

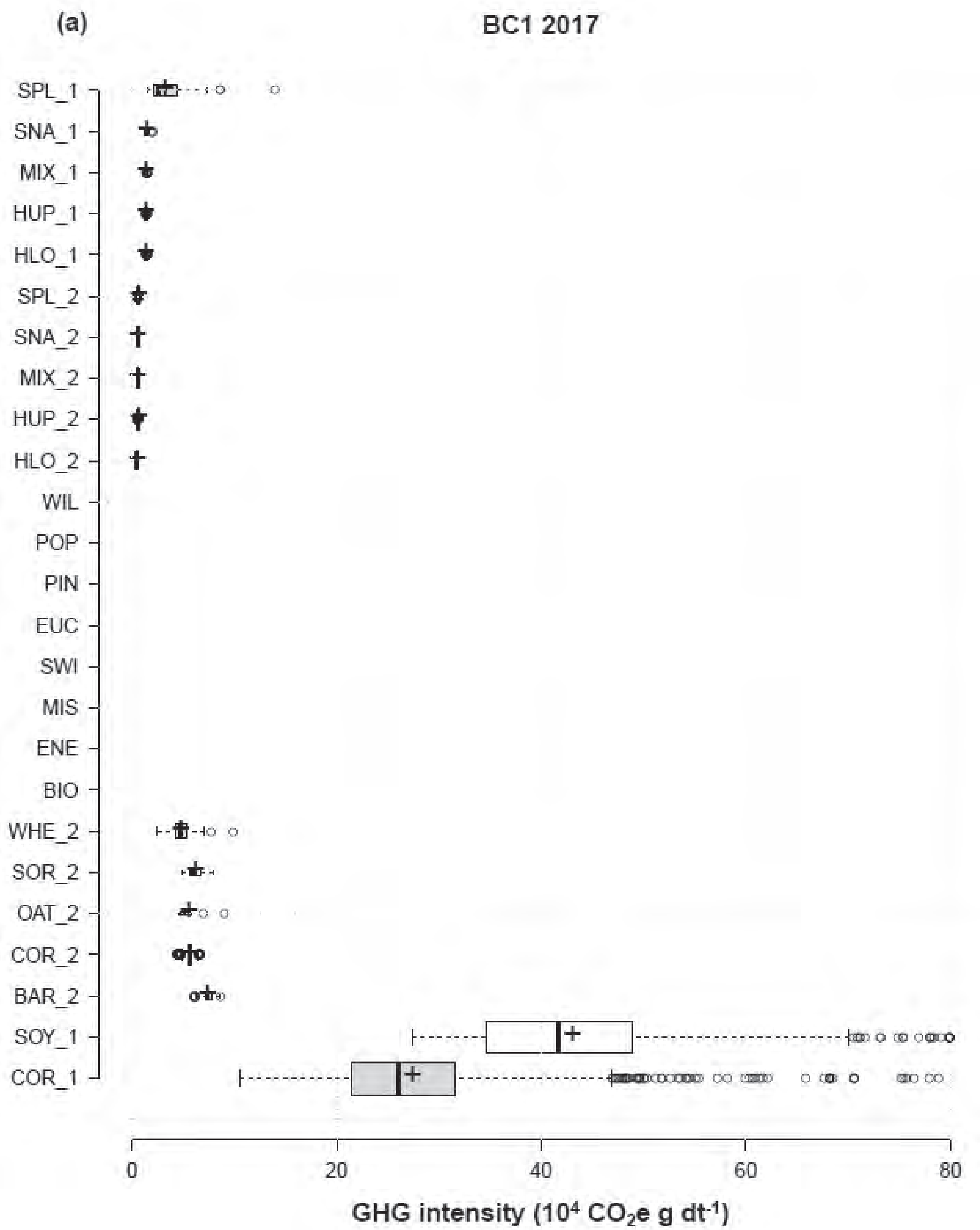


Figure 4.13 | *continued*

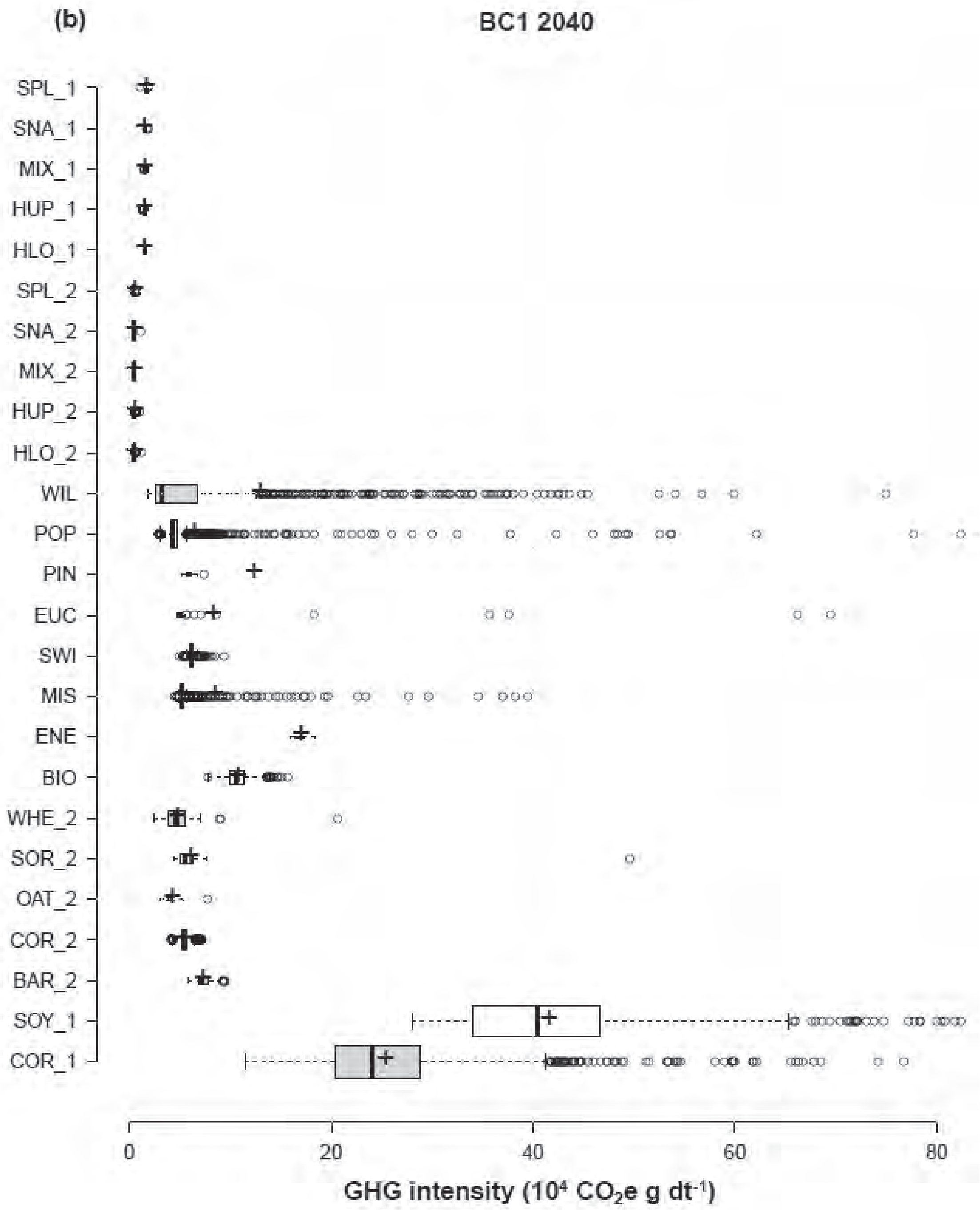
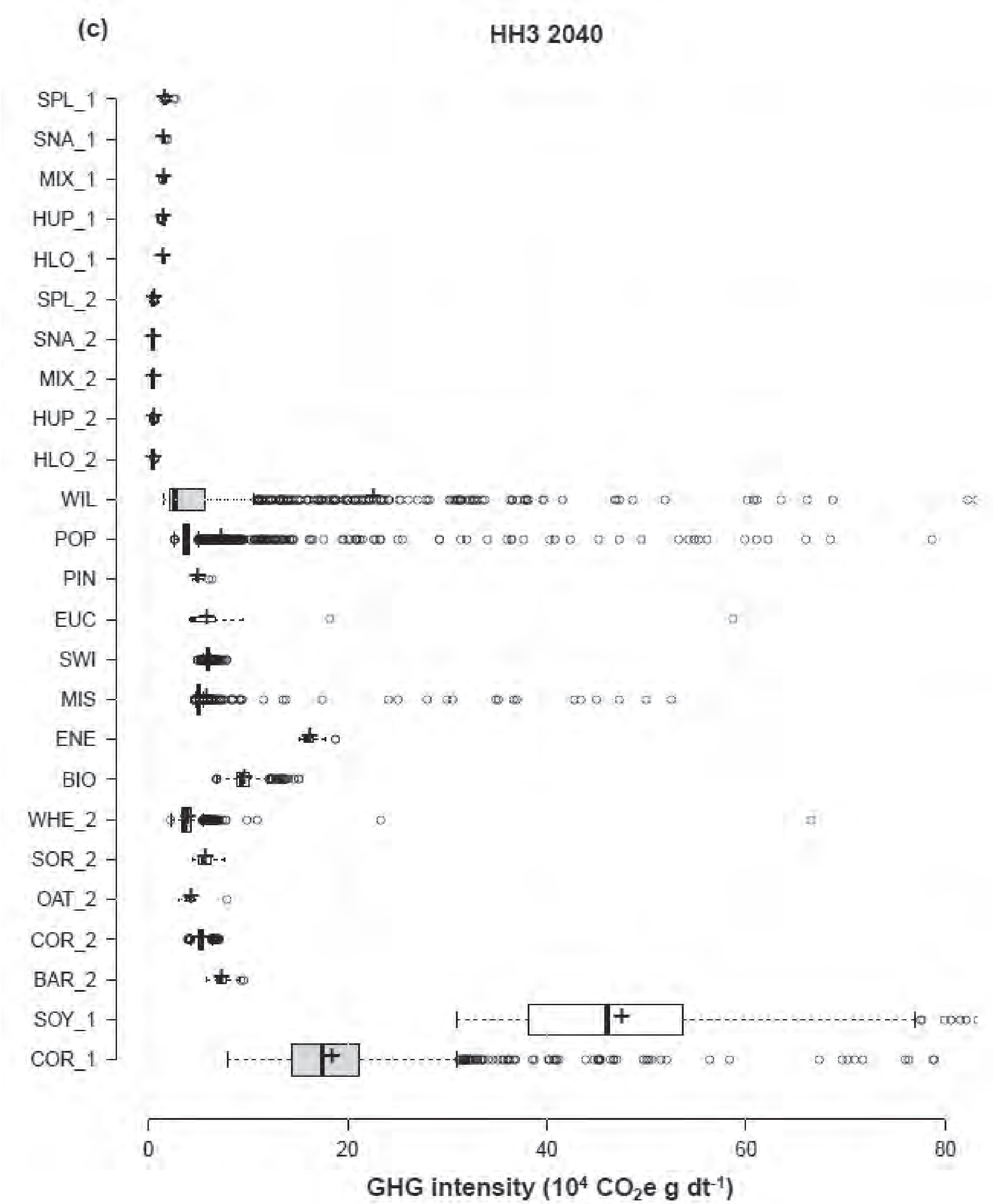


Figure 4.13 | *continued*



In the BC1 2040 scenario (fig 4.13*B*), the harvested yields for willow are slightly less variable than for poplar, ranging between 10–36 dry tons per acre. Potential willow yields also exhibit less variation than poplar yields (10–45 dry tons per acre) in the HH3 2040 scenario (fig 4.13*C*). Switchgrass harvested yields range from 1.1–9.8 dry tons per acre in the BC1 2040 scenario and from 1.2–14.1 dry tons per acre in the HH3 2040 scenario. It should be noted that these harvested yields are lower than those for poplar and willow because switchgrass is harvested every year while poplar and willow are harvested every 4 and 8 years, respectively. The GHG-intensity range for switchgrass (fig 4.13 *B* and *C*) is smaller than the range for the short-rotation woody crops (SRWCs) both because of this narrower yield range and because a good portion of fertilizer consumption for willow is independent of yield (applied on a per-dry-ton basis). In short, when agricultural inputs are yield-dependent, the variation in potential biomass yield seen in different counties across the United States has a significant influence on the range of GHG intensities for any one type of biomass.

For forestry-derived biomass, the diesel that would be consumed during harvesting and collection is the main contributor to GHG emissions. The GHG intensity for forestry residues ranges between 5,200 and 6,400 g-CO_2e per dry ton for all three scenarios (fig 4.13 *A*, *B*, and *C*). The only input for residues is fuel, which is used on a per-dry-ton basis. Some variation by location is based on the type of equipment used (medium versus large chipper, and small versus medium loader). For whole-tree harvesting, the simulated GHG intensity ranges from 14,000–15,000 g-CO_2e per dry ton. However, for the softwood planted whole trees, additional fuel would be consumed, and fertilizer is applied during site preparation, which could result in GHG intensities as high as 415,000 g-CO_2e per dry ton. For the softwood-planted biomass types, especially under the ML 2017 scenario (fig. 4.13A), there is a much larger variation in the GHG emissions per-dry-ton values because of significant variation in the quantity of biomass harvested per acre. Again, the same amount of fertilizer and chemicals are consumed per acre regardless of yield in each county, so counties with high harvested biomass per acre see less-GHG-intensive softwood biomass. If production per acre is low, GHG intensities run higher. Both the ML and HH 2040 scenarios have some counties with small amounts of biomass harvested per acre, but not to the same degree as ML 2017. Again, changes in aboveground biomass for forestry-derived feedstocks were not considered because the amount of forested land did not change, given restrictions placed on transitions between agricultural and forested lands in volume 1. If the amount of forested land did change or significant changes in forest management practices occur, this would result in changes in above ground carbon, and additional considerations inherent to temporal forest carbon analyses would need to be adopted into the analysis. These considerations, which include the spatial scale and the timing of emissions pulses, are described in detail in Lamers (2013), EPA (2014), and Daystar (2016) among other references.

4.3.2 GHG Emissions from SOC Changes on Agricultural Lands

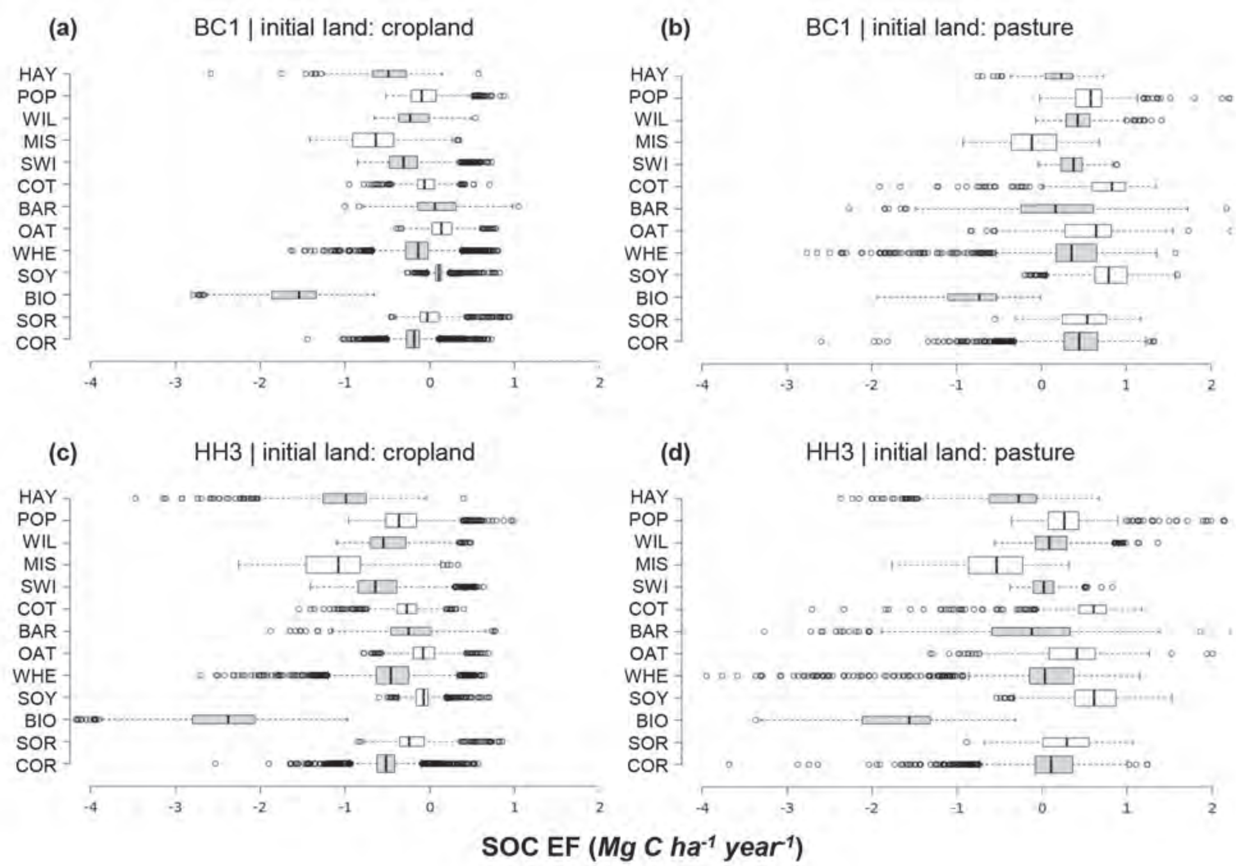
According to POLYSYS simulations, the area of land that would be allocated to different uses in 2040 as compared to 2015 totals about 41 and 50 Million hectares (Mha) under BC1 and HH3 scenarios, respectively. Overall, cropland and pasture areas would decline, while areas planted in major energy crops would expand on net (chapter 3). Most of the land producing major energy crops in 2040 scenarios is either pasture or one of the three major cropland types of corn, soybeans, or wheat. Under the 2040 BC1 scenario, these land types are mainly allocated to switchgrass (27%) and miscanthus (20%); corn and poplar each share about 9% of the total amount of transitioned land. However, under the HH3 scenario, crop management on half of these lands would be altered to grow miscanthus (30%) and switchgrass

(19%), and another 19% to poplar (10%) and willow (9%). Regardless of the scenarios, only a very small amount (less than 0.2% each) of the lands would be planted in eucalyptus, pine, or energy cane. Barley, oats, rice, and hay each share less than 1% of the total land converted. Note again that POLYSYS contains a land category termed "idle" that is used as a pool to balance total land-use transitions. This analysis assumes land transitioning into and out of this category—a sizeable quantity—does not experience SOC change because it is not a land category in practice, and therefore, it is very difficult to establish a reasonable land-use history to inform SOC modeling.

The estimated SOC change varies spatially and among different land transitions. In general, when cropland represents the initial 2015 land allocation, SOC EFs are lower than when pasture represents the initial land allocation (fig 4.14). On average, growing energy crops on historical cropland typically leads to soil carbon gains (fig 4.14*A* and *C*). When pasture is used to produce biomass, however, only a few energy crops such as miscanthus and biomass sorghum, which both have high biomass yields, are estimated to sequester carbon in soil (fig 4.14*B* and *D*). Biomass yield is a key factor in determining the SOC balance. Often, high yield means more biomass can

Figure 4.14 | Soil organic carbon EFs for lands transitioned from initial 2015 cropland or pastureland to land with different crop types under BC1 and HH3 2040 scenarios. In the box plot, the box limits indicate the 25th and 75th percentiles, the center line shows the median, the whiskers are 1.5 times the interquartile range, and the box width is proportional to the square root of the number of observations. A positive value indicates SOC loss while a negative value indicates SOC gain.

Acronyms: COR – corn; SOR – sorghum; BIO – biomass sorghum; SOY – soybeans; WHE – wheat; OAT – oats; BAR – barley; COT – cotton; SWI – switchgrass; MIS – miscanthus; WIL – willow; POP – poplar; and HAY– hay.



be returned to the soil, which adds soil organic matter. Yield is also one of the most important determinants affecting the differences between emissions in BC1 and HH3 scenarios (fig. 4.14*A* compared to fig. 4.14*C* and fig. 4.14*B* compared to fig. 4.14*D*). However, it should be noted that, besides land-transition types and yield, many factors contribute to SOC dynamics, including spatially specific climate and soil conditions, and agricultural management practices. This is partly the reason why the SOC EFs vary spatially even under the same land-transition type. For instance, residue return is a common practice in the United States; however, the estimated return rate, the proportion of biomass residue that is returned to soil, differs from county to county, depending on modeling constraints applied in POLYSYS to limit soil erosion and maintain soil quality in general (chapter 4 in *BT16* volume 1). Therefore, in some cases, especially for land that is allocated to conventional crop production, EFs can vary significantly because residue return amounts vary even though the crop yield is relatively stable across the nation (fig. 4.14). For example, the estimated return rate varies from 10%–100% for barley straw and 20%–100% for corn stover and wheat straw (100% means full return). Return rates in the *BT16* analysis are determined through specific POLYSYS modeling for *BT16* scenarios as described in volume 1. Additionally, for crops that are not widely grown for biomass (e.g., biomass sorghum or barley) or are not significantly affected by land transitions (e.g., hay), based on POLYSYS output, we estimated SOC EFs for only a limited number of counties. EFs for these crops could therefore exhibit a wider range than others (fig. 4.14).

With POLYSYS-estimated land transitions and model-derived estimates of SOC changes in the scenarios, GHG emissions stemming from SOC changes at the county level are calculated on both a per-dry-ton feedstock basis (fig 4.15*A–C*) and in total for each county (fig. 4.15*D–F*). The results indicate that for the BC1 scenario (fig. 4.15*A* and 4.15*B*), the Midwest and the southeastern coast have significant potential for SOC gains and, correspondingly negative GHG emissions per mass of dry ton feedstock. The highest GHG emissions sink for BC1, occur in the Midwest for 2017 (fig. 4.15*D*) and eastern Kansas, northern Missouri, and southern Illinois for 2040 (fig 4.15*E*). Significant SOC losses that translate into high GHG emissions are more dominant in the BC1 scenario with its lower yields and occur mostly in the South. Notable hotspots that could experience significant SOC losses from feedstock production are several counties in North Dakota, Montana, and Colorado for the BC1 2017 scenario; these hotspots are focused in Oklahoma, eastern Texas, and western Arkansas for the BC1 2040 scenario (fig. 4.15*C*). As biomass yields are highest in the 2040 HH3 scenario, SOC losses are less severe and SOC gains are more significant in this scenario than in the BC1 2040 scenario. Texas, the Midwest, and the East Coast have the highest potential to act as GHG sinks on a per-dry-ton feedstock basis. Counties with the greatest SOC gains overall, and therefore the highest negative GHG emissions, are in the Midwest and South, most notably in central Texas. (fig. 4.15*F*).

The SOC-related GHG emissions are directly driven by the area of land in a county that changes in allocation from one use to another (based on POLYSYS output) and the corresponding SOC change for that allocation shift (derived from SCSOC). This analysis suggests that the areas with the greatest potential for SOC gains in 2040 are significant miscanthus producers, including counties in Illinois, Indiana, Iowa, Kansas, and Missouri (fig. 4.15*E* and 4.15*F*). Biomass sorghum has great SOC-sequestration potential, but its planting area is limited, and its contribution to SOC increases in the national landscape as conceived in this study is not significant compared with miscanthus (fig. 4.14).

A primary reason for SOC-related GHG-emission hotspots in the scenarios is the transition of pastures to crops that deplete soil carbon. Under BC1 scenarios, the use of permanent pasture to produce energy crops, especially switchgrass and poplar, caused significant SOC-related GHG emissions in counties in Texas and Oklahoma (fig. 4.15*D* and 4.15*E*). Results for California, in particular, illustrated that several counties could exhibit high GHG intensities under this scenario (fig. 4.15*A–C*). In addition to transitions from pasture to poplar that cause SOC loss, crop-residue removal (e.g., corn stover or barley straw) contributed significantly to GHG emissions. For locations that do not grow dedicated energy crops, residue removal could be one of the biggest factors contributing to overall GHG emissions. For instance, removing straws of wheat, oat, and barley, which reduces SOC, is one of the reasons why many Montana counties show GHG emissions as a result of soil carbon changes in scenarios (fig. 4.15). These results suggest the importance of further developing strategies that can mitigate GHG emissions from declining soil carbon levels including manure application and cover crop adoption (Qin et al. 2015).

At the national level, the total SOC-related GHG emissions are negative for all three scenarios (BC1 2017, BC1 2040, and HH3 2040) (fig. 4.15*D–F*), which suggests that land shifts overall result in a net SOC sink. For BC1 2017 and BC1 2040, the size of the sink is 3.0×10^{12} g CO_2e and 3.9×10^{12} g CO_2e, respectively. HH3 2040, however, has a much larger sink with 89.8×10^{12} g CO_2e.

4.3.3 Spatial GHG Emissions Including Agricultural and Forestry Operations, Logistics and Preprocessing, and SOC Changes

Figure 4.16 and Figure 4.17 present the spatially explicit GHG intensities and total GHG emissions, respectively, associated with potential agricultural and forestry biomass production from the scenarios. Agricultural GHG emissions include estimates

Figure 4.15 | County-level SOC change-induced GHG intensity and total GHG emissions associated with potential biomass production from the agriculture sector under 2017 and 2040 BC1, and 2040 HH3 scenarios, compared to a 2015 reference.

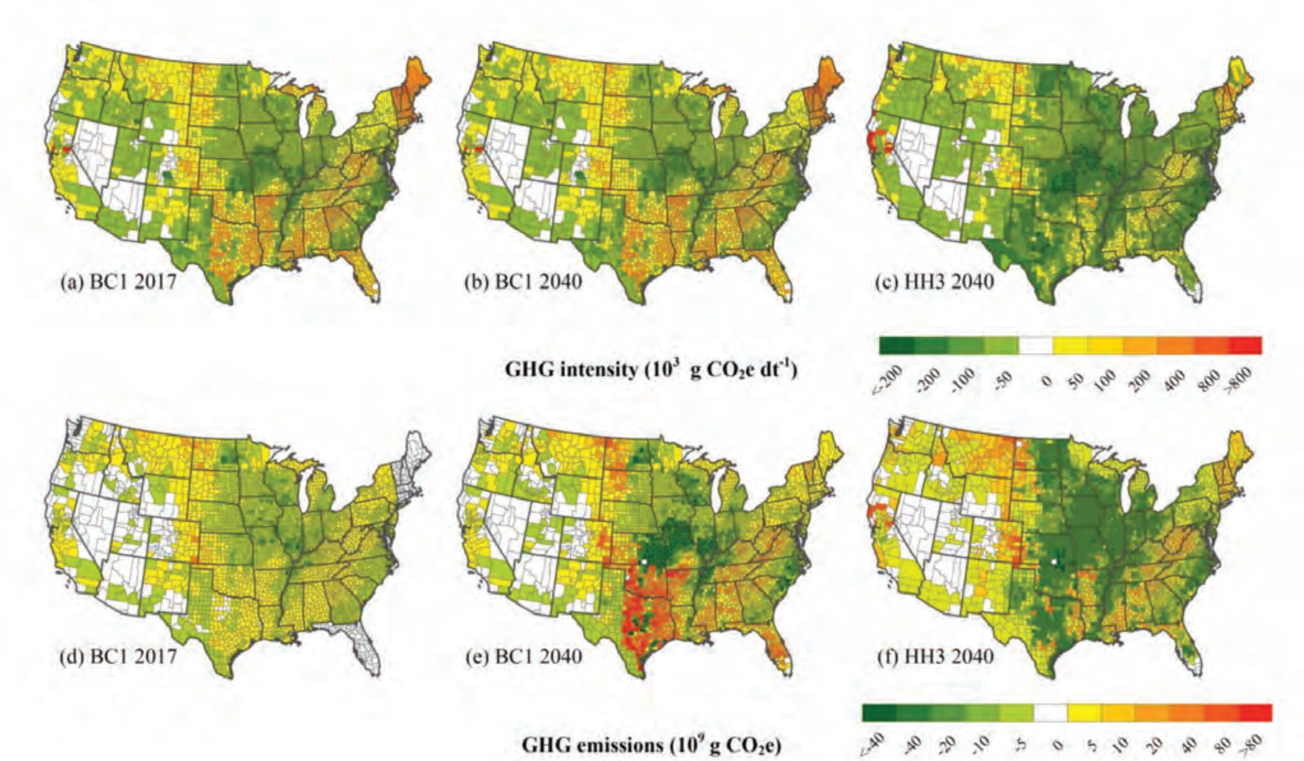
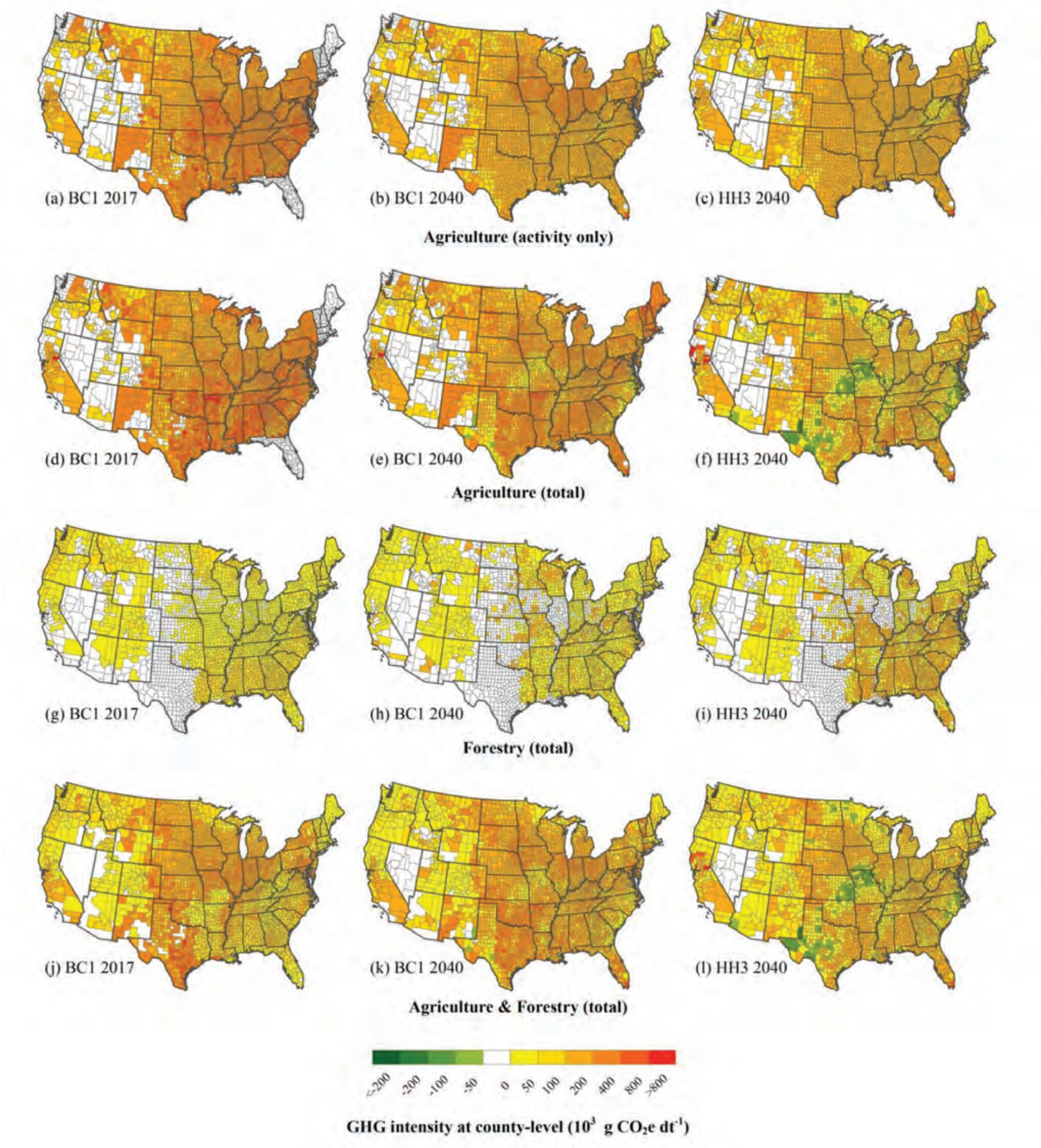


Figure 4.16 | Estimated county-level GHG intensity associated with biomass-feedstock production from agriculture and forestry sectors under 2017 and 2040 BC1, and 2040 HH3 scenarios. From top to bottom: agricultural activities without considering SOC change, agricultural biomass total GHG emissions with SOC change, forestry operations total (which does not consider SOC changes), and total emissions associated with producing all biomass, agricultural and forest-derived.



for farming operations, SOC changes, and logistics where biomass would be delivered to the biorefinery. Forestry GHG emissions, however, do not include SOC changes. Prohibition of land area changes between forestry and agriculture sectors in *BT16* scenarios translated into zero change in above ground carbon for forests that produce feedstocks. For GHG intensity in terms of a GHG-emissions-per-mass basis, agricultural feedstocks in the scenarios generally have relatively higher intensities than forestry-derived feedstocks when SOC changes are not considered (fig .4.16, first row compared to the third row). With SOC gains included for agricultural biomass, however, the GHG intensity in many counties could even be negative, which suggests net GHG sinks in these areas. For example, with agricultural activities and SOC changes considered under the HH3 2040 scenario, the modeled GHG intensity is well below zero, reaching to more than 100 kg CO_2e net GHG sink per dry ton biomass production in western Texas, eastern Kansas, and northern Missouri (fig. 4.16*F*). Even with additional GHG emissions from forestry-derived feedstocks, these areas could still result in a considerable GHG sink (fig. 4.16*L*). The reasons for potential SOC sequestration are explained in Section 4.3.2. Among three scenarios for agricultural feedstocks (fig. 4.16*A–F*), the HH3 2040 scenario has the lowest overall GHG intensity while BC1 2017 has the highest (fig. 4.16). This is mainly attributed to feedstock type and yield difference. Compared with BC1 2017, BC1 2040 assumes newly grown energy crops, which generally have lower GHG intensities than corn and soybeans—the crops predominantly used in BC1 2017. The HH3 2040 scenario, alternatively, has energy crops and highest crop yields (for both conventional and energy crops), resulting in lower GHG intensity.

To show total GHG emissions from biomass production under each yield scenario, fig. 4.17 combines analysis for GHG intensity and specific biomass production in each county. Without SOC changes included, the GHG emissions are primarily dominated by total biomass production in the county. For example, total GHG emissions are highest in the Midwest for agricultural feedstocks (fig. 4.17*A–C*), and in the Northwest and Northeast for forestry-derived feedstocks (fig. 4.17 *G–I*). When SOC changes are considered, noticeable changes are apparent in the western Texas, eastern Kansas, and northern Missouri areas of the HH3 2040 scenario where total GHG emissions are negative, suggesting that these areas could still act as GHG sinks after accounting for all GHG emissions from feedstock-production activities and logistics (fig 4.17*F*). Of course, there are other, less-noticeable changes, including decreased GHG emissions in some areas (e.g., Missouri in BC1 2040, fig. 4.17*E* compared to fig. 4.17*B*), or slightly increased GHG emissions in other areas or scenarios (e.g., eastern Texas, fig. 3.17*E* compared to fig. 4.17*B*). These changes are in line with the distribution of SOC changes (fig. 4.14). As feedstock production increases, either because of newly grown energy crops (i.e., BC1 2040 or HH3 2040) or higher yields (i.e., HH3 2040), the total GHG emissions tend to increase from BC1 2017 to BC1 2040 and then HH3 2040 (fig. 4.17*A–C*), except where SOC sequestration plays a significant role (e.g., fig 4.17*F*). For forestry-derived biomass, the GHG emissions trend is not as clear as for agricultural feedstocks because of the relatively smaller production of forest-derived biomass (fig. 4.17*G–I*).

To gain a sense of GHG-emissions drivers and spatial variations, contributors to total GHG emissions in the HH3&HH 2040 scenario are displayed for two counties, Vernon County, Missouri, and Gonzales County, Texas. The biomass produced in each county and corresponding GHG emissions are depicted in figures 4.18 and 4.19, respectively.

Figure 4.17 | Estimated county-level total GHG emissions associated with biomass production from the agriculture and forestry sectors under 2017 and 2040 BC1&ML, and 2040 HH3&HH scenarios. From top to bottom: agricultural activities (including transportation and logistics) without considering SOC change, agricultural total with SOC change, forestry total (which does not consider SOC changes), and total of both agriculture and forestry.

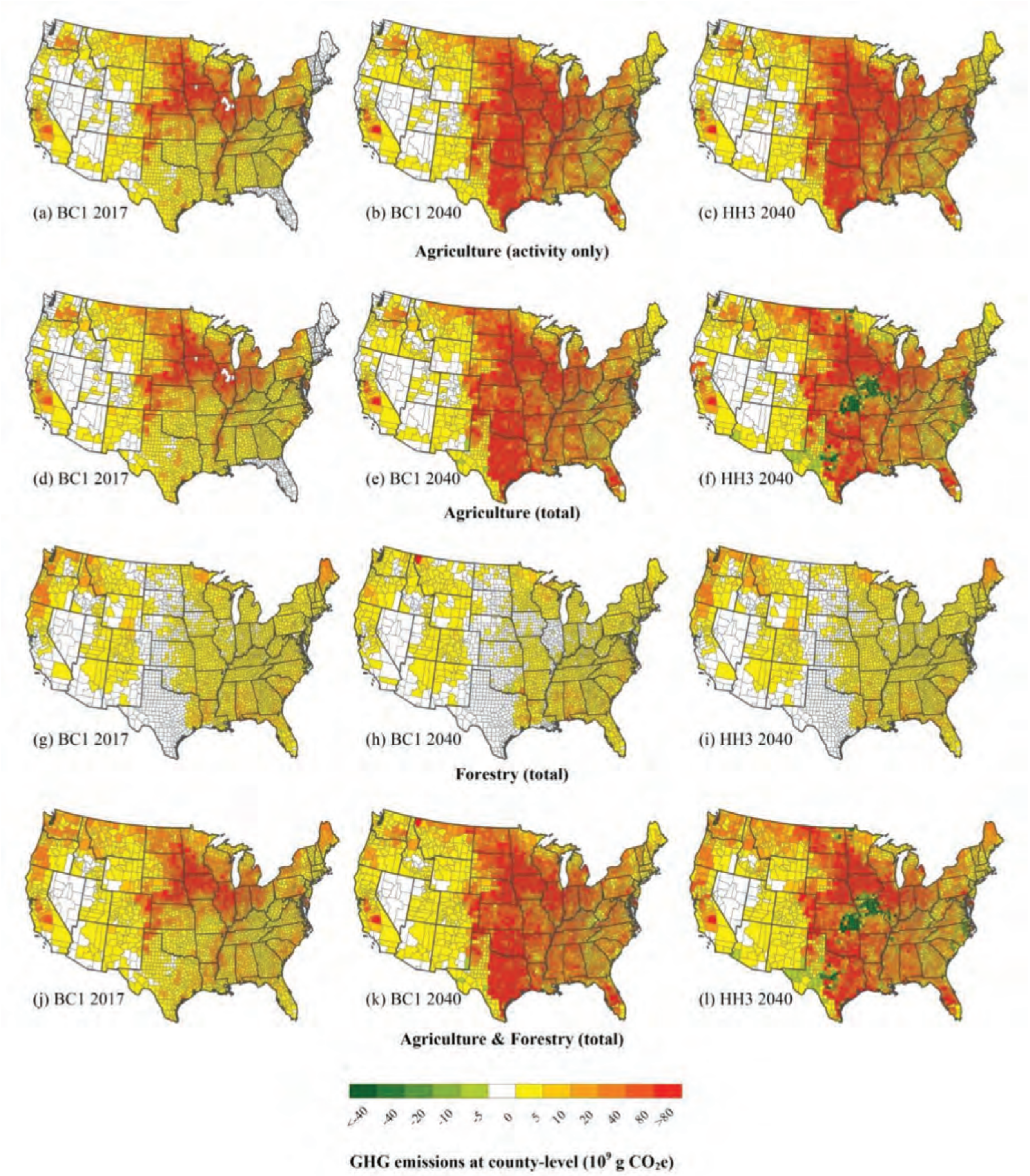


Figure 4.18 | Total potential biomass production by crop type in Vernon County, Missouri, and Gonzales County, Texas, in the HH3&HH 2040 scenario.

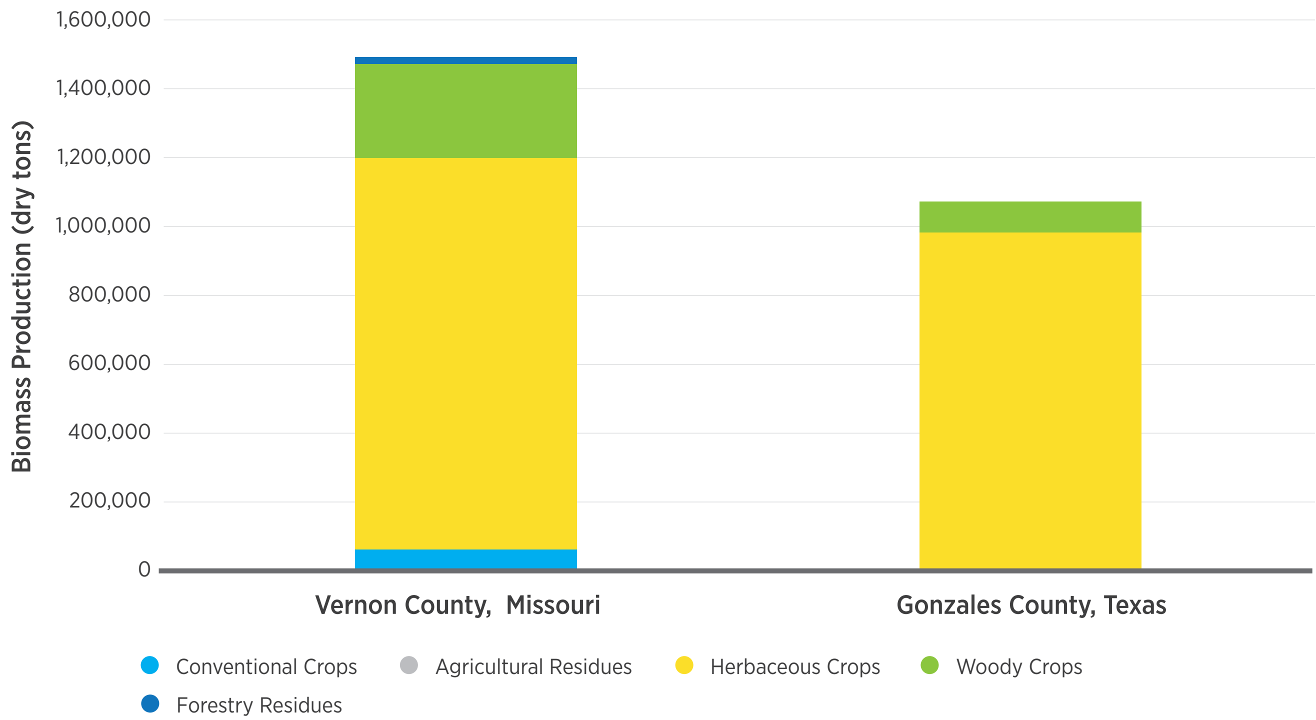
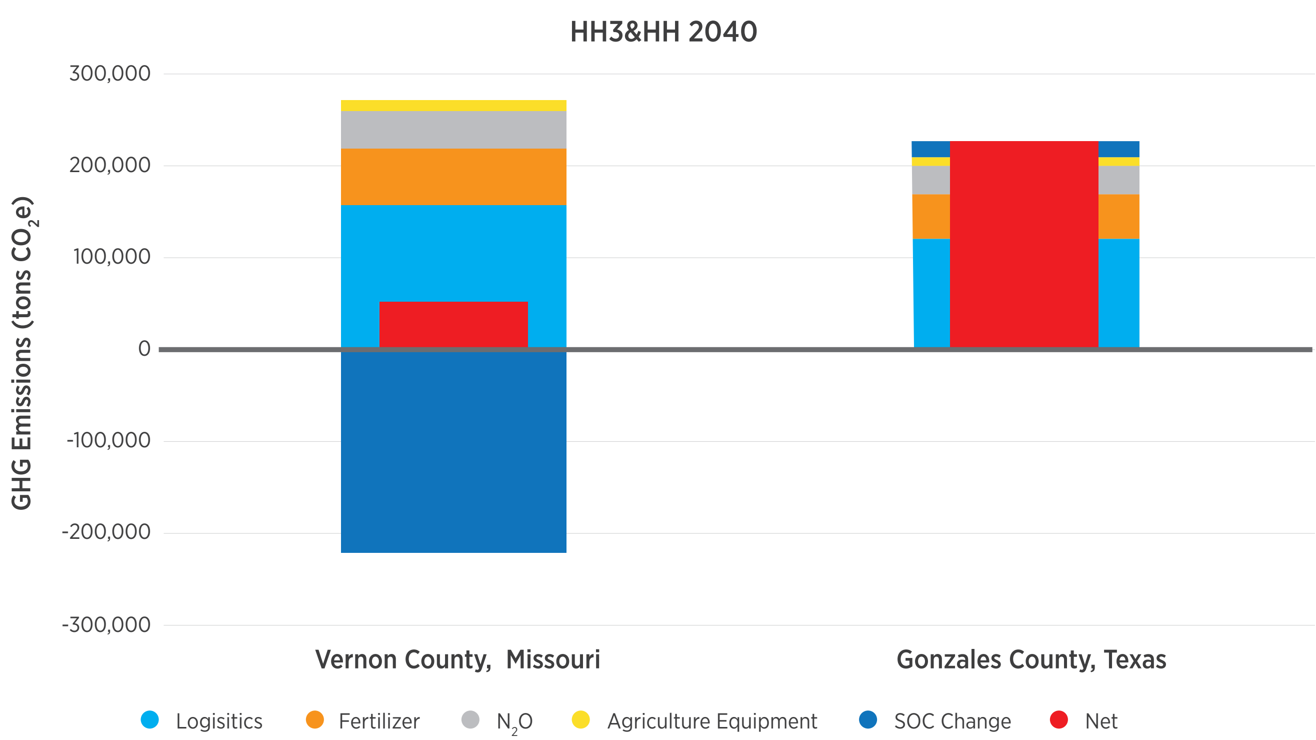


Figure 4.19 | Breakdown of estimated GHG emissions in Vernon County, Missouri, and Gonzales County, Texas. Each category represents the sum of the emissions of all of the feedstocks produced in the county in the HH3&HH 2040 scenario.



Both counties produce mostly herbaceous and woody crops in the 2040 scenarios. Vernon County would also produce conventional crops, as well as more biomass overall. In both counties, logistics contribute more than 50% to GHG emissions (excluding soil-carbon change-related emissions). The advanced logistics operations employed in the 2040 scenarios are energy-intensive. The second-largest contributor to modeled GHG emissions, aside from soil carbon-related emissions, is consumption of fertilizer and agricultural chemicals followed by nitrous oxide emissions stemming from fertilizer use. The operation of agricultural equipment is a minimal contributor to GHG emissions in these counties. Setting aside soil carbon changes, to reduce GHG emissions associated with biomass production, the energy efficiency of logistics operations and fertilizer efficiency should be improved.

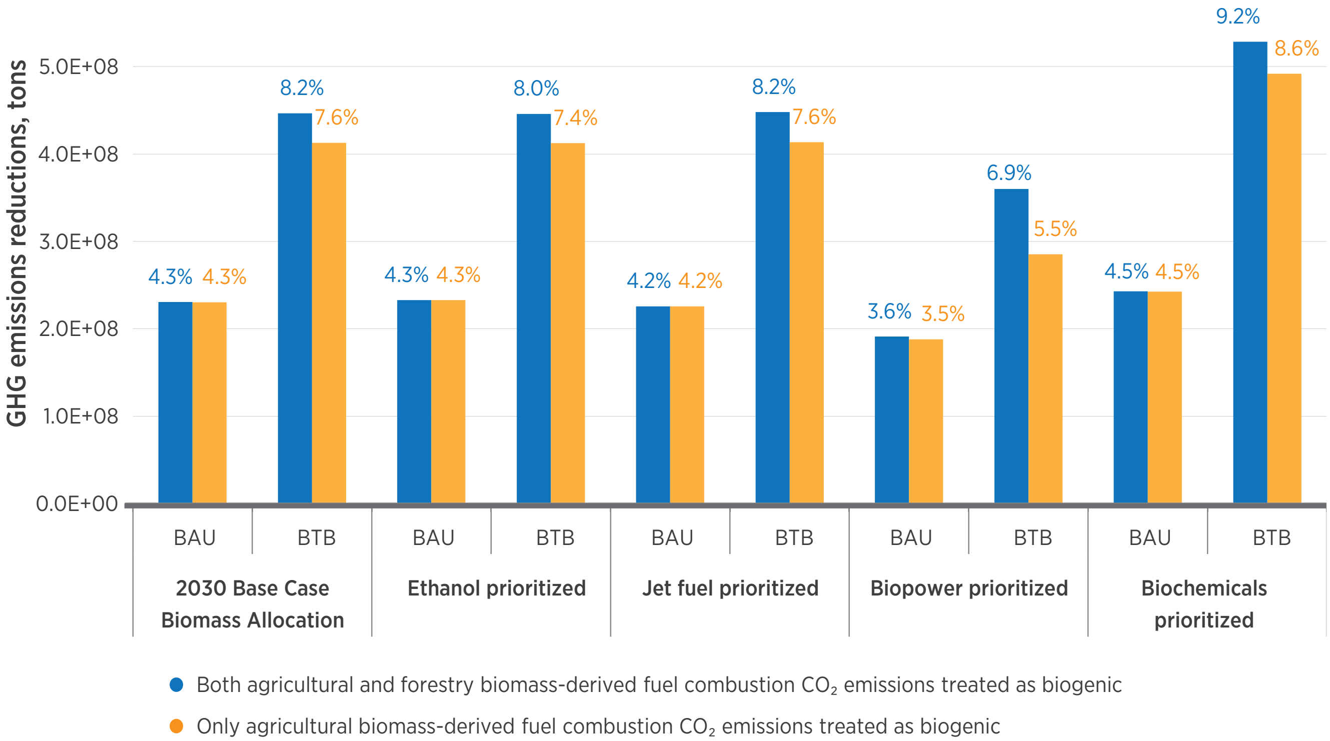
County-level SOC changes reported in this chapter are subject to the limitations described earlier. Nonetheless, figure 4.19 shows that potential production of over 1 million tons of herbaceous crops in Vernon County, 90% of which is miscanthus, significantly contributes to SOC gains in that location. In Gonzales County, SOC would decline despite significant production of herbaceous crops, 99% of which is switchgrass, which has lower yield than miscanthus and is less of a contributor to soil carbon sequestration. Miscanthus yield in Vernon County is 15.3 dry ton per acre whereas switchgrass yield in Gonzales County is 6.8 dry ton per acre. Growing high-yielding crops as energy crops can drive downGHG emissions associated with producing biomass.

4.3.4 Reduction in GHG Emissions for Representative Bioeconomy Cases

Examining GHG emissions associated with potential biomass that is produced and delivered to the reactor throat does not address the systems-level question of whether using bio-derived rather than conventional fossil feedstocks for fuel, power, and chemicals offers a GHG benefit on a life-cycle basis. There are many potential end uses for biomass—this chapter adopts end-use cases developed by a team of researchers as part of a BTB analysis to examine potential economy-wide GHG reductions from increased use of biomass, either at BAU biomass availability or at biomass availability levels as estimated in *BT16* (HH3 2040 scenario). Cases include a base case as well as cases that emphasize the production of ethanol, power, jet fuel, and bioproducts. These cases were developed with input from the U.S. Department of Energy, USDA, and other bioeconomy stakeholders and are documented in a journal article (Rogers et al. 2016). The methodology for this analysis is described in section 4.2.4.

Figure 4.20 summarizes the estimated GHG-emission reductions in various bioeconomy cases in 2030 as defined in Rogers et al. (2016), with the biomass availability in a BAU case and in the *BTB* case, in comparison to the estimated GHG emissions in the respective "all fossil" scenarios. The "all fossil" cases derive all fuel, power, and chemicals from fossil sources. The figure contains five cases that reflect different prioritizations of biomass use. In the 2030 base case, no one particular application of biomass is prioritized, but the remaining cases prioritize biomass use for ethanol, jet fuel, biopower, and biochemicals. In this analysis, two treatments of fuels produced from forest-derived biomass are considered. In the first, combustion CO_2 emissions of energy products produced from these feedstocks are treated as offset by biogenic carbon in the fuel. This is a conventional treatment for combustion emissions from annual and perennial feedstock-derived energy products, but it is under examination for fuels produced from forest-derived biomass (Daystar et al. 2016). In the second treatment, emissions from forestry biomass-derived fuels are treated as fossil carbon emissions that are not offset by biogenic carbon in the fuel. This result is a bookend case. When biogenic carbon dioxide emissions from forest-derived bioenergy are assumed

Figure 4.20 | Future potential (2030) avoided annual GHG emissions in various bioeconomy cases developed in Rogers et al. (2016).



to be carbon neutral, the GHG-emission reductions in the BTB cases can range from 6.9% in the heat and power end-use case to 9.2% in the bioproduct end-use case. When these emissions are not treated as carbon neutral, the GHG-emission reductions are lower and range from 5.5% in the heat and power end-use case to 8.6% in the bioproduct end-use case.

4.4 Discussion

It is important to note that the results reported in this chapter are a function of the *BT16* framework established in volume 1 (DOE 2016). Key parameters established in that volume influencing these results include crop residue-removal rates, land-allocation changes, limitations on land conversion that hold forested area constant, budgets for fertilizer, and agricultural- and forestry-equipment use. The magnitude of influence of each of these parameters on the results is dependent on the feedstock.

4.4.1 Implications of Results

In this chapter, GHG emissions and fossil energy-consumption estimates associated with scenarios BC1&ML 2017, BC1&ML 2040, and HH3&HH 2040 are as reported in Table 4.3.

Drivers of the national-level estimated GHG emissions vary by county. One common driver is logistics operations, especially under the long-term advanced logistics scenario. Efforts to improve the energy efficiency of logistics would improve GHG and energy impacts of biomass. Another driver of estimated GHG emissions is the yield for each feedstock. In general, counties with higher yields experience lower GHG emissions intensities, especially those where most or all of the agricultural inputs (energy and fertilizer) are applied on a per-acre basis regardless of yield (e.g., corn, soybeans). For example, conventional tilled corn produced in the BC1 2040 scenario has yields ranging from 334 bushel per acre down

Table 4.3 | Estimates of total biomass produced, GHG emissions, and fossil energy consumption for evaluated scenarios

Scenario	Total biomass produced (million dry tons per year)*	GHG emission (million tons CO_2e per year)	Fossil energy consumption (million Btu)
BC1&ML 2017	330	54	$4.0x10^8$
BC1&ML 2040	810	150	$1.3x10^9$
HH3&HH 2040	1,100	200	$2.3x10^9$

* Total includes biomass that would have total delivered costs exceeding $100/dry ton

to 38 bushel per acre. The GHG emissions for these scenarios to the farmgate (excluding transportation and preprocessing emissions) are 1,023,000 and 94,000 g-CO_2e per dry ton of biomass produced for the lowest and highest yields, respectively. On the other hand, the fertilizer, chemical, and diesel inputs of some other feedstocks (e.g., perennial crops) have less of an overall influence on results. For miscanthus, the highest and lowest yields for the BC1 2040 scenario were 12.2 and 1.9 dry tons per acre, respectively. This corresponded to GHG emissions to the farmgate of 50,900 and 77,900 g-CO_2e per dry ton of biomass for the highest and lowest yields, respectively. GHG emissions are also dependent on the FRR in which a county is located. Budgets that dictate energy and fertilizer inputs vary by FRR, but not greatly. For corn, fertilizer amounts are different in each FRR, as well as the amount of herbicides and insecticides applied. On the other hand, the FRR budgets for miscanthus vary only in the amount of potassium and lime applied per acre. Other fertilizer and chemical application rates on a per-dry-ton basis are the same in all counties for miscanthus.

An additional GHG emissions driver is soil carbon changes. In general, planting of deep-rooted species like miscanthus and biomass sorghum could contribute to soil carbon storage. The SOC implications of other energy crops like switchgrass and SRWCs vary depending on local factors like yield, soil type, and weather. Soil carbon change estimation in this analysis faced several limitations as discussed in section 4.2.2.2.

Even though biomass production results in GHG emissions, life-cycle analyses illustrate that net GHG reductions are possible when biomass feedstocks are used instead of fossil feedstocks to produce fuel, power, and chemicals. The examples considered in section 4.3.4 illustrate that for the portfolio of end uses considered in various 2030 cases , GHG-emissions reductions (between 4%–9%) and fossil energy reductions could be expected from broader use of biomass-derived energy and products that displace conventional energy and products produced from fossil fuels.

One important point regarding the results in this chapter is that they are estimates and aim to indicate potential GHG-emissions hotspots from producing biomass and to illuminate GHG drivers so that efforts can be made to mitigate them.

4.4.2 Uncertainties and Limitations

In addition to the limitations in SOC modeling discussed earlier in this chapter, some limitations of this study include not considering temporal aspects associated with forestry-derived feedstocks or soil-carbon changes associated with producing this biomass. Additionally, the development of estimates of SOC changes was limited by the absence of land-use history prior to land-conversion information from POLYSYS. It was also limited by a lack of information regarding which land types were used directly for crop production, necessitating the development of the land-use matrices described earlier. Additional limitations include assuming conventional tillage for conventional crops and accounting only for corn-corn rotations. Finally, increased validation of SOC modeling results for energy crops, for which few data are available, will help improve estimates of SOC changes in future analyses.

4.5 Summary and Future Research

In this analysis, we estimated the GHG emissions and fossil energy consumption that would be associated with scenarios BC1&ML 2017, BC1&ML 2040, and HH3&HH 2040 at the county level. The scenarios were selected to examine potential effects of national biomass expansion and yield changes on GHG emissions. For agricultural feedstocks, we incorporated SOC changes in the analysis. Furthermore, we considered illustrative scenarios in which the biomass resource estimated in 2040 was put toward a number of end uses and compared modeled GHG emissions and fossil energy consumption in the bioeconomy cases to a BAU scenario. We also considered and discussed carbon accounting considerations related to aboveground biomass and forest-derived feedstocks.

Overall, GHG emissions associated with the BC1 & ML 2017, BC1 & ML 2040, and HH3 and HH 2040 scenarios were 54, 150, and 200 million tons CO_2e, respectively. Key drivers of results were preprocessing in advanced logistics operations in place in 2040, which consumes a good deal of energy, and SOC changes, especially where deep-rooted feedstocks are estimated to grow.

Several aspects of future research are envisioned to build upon this analysis. One of these is to explore sensitivity of SOC changes to assumptions, including the treatment of tillage (the current analysis assumes all corn is produced with conventional till) and the effects of rotation. All corn is assumed to be in a corn-corn rotation—the influence of adopting corn-soy rotations and other rotations as informed by USDA data can be investigated in the future. Moreover, the influence of assumptions regarding crop yield, land-use history, and land-transition matrices on results can also be investigated. SOC-change hotspots and techniques to mitigate factors that cause them will also be a focus of this additional work, as will quantifying aboveground carbon changes to compute these and assess their relative contribution. A second aspect of future research is to introduce temporal-emissions accounting to our treatment of forest-derived feedstocks. Another area to explore in the context of emissions is advanced logistics, as they can be GHG-intensive. Ways to improve efficiency for biomass preprocessing can also be evaluated. The investigators in this and other chapters have noticed some modeling differences among chapters. For instance, evapotranspiration was estimated using the Blaney-Criddle method in SOC modeling (Kwon et al. 2013), while Penman-Monteith's approach was used in the water analysis (chapter 5). Even though both modeling approaches have been independently validated, it would be valuable to harmonize the methodology among analyses for different environmental indicators in future work. Finally, with all environmental effects (e.g., SOC, GHG emissions, water quality, water quantity, air quality, and biodiversity) of biomass production quantified spatially, it is necessary and feasible to identify hotspots considering these effects jointly and to provide information on potential preventive measures that can protect vulnerable regions.

4.6 References

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Appendix to Chapter 4 – Fossil Energy Consumption and Greenhouse Gas Emissions of Producing Agriculture and Forestry Feedstocks

Appendix 4-A: Detailed Methodology

In this appendix, we provide a detailed account of how Farm Resource Region (FRR) budgets, POLYSYS model outputs, and forest-biomass related data were used to generate the results in this chapter.

Calculating County-Level Greenhouse Gas Emissions and Fossil Energy Consumption

Potential greenhouse (gas) GHG emissions and fossil energy consumption for annual feedstocks, perennial crops, short rotation woody crops, and forest biomass, as described in the following subsections, are calculated based on the agricultural and forestry budgets that provide material and energy intensity of feedstock production.

Rather than use county-level POLYSYS outputs of fuel, fertilizer, and chemical consumptions, which stem from interpolation of raw crop budget data at the FRR level down to the Agricultural Statistical District (ASD) level (309 in the United States) and again down to the county level (3,000 in the United States), we used the FRR-level crop budgets themselves. In the current analysis, all assumed fertilizer, herbicide, and energy consumption amounts are taken from the FRR-level budgets. The amounts of energy, fertilizer, and chemicals to produce a given biomass type are the same for each county in one FRR. Intensities (e.g., the amount of energy consumed per dry ton of biomass) vary based on county-specific yield.

Fertilizer consumption in FRR budgets is reported as the amount of active ingredient (i.e., nitrogen, phosphorus, potassium, and lime). GREET contains emission factors for all of these active fertilizer ingredients. Emission factors for potassium and lime fertilizers were calculated directly, and those for nitrogen and phosphorus fertilizers were calculated by weighting the total fertilizer use by different fertilizer types in the United States and using their corresponding emission factors (Johnson et al. 2013). POLYSYS contains seven herbicides that are used for the cellulosic feedstocks including quinclorac; atrazine; 2,4-D amine; glyphosate; metolachlor; pendimethalin; and metribuzin. Of those herbicides, GREET only contains parameters for atrazine and metolachlor, in addition to two other herbicides, acetochlor and cyanazine. For this analysis, it is assumed that the other five herbicides have energy and GHG intensities that have the same average as that for the existing herbicides in GREET. Based on information in the crop budgets, herbicides account for a small percentage of the total mass of fertilizers and herbicides consumed in production of each feedstock (approximately 2%). For this reason, and based on previous analyses that showed a minimal contribution to biofuel life-cycle GHG emissions from insecticides and herbicides (Wang et al. 2012), it is expected that the contribution of these chemicals to biomass-production GHG emissions would be small. Therefore, we do not expect that using an average herbicide value from GREET (average of all four available in GREET) would significantly affect the energy and GHG estimates presented in this chapter.

Energy, including fuels and electricity, is directly consumed in farming equipment during feedstock production and harvesting, as well as in trucks during feedstock transportation. Diesel is the primary fuel type consumed. In adapting the FRR budgets for use in GREET, off-road diesel is assigned to be the fuel used in farming equipment such as tractors, while fuel consumed in on-road trucks that transport biomass is assumed to be on-road diesel. One key difference between off-road and on-road diesel is sulfur content, with on-road diesel containing 15 parts per million (ppm) sulfur, and off-road diesel containing 163 ppm sulfur. Electricity, natural gas, or propane may also be consumed during agricultural operations. GREET contains upstream production data for all of the energy types consumed in feedstock production. The development of these data are documented in several sources (Burnham et al. 2012; Cai et al. 2012, 2013; Elgowainy et al. 2014). One key note about electricity is that the grid composition varies by region. This analysis assigns a spatially explicit electricity grid based on the county where feedstock production is occurring. Each county was assigned a grid mix based on the North American Electric Reliability Corporation (NERC) region mixes (EIA 2015). Counties that are located in multiple regions are assigned the region in which most of the county's area lies.

Per–land-area consumption of energy, fertilizers, and chemicals is divided by yield per land area for the different crops at the county level to generate estimates of the material and energy intensity of producing each of the feedstock types. County-level yields that are used for this purpose are generated in two different ways (fig. 4.2). For conventional crops (e.g., corn or soybeans), yield data come directly from the U.S. Department of Agriculture's (USDA's) National Agricultural Statistics Service (NASS) (2015). Cellulosic crop yields (e.g., miscanthus or switchgrass) are estimated with PRISM. PRISM has been calibrated with yields from the Sun Grant Regional Partnership field trials. The model estimates yields for each cellulosic feedstock from climate and soil data. As a result, yields are spatially explicit at the county level. Dividing FRR-level material and energy data per area of land by county-level yield data produces per-feedstock mass intensity values that are also at the county level and are used as GREET inputs (e.g., grams of nitrogen/dry ton feedstock).

This analysis explores whether production of seeds and rhizomes should be included in the system boundary. POLYSYS output includes the cost of seeds and rhizomes but does not estimate the materials and energy consumed for their production. GREET default material and energy intensity data for feedstock production do not include the seed or rhizome production stage. It is expected that the energy and material intensity of seed production can be neglected in the *BT16* volume 2 analysis because previous analyses have shown that seed production contributes little to life-cycle GHG emissions of first-generation biofuel crops (e.g., corn and soy) (Shapouri et al. 2010; USDA 2013). For example, Landis, Miller, and Theis (2007) estimate that seed production contributes less than 0.002% to the energy for corn farming and transportation to the refinery. Planting stock production for willow is also a small contribution to the life-cycle GHG emissions of producing and transporting willow chips to a biorefinery (Caputo et al. 2014). Production data for other cellulosic crops are scarce, but it is assumed that seed production is also a minor contributor to life-cycle GHG emissions for cellulosic crop production. Currently, rhizome production for miscanthus is excluded from energy and GHG-intensity calculations. The contribution from these rhizomes may be small because they may have similar energy intensities as the planting stock for willow. However, they may be included in future analyses.

Annual Feedstocks (Corn and Soybeans)

Fuel, fertilizer, and agricultural chemical inputs for each feedstock produced in each county are multiplied by their respective, GREET-derived GHG emission factors and fossil energy consumption factors (table 4A.1). The GHG emission calculations also include the carbon dioxide emissions from lime (calcium carbonate) application. This value is estimated by multiplying the lime application rate by a 49.2% loss rate and converting subsequent carbon loss to carbon dioxide loss. Nitrous oxide emissions from nitrogen fertilizer nitrification-denitrification and biomass decomposition are included as well. For all feedstocks in BT16, it is assumed that 1.525 % of the nitrogen, on a kg-N basis, in applied fertilizer is emitted as nitrous oxide (Wang et al. 2012). For all feedstocks, it is assumed that 1.225% of the nitrogen in the biomass remaining on the field (assumed to be 10% by weight of the feedstock) is emitted as nitrous oxide during decomposition (ANL 2015). This percentage is multiplied by the nitrogen content of the relevant feedstock (table 4A.2). The nitrogen contents taken from the Bioenergy Feedstock Library (INL 2016) are for the type of feedstock that would be used in a final application (e.g., stem wood, not leaves, for willow). To convert GHG emissions per acre and per mass of feedstock produced, we used the amount of feedstock produced per planted acre because fertilizer, tilling, and other management practices would be carried out for all planted acres, not solely harvested acres. It is important to note that in calculating per-dry-ton GHG and energy intensity estimates, this analysis takes into account that some of the corn and soybeans in each county would be used for other industries (e.g., animal feed). The results are based on the portion of these feedstocks that are used as a bioenergy or bioproduct feedstock as determined by POLYSYS.

Table 4A.1 | Emission Factors Used to Calculate GHG Emissions (ANL 2015)

		GHG emissions ($g\text{-}CO_2e$/gal or lb)	Fossil energy consumption (Btu/gal or lb)
Fuel and Fertilizer	Off-road diesel	10,080	155,477
	On-road diesel	12,355	154,230
	Nitrogen	1,768	26,526
	P_2O_5	678	9,146
	K_2O	293	3,557
	$CaCO_3$	6.1	76.5
Herbicides	*Corn*	9,182	112,997
	Willow, poplar, eucalyptus, switchgrass, miscanthus	9,162	112,742
	Soybeans	9,487	116,746
Insecticides	*Corn, poplar, soybeans*	10,604	133,159

Table 4A.2 | Nitrogen Content of Annual, Perennial, and Wood Feedstocks

Feedstock	Nitrogen content of above- and below-ground biomass	Unit	Source
Biomass sorghum	9,343	g-N/dry ton	(INL 2016)
Corn	5,900	g-N/dry ton	(ANL 2015)
Energy cane	3,900	g-N/dry ton	(ANL 2015)
Eucalyptus	1,996	g-N/dry ton	(INL 2016)
Loblolly pine	3,991	g-N/dry ton	(INL 2016)
Miscanthus	3,175	g-N/dry ton	(INL 2016)
Poplar	3,629	g-N/dry ton	(INL 2016)
Soybeans	15,782	g-N/dry ton	(ANL 2015)
Switchgrass	5,715	g-N/dry ton	(INL 2016)
Willow	2,449	g-N/dry ton	(INL 2016)

Agricultural Residues (Corn Stover, Barley Straw, Sorghum Stubble, Wheat Straw, and Oat Straw)

Budget information for residues includes the energy used for residue collection and the amount of supplemental fertilizers applied as a result of residue collection to maintain soil nutrient levels. (GHG emissions associated with fertilizer applied to the crops themselves are not credited to the residue.) As with the annual crops, the fuel and fertilizer consumption were multiplied by their GHG emission factors from table 4A.1. The GHG-emission calculations also include nitrous oxide emissions from supplemental nitrogen fertilizer nitrification-denitrification and avoided nitrous oxide emissions from biomass that was removed from the field. (If this biomass had remained on the field, it would have emitted nitrous oxide as it decomposed.) The avoided nitrous oxide emissions are calculated using the aboveground nitrogen content of the biomass (table 4A.3).

Table 4A.3 | Aboveground Nitrogen Content of Harvest Residues

Feedstock	Nitrogen content of aboveground biomass (g-N/dry ton)	
Barley straw	6,350	(de Klein et al. 2006)
Corn stover	7,000	(ANL 2015)
Sorghum stubble	7,000	(de Klein et al. 2006)
Oat straw	6,350	(de Klein et al. 2006)
Wheat straw	7,000	(de Klein et al. 2006)

Perennial Feedstocks (Switchgrass, Miscanthus, and Energy Cane)

The emission calculations for the perennial feedstocks differ from those undertaken for the annual crops and residues because perennials undergo a multiple-year, multiple-harvest rotation. Each year in the rotation differs in terms of energy expended, fertilizers applied, and biomass yielded. Although POLYSYS output relates how much perennial crop is harvested from each county in each year, the output does not convey at what point in the rotation the perennial crop is harvested in a specific farm or plot. Fertilizer application for perennials tends to be concentrated in the initial rotation years, but feedstock harvested in later years has still benefitted from this fertilizer application. To spread the burden of fertilizer application and energy consumption across biomass produced over the entire rotation, these burdens are amortized over the rotation length at the county level (table 4A.4).

Table 4A.4 | Rotation Length and Yearly Maximum Yield Percentage

Feedstock	Rotation length (years)	Percentage of maximum yield by year
Switchgrass	10	Year 1: 50%; Year 2: 75%; Years 3–10: 100%
Miscanthus	15	Year 1: 0%; Year 2: 50%; Years 3–15: 100%
Energy cane	7	Year 1: 75%; Years 2–7: 100%

Equation 4A.1 relates the technique used to calculate an annual average fuel consumption and nutrient use when these values come from POLYSYS on a per acre basis.

Equation 4A.1:

$$\text{Fuel or Nutrient Use}\left(\frac{\text{gal or lb}}{\text{ac}}\right)=\frac{\sum_1^{10}\text{Yearly use of all operations}}{\text{Rotation Length}}$$

On the other hand, some fertilizer application rates after the establishment year are reported on a per-dry-ton basis because this rate is dependent on the amount of biomass removed from the field. In this case, a rotational average fertilizer consumption is developed because the amount of biomass produced varies by year. The application rate and the maximum yield (maximum output values based on the PRISM runs for each location, see section 4.2.4 of volume 1), summarized in equation 4A.2 are used for this calculation. This equation sums the product of the nutrient application for each year and the amount of biomass produced. This value is divided by the total biomass produced.

Equation 4A.2:

$$\text{Rotational Average Nutrient Use}\left(\frac{\text{lb}}{\text{dt}}\right)=\frac{\sum_{i=1}^{10}(\text{Nutrient Application})_i*(\text{Percent of Max Yield})_i*(\text{Max Yield})_i}{\sum_{i=1}^{10}(\text{Percent of Max Yield})_i*(\text{Max Yield})_i}$$

The final rotational average nutrient use is determined in Equation 4A.3.

Equation 4A.3:

$$\text{Rotational Nutrient Use}\left(\frac{\text{lb}}{\text{dt}}\right)=\frac{\sum_{i=1}^{10}(\text{Nutrient Application})_i*(\text{Percent of Max Yield})_i}{\sum_{i=1}^{10}(\text{Percent of Max Yield})_i}$$

The emissions due to nitrous oxide loss from fertilizer and biomass decomposition along with carbon dioxide emissions from applied lime are also considered for perennials as with the other agricultural crop types. The chemical- and fuel-use values are multiplied by their respective emission factors (table 4A.1) to arrive at county-level GHG emissions on a per-acre and per-dry-ton basis. These values are multiplied by the planted acres and total dry tons produced to arrive at the total GHG emissions.

Woody Feedstocks (Willow, Eucalyptus, Poplar, and Loblolly Pine)

The analysis for woody feedstocks is similar to that for perennials, but biomass is not harvested every year. The emissions are calculated in a similar way by using equations A4.2 and 4A.3 and considering nitrous oxide emissions from decomposing biomass and fertilizer undergoing nitrification-denitrification. However, compared to perennial scenarios, some of the fuel used for harvesting is reported on a per-dry-ton basis. As a result, the per-acre values are amortized over the rotation of the woody biomass. The per-dry-ton fuel consumption is calculated with equation 4A.3, with the percentage of maximum yield replaced by the percentage of biomass harvested. Loblolly pine and poplar are harvested only once, but eucalyptus and willow rotations undergo multiple harvests. For this analysis, it is assumed that 100% of the biomass for these feedstocks is removed during each harvest. GHG emissions of nitrous oxide emissions from fertilizer nitrification-denitrification and carbon dioxide emissions from lime decomposition were included. The GHG emissions were multiplied by the planted acres and dry tons produced to arrive at the total GHG emissions in each county.

Forestry-Derived Feedstocks

Forestry budgets derived from the CORRIM database and literature (see chapter 3 of volume 1) were used to generate the fossil energy and GHG intensity of forest-derived biomass including lowland and upland hardwoods, mixed woods, and natural and planted softwoods, in addition to their residues. The calculations for the forestry sector are similar to those for the agricultural sector. One important consideration is the technique used to assign burdens to residues—the forestry analysis approach (chapter 3 of volume 1) assigns 10% of energy and fertilizer resources to residues as opposed to the rest of the harvested trees. For consistency, this chapter uses the same level of fuel and nutrient intensity for the estimation of fossil energy consumption and GHG emissions associated with forest residues as chapter 3 of volume 1. In the future, other methods of allocation may be explored, including mass allocation.

Feedstock Logistics

Potential logistics scenarios to deliver biomass to biorefineries were developed in the *BT16* volume 1 analysis and are presented in chapter 6 of that volume, for a selected group of feedstocks. The analysis estimates the transportation distances required for each type of biomass produced in each county for both a near-term scenario using conventional logistics systems to deliver bales or wood chips to the biorefinery and in 2040 using advanced logistics (pelletization at regional depots). For conventional logistics, the biomass is transported to the biorefinery as is, while in advanced logistics the biomass is first taken to a depot, processed into blended feedstock pellets, and then transported to the biorefinery. The feedstocks that are considered in the logistics analysis of *BT16* volume 1 include corn stover; miscanthus; switchgrass; biomass sorghum; woody feedstocks including eucalyptus, pine, willow, and poplar; and forestry-derived whole trees and residues. Transportation parameters (e.g., payload, fuel economy with payload, and fuel economy without payload [for backhaul trips]) for these feedstocks are provided in table 4A.5. Transportation and logistics inputs for other biomass types not analyzed in *BT16* volume 1, including corn, soybeans, energy cane, and other non-corn stover agricultural residues, were estimated separately. For example, transportation fuel consumption for corn and soybeans is taken from GREET, which uses trucks. The transportation distances for both of these feedstocks in GREET is 10 miles one-way to the collection stack and 40 miles one-way to the biorefinery. The GHG emissions for transporting both of these feedstocks is 18,000 g-CO_2e/dry ton. The analogous transportation information was not available in GREET® for agricultural residues and energy cane. Therefore, we assumed all crop residues not subject to logistics analysis in volume 1 have the same transportation-related energy intensity as does corn stover in GREET. The related parameters include a transportation distance of 53 miles, a fuel economy of 5.7 miles/gallon, and a load capacity of 17 dry tons/load. For these residues, the resolution of logistics energy use and GHG emissions is available at a national level, rather than at the county level. An important assumption for this portion of the analysis is that we assigned all burdens to the county of feedstock origin rather the county where the biorefinery may reside or any intervening county.

Table 4A.5 | Logistics Information for Transportation of Feedstocks (taken from BT16, volume 1, chapter 6)

	Payload (dry ton/ load)	Fuel economy with load (miles/ gal-diesel)	Fuel economy without load (miles/gal-diesel)	Data source
Corn stover	17	5.6	7.7	(Webb, Sokhansanj, and Turhollow 2013a)
Switchgrass/ miscanthus	17	5.6	7.7	(Webb, Sokhansanj, and Turhollow 2013b)
Sorghum cane	21	5.5	7.7	(An and Searcy 2012)
All woody feedstocks	17	4.8	7.7	(INL 2014)
Pellets	21	5.5	7.7	(Webb, Sokhansanj, and Turhollow 2013b)

At the biorefinery itself, the feedstock undergoes some basic processing steps for delivery to the reactor throat. The Idaho National Laboratory Design Case (INL 2013) provided an estimate of this energy intensity, which is primarily due to consumption of electricity on a per dry ton feedstock basis for the same feedstocks considered in *BT16* volume 1 (chapter 6 in that volume). The diesel consumption and preprocessing electricity for these feedstocks are summarized in table 4A.6. Electricity values are similar for all conventional feedstocks and all advanced feedstocks. However, the diesel consumption for corn stover is lower than forestry feedstocks because preprocessing of stover only consumes diesel for vehicle loading, while preprocessing of forestry feedstocks consumes diesel for chipping, which is a more energy-intensive process than vehicle loading. For corn, the preprocessing information is used for a corn dry-grind biorefinery at 8 kilowatt-hours per dry ton of corn (Kwiatkowski et al. 2006). The same type of information was not found for soybeans, but given that they are also ground before the reactor throat, the same electricity consumption on a per-ton-biomass basis is used. *BT16* volume 1 analyses did not assess preprocessing energy consumption for agricultural residues described earlier as also lacking transportation and logistics analysis in volume 1. To estimate preprocessing energy consumption for these residues, we assumed that it is as energy-intensive to preprocess them as it is corn stover (table 4A.6). *BT16* volume 1 (chapter 6) assumes that any biomass with a delivered cost greater than $100/dry ton is not considered feasible and would in essence be left on the field. Therefore, we do not consider GHG and energy consumption emissions associated with transportation and preprocessing of this biomass, but do account for the GHG emissions associated with its production (e.g., fuel, fertilizer, chemical consumption on the farm).

Table 4A.6 | Preprocessing Energy Consumption for Feedstocks (taken from *BT16*, volume 1, chapter 6)

	Diesel (Btu/dry ton)	Source (Btu/dry ton)
Conventional logistics		
- Corn stover	26,300	123,000
- Forest whole tree and residues	154,000	136,000
Advanced logistics		
- Corn stover		655,000
- Biomass sorghum		592,000
- Miscanthus		643,000
- Switchgrass		643,000
- Forest whole tree and residues		653,000

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Appendix 4-B: Sustainability of Extracting Primary Forest Residue Biomass

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4B.1 Introduction

Harvesting timber from forests creates ecological disturbances that affect myriad properties and processes. These disturbances have been studied for decades. The disturbance type and severity, coupled with the ecosystem properties, determine whether the ecosystem can be resistant (i.e., little change is evident in the ecosystem), resilient (i.e., initial change is followed by recovery to similar conditions), or irreversibly altered. While specific, long-term responses of all processes are not yet known, the effects of harvesting timber on a site's productivity are well understood. The harvesting of other materials in addition to those traditionally removed for wood products (e.g., smaller-diameter trees, branches, or leaves) as well as potentially higher trafficking, can increase the severity of the ecological disturbance; this increase in severity raises additional questions regarding ecosystem responses and the sustainability of site productivity (Janowiak and Webster 2010). Research on these impacts began in the 1970s and has increased recently due to a rise in general interest in woody biomass for energy.

4B.2 Research on Site Productivity Following Biomass Harvests

Harvesting biomass for energy from forests occurs in a wide variety of management types from short-rotation, purpose-grown woody crop systems to intensively managed plantations, to extensively managed forests and woodlands (Stone 1975). Within the most intensive woody-biomass feedstock systems, maintaining site productivity is imperative to efficient management. Nutrient deficiencies that may be present are mitigated as a matter of course through fertilization. The management of these systems in terms of technological inputs to manage water, nutrients, and non-crop vegetation is more intensive than traditional forestry, but usually less intensive than typical agricultural systems. Similarly, the ecological sustainabiilty of these systems must be considered relative to previous land use (Blanco-Canqui 2010; Holland et al. 2015). In comparison to annual systems, short-rotation woody crops offer several environmental advantages. For example, when sited on marginal agricultural land, these systems improve soil productivity and offer additional environmental benefits such as improved water quality and wildlife habitat.

Within conventionally managed forest ecosystems, there are concerns over biomass harvesting, thinning operations, and ecological impacts from the removal of additional wood following conventional stem-only harvests (Page-Dumroese, Jurgensen, and Terry 2010). Some dead woody biomass is left on-site as it serves several important ecological functions in forest ecosystems that are affected by harvesting (Harmon et al. 1986). This dead woody material serves as a habitat for a variety of organisms, including fungi, mosses, liverworts, insects, amphibians, reptiles, small mammals, birds, and regenerating plants. In cool climates, downed logs act as nurse logs for seed germination and stand establishment. Birds forage, nest, and hunt in and on dead wood. Dead woody material affects ponding, sediment trapping, and aeration in streams; it also impacts site productivity through several mechanisms.

This dead biomass alters a site's water balance and quality by storing and releasing water and by reducing runoff and erosion. Dead woody material supports biological nitrogen fixation, thereby increasing on-site levels of nitrogen, and it contains nutrients that are cycled back into the soil. It is also commonly used during harvest operations to protect wet soil areas from compaction and rutting and is used post-harvest to help limit runoff and erosion from skid trails and forest roads.

4B.3 Compaction

Biomass harvesting operations cause ground disturbance and some result in increased trafficking compared to traditional harvesting. These disturbances result in physical changes such as compaction, soil mixing, and altered surface hydrology; however, the extent, duration, degree, and distribution of the impacts are site-, soil-, and harvest method-specific (Cambi et al. 2015). In addition, woody debris is sometimes used to protect soils from disturbance or from erosion, and biomass harvesting could reduce this resource.

Under the Long-Term Soil Productivity experiment (LTSP) in North America, compaction has had mixed effects on tree growth over a period of 10–15 years. In most cases, compaction has had little to no significant impact on early survival or productivity (Ponder Jr. et al. 2012). Sites with clayey soil textures have reported declines in young tree growth due to compaction (Gomez et al. 2002), while productivity increased on loamy and coarse-textured soils after compaction due to improvements in water-holding capacity or other physical attributes. Compaction effects occurring across a range of textures in southern pine sites resulted in increased tree productivity due to a reduction in competing vegetation (Scott et al. 2014).

The loss of nutrient capital and organic matter due to biomass harvesting is of particular concern for sustaining site productivity and carbon sequestration potential. While biomass harvesting includes more sources than just residue from conventional harvest systems, the majority of research in the United States on nutrient removals from biomass harvesting focuses on the impact of whole-tree harvesting relative to conventional harvesting and the removal of small-diameter trees for silvicultural and fire-protection purposes. Whole-tree harvest is usually defined as all woody biomass contained in standing trees aboveground, where complete-tree harvest removes the stump and large root biomass, as well. More-intensive biomass harvesting removes existing dead wood from the site. Logging residues, or the remainder of the standing tree after the conventionally merchantable bole is removed, contain a disproportionately high nutrient content relative to the bole. For example, whole-tree harvesting removed 47% more biomass (165 Mg ha^{-1} versus 112 Mg ha^{-1}) on average than stem-only harvesting from 6 hardwood and 5 conifer stands, but 86% more nitrogen (321 versus 172 kg ha^{-1}), 105% more phosphorus (37 versus 18 kg ha^{-1}), and 112% more calcium (216 to 459 kg ha^{-1}), respectively (Mann et al. 1988). Small-diameter trees removed in thinning operations or in dedicated short-rotation woody crop systems also have a comparatively high nutrient capital due to a larger proportion of high nutrient-concentration biomass (e.g., leaves, needles, branches, or bark). Thus, the nutrient removal is much greater in biomass-harvesting systems than in conventional harvesting systems relative to the actual amount of biomass harvested. Therefore, it is important to manage the retention of portions of the biomass to ensure long-term productivity by leaving residues or by time of harvest.

Two recent reviews (Thiffault et al. 2011; Achat et al. 2015) analyzed existing studies regarding the soil and tree growth impacts of whole-tree harvesting compared to stem-only harvesting. Based on these empirical data sets, it is clear that removing the more nutrient-rich materials (e.g., branches or foliage) can cause reductions in soil fertility and affect tree growth by altering microclimate, fertility, and other vegetation; however, these impacts are minor and inconsistent. For example, a global meta-analysis (Achat et al. 2015) found that subsequent tree growth (e.g., volume, basal area, or biomass) was reduced by a median 3.1% (-15.1% in Quartile 1 to 2.8 % in Quartile 3) across 48 studies when branches and foliage were harvested in addition to boles. Experimental treatments often have a greater impact on tree growth by affecting competing vegetation (Thiffault et al. 2011), which is not an indicator of long-term site productivity (Burger 1994).

Within the United States, the LTSP experiment was initiated specifically to answer questions about the impact of varying degrees of organic-matter removal on soil and site productivity. The most recent network-wide review of the first 10 years following treatment found no consistent impact of intensive organic-matter removal on tree growth (Ponder Jr. et al. 2012). By age 15—the time when nutrient deficiencies tend to be most prevalent—most of the U.S. sites in the LTSP study had reached canopy closure. Recent regional and individual site reports concluded that the most intensive treatment, which removed all organic material including the forest floor (which was not intended to be an operational treatment), resulted in minor reductions in growth on some sites, but that whole-tree harvesting vary rarely reduced tree growth (Holub et al. 2013; Scott et al. 2014; Curzon et al. 2014). One exception of this occurred on sites inherently deficient in phosphorus in Louisiana, Mississippi, and Texas (Scott and Dean 2006; Scott 2016). Other trials in the United States have similarly shown little, if any, response in changes to a site's productivity or to most mineral soil properties (Johnson et al. 2002; Roxby and Howard 2013; Jang et al. 2015); when responses do occur they are highly site specific.

While empirical evidence indicates that biomass harvesting in the United States will not cause widespread or severe reductions in productivity due to decreases in fertility or soil porosity, few studies have examined long-term (rotation-age or longer) results or results from repeated biomass harvests. Thus, a cautionary approach has been suggested by most reviews (Janowiak and Webster 2010). In addition, there are some regional-, soil-, and forest-specific concerns. For example, some forests in the eastern United States are at a relatively high risk of calcium loss from harvest (Adams et al. 2000; Huntington 2000). The loss is due to low-calcium geologic parent materials, decades of acid precipitation that have leached much of the natural calcium capital from the soil, and, in the southeastern United States, the high degree of weathering. In southeastern pine forests, certain geologies are markedly low in phosphorus and routinely fertilized to overcome their natural deficiency and to avoid induced deficiency by harvest removals. Nitrogen is a limiting factor throughout the United States, with the exception of the Northeast. However, in dry or cold forests where nitrogen cycling is retarded due to climate, nitrogen losses in harvested materials may substantially reduce productivity by lowering decomposition and nitrogen-mineralization rates. Continued research is needed to identify specific forest and soil types where nutrient removals may exacerbate potential deficiencies or where soil disturbance from biomass harvesting will not be sustainable (Vance et al. 2014; Vadeboncoeur et al. 2014).

Based on the ecological and productivity-related roles of dead woody debris and the fact that some timberland owners may not want or be able to fertilize, in order to mitigate potential productivity loss from increased nutrient removals, some level of organic matter should be retained to protect these functions. Some of the material may be present in a stand prior to harvest, while some is created as logging residue or by density-induced natural mortality.

Because dead wood is important in many complex functions, and the amount needed to perform these functions varies widely across climatic, geologic, edaphic, and vegetation gradients, a single retention percentage should not be used as an actual guideline. Rather, retention guidelines should be developed at state-to-local geographic scales, by forest type, and by harvesting intensity. Several states and the two largest certification programs in the United States (Sustainable Forestry Initiative® and Forest Stewardship Council) have released guidelines that address the productivity and ecological functions of dead wood (Evans et al. 2013). Most of the guidelines are for general timberland conditions, with some additional restrictions for special areas, such as critical plant or animal habitat, shallow soils, or steep slopes.

For example, Maine requires all coarse woody material that exists prior to harvest to be retained after harvest and at least 20% of the logging residues with less than 3-inch diameters should be retained. Minnesota recommends that 20% of the logging residues be retained and scattered throughout the harvest tract. Wisconsin's guidelines require 5 tons per acre of woody material to be retained, but the material can be derived from either logging slash or woody material present prior to harvest. Pennsylvania's guidelines call for 15%–30% of the harvestable biomass to be retained, while Missouri requires 33% retention. Sensitive sites and soils are also protected. Minnesota suggests avoiding biomass harvesting in areas with threatened, endangered, or otherwise sensitive plant or animal habitats formed within riparian management zones, on certain organic soils, and on shallow soils with aspen or hardwood cover types. In general, the literature and harvest guidelines indicate that a 30% retention rate of logging residues on slopes less than a 30% grade and a 50% retention rate on steeper slopes are reasonable and conservative estimates of the amount of material needed to maintain productivity, biodiversity, and carbon sequestration and to prevent erosion and compaction.

For the United States, Janowiak and Webster (2010) offer a set of guiding principles for ensuring the sustainability of harvesting biomass for energy application. Others (Vance et al. 2014; Gollany et al. 2015) offer strategies for continued research. These principles include:

- Increasing the extent of forest cover, including the afforestation of agricultural, abandoned, and degraded lands, as well as the establishment of plantations and short-rotation woody crops
- Adapting forest management to site conditions by balancing the benefits of biomass collection against ecological services provided (e.g., old-growth forests provide ecological services and habitat benefits that greatly exceed bioenergy benefits); using best management practices
- Retaining a portion of organic matter for soil productivity and deadwood for biodiversity; considering forest fertilization and wood-ash recycling
- Using biomass collection as a tool for ecosystem restoration where appropriate.

When these principles are applied through state-based best management practices or biomass-harvesting guidelines or certification, biomass harvesting can be sustainably practiced with reduced negative impacts on the environment, and harvesting can be a much-needed tool for achieving forest health-restoration objectives.

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05 Water Quality Responses to Simulated Management Practices on Agricultural Lands Producing Biomass Feedstocks in Two Tributary Basins of the Mississippi River

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5.1 Introduction

Water quality is a legitimate concern for any proposed shift in the nation's energy portfolio. Of the total length of wadeable U.S. streams, 42% are in poor condition (Paulsen et al. 2008). Increasing human exposure to nitrates in drinking water is a significant health concern in the Midwest because of its increasing trend in groundwater of this region (Stets, Kelly, and Crawford 2015). Nitrogen enrichment has played a role in the imperilment of aquatic species (Hernandez et al. 2016). Decomposition of algal blooms during summer periodically depletes water of oxygen in a significant number of rivers, lakes, and reservoirs. Downstream nutrient excesses have degraded more than 60% of coastal rivers and bays in the United States (Simpson et al. 2008). Furthermore, climate warming is likely to exacerbate problems and increase the potential for harmful algal blooms and the incidence of hypoxic conditions in rivers, lakes, and estuaries.

Given the state of the nation's waters, it is important to understand the water quality implications of future biomass feedstock production systems. Will future production have positive or negative impacts on water quality? The answer likely depends on the choice of crop (feedstock) and how the energy crop is managed relative to the previous non-energy crop. At one end of the spectrum, expansion of corn acreage to support grain-based ethanol production might be expected to degrade water quality in the same way that corn grown for food and animal feed would. This is because corn is inefficient in nitrogen uptake (Simpson et al. 2008). Under this 'worst-case' scenario, increasing grain (corn) production might put the goal of reducing the hypoxic 'dead' zone in the Gulf of Mexico farther out of reach (Donner and Kucharik 2008). Assuming an 80% increase in corn acreage, the estimates of nutrient losses from the Mississippi-Atchafalaya River Basin (MARB) using the SPARROW model were 37% nitrogen and 25% phosphorus, respectively (Simpson et al. 2008). This highlights the potential benefits associated with alternative cellulosic and perennial biomass feedstocks, combined with targeted best management practices applied to areas planted in corn.

5.1.1 Cellulosic and Perennial Feedstocks

The outlook for water quality has changed with the prospect of growing and using cellulosic and perennial feedstocks. Compared with corn, cellulosic and, especially, perennial feedstocks, including short-rotation woody crops (SRWCs), have considerable benefits for improving water quality (Simpson et al. 2008) by potentially reducing nutrient loadings by half (Alshawaf, Douglas, and Ricciardi 2016, Evans et al. 2009). Research is showing that regional-scale production of feedstocks consistent with the Energy Independence and Security Act of 2007 and/or the Billion-Ton Update (DOE 2011) could improve water quality (Costello et al. 2009; Jager et al. 2015), particularly when perennial biomass feedstocks replace more intensively managed crops (Love and Nejadhashemi 2011).

5.1.2 Conservation Practices

In this chapter, the question posed is, "How can future biomass feedstocks be managed to protect water quality with minimal decrease in feedstock supply?" Thus, our emphasis is on identifying the 'swing potential' of different management practices (Davis et al. 2013). In other words, which practices have the highest potential for protecting water quality? We ask whether water quality can be protected by choosing perennial feedstocks and/or incorporating suitable combinations of best management practices into biomass-feedstock production. Practices evaluated in the past have included more precise application of fertilizer; use of cover crops, filter strips, and riparian buffers; no-till management; and mitigation of agricultural drainage. Although most studies focused on the watershed scale, water quality benefits of such practices have also been demonstrated at the scale of a large river basin, using models, for example, in the Upper Mississippi River Basin (UMRB) (Wu, Demissie, and Yan 2012; Demissie, Yan, and Wu 2012).

From a crop-management perspective, reduced or targeted fertilizer management can enhance the efficiency of nitrogen application and, thereby, provide farmers with flexible options for maintaining high-yielding production systems (Nelson, Motavalli, and Nathan 2014; Noellsch et al. 2009) and reducing nitrogen runoff. Using cover crops with corn and interplanting SRWCs have been shown to prevent excess nutrients from flowing into adjacent water bodies (Nyakatawa et al. 2006). In a comparison of management practices, nitrate leaching from Midwest fields growing annual crops (wheat, corn, and soy) was highest under conventional management, followed by no-till, reduced-input (20% to 50% fertilizer with leguminous cover crop), and organic production with no fertilizer inputs (Syswerda et al. 2012).

Planting perennial crops has been shown to reduce nitrate leaching more than the conservation practices applied to corn-based production systems (Syswerda et al. 2012). One of the most effective strategies—implementing a conservation buffer in riparian areas—can significantly decrease losses of nitrogen, phosphorus, and soil by trapping overland flow (Blanco et al. 2004; Dosskey et al. 2010; Balestrini et al. 2011). A review of widths of riparian buffers and filter strips by Fischer and Fischenich (2000) recommends a 5 meter (m) to 30 m width for water quality protection. Zhang et al. (2010) found that a 30-m buffer was required to remove 85% of nutrients on slopes up to 10%. Similarly, Gharabaghi, Rudra, and Goel (2006) found that more than 95% of sediment aggregates were removed by the initial 5 m of the vegetative filter's width.

The above practices might be rendered completely ineffective by artificial drainage (Petrolia and Gowda 2006; Petrolia, Gowda, and Mulla 2005). Excess nutrients (especially nitrate) bypass surface improvements, such as conservation tillage or riparian buffers, and flow through the soil into tile lines (Lemke et al. 2011). In addition, mitigation efforts that target drainage can be very effective—for example, controlled drainage (permitting water on fields during the fallow season). Filter strips can still be effective if they are located where they intercept shallow flow paths (Ssegane et al. 2015). Similarly, placement of filters at the inlet of tile-drain systems and placement of filter strips or wetlands at outlets can reduce nutrient losses. Addressing nutrient pathways through tile drains is critical to the success of nutrient-management efforts in the Midwest, where tile drains prevent waterlogging of crops and permit access by farm equipment.

5.1.3 Co-Optimizing Production and Water Quality

Is it possible to have the best of both worlds—high yields of biomass feedstocks and high water quality? Previous research at the watershed scale has found that balancing economic and environmental objectives using a spatially optimized landscape of biomass plantings can help move toward sustainable biomass-production systems (Parish et al. 2012). In a recent study of a typical Corn Belt watershed in the Iowa River Basin (IRB), Ha and Wu (2015) demonstrated the ability to harvest adequate levels of corn stover without adverse effects on water quality by implementing beneficial practices. Other studies have demonstrated that the use of cover crops can reduce water quality impacts of farming operations (Graham et al. 2007; Mann, Tolbert, and Cushman 2002), while reducing soil erosion, maintaining land productivity (Kaspar, Radke, and Laflen 2001; Snapp et al. 2005; Wyland et al. 1996), and reducing nutrient loadings.

In this chapter, we present research investigating the benefits of conservation practices that co-optimize the production of cellulosic energy feedstock and water quality improvements. Specifically, we look at landscapes produced that are consistent with a future 2040 economic scenario with \$60/dry ton (dt) and 1% annual yield increases (BC1 2040; see chapter 2). Our central hypothesis is that the use of conservation practices and better management protocols can reduce the environmental effects of biomass production, without a significant sacrifice in production. Two goals of this chapter are to identify conservation practices that minimize water quality impacts and maximize feedstock yields. Thus, for watersheds located in different regions, we ask how can we apply conservation practices to lands producing biomass feedstocks that improve water quality with the least possible reduction in feedstock supply?

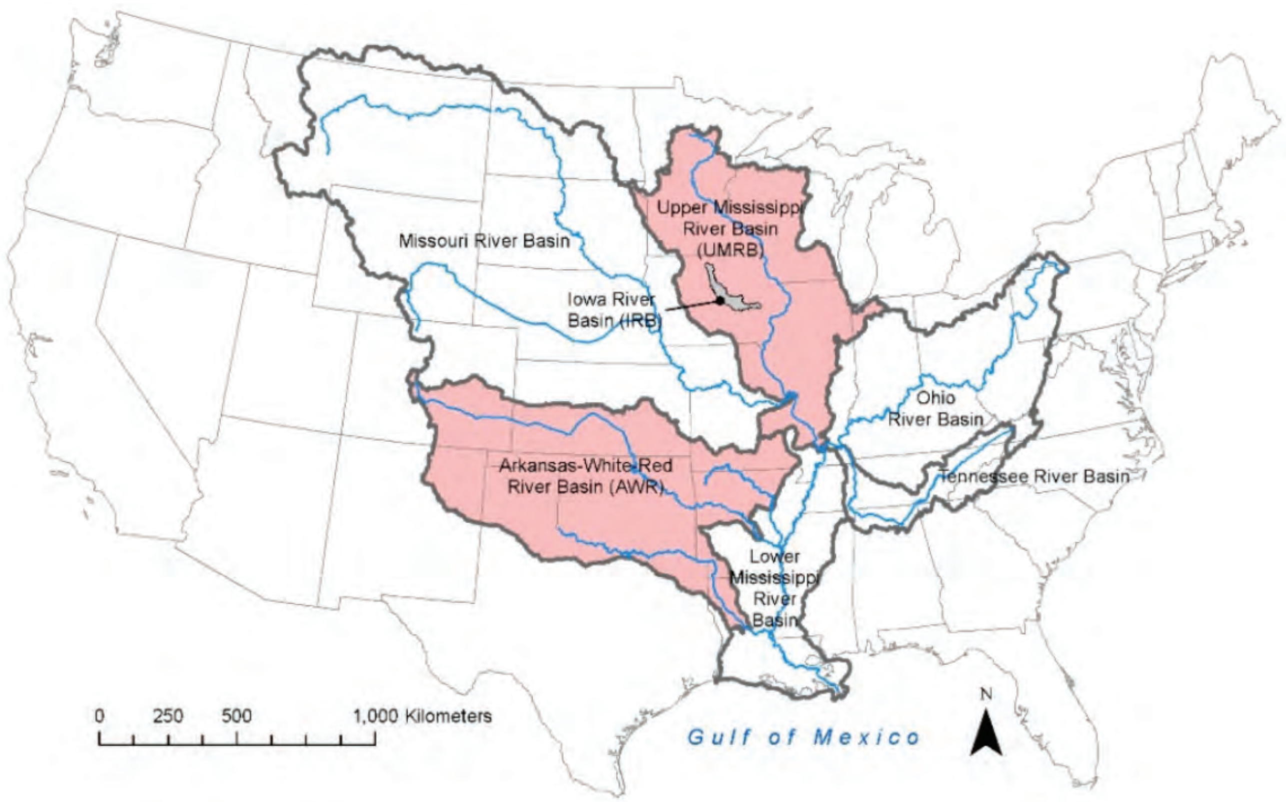
We simulate conservation-management practices relevant to feedstock cultivation for two dominant feedstock systems located in different regions within the Mississippi River Basin. We examine relationships (tradeoffs and complementarities) among the following environmental indicators: (1) productivity, (2) nitrate loadings, (3) phosphorus loadings, (4) suspended sediment loadings, and (5) water yield.

Our assessment seeks to understand how allocating conservation practices across future landscapes can help to achieve increases both in water quality and in biomass feedstock supply. Furthermore, we seek to understand general patterns that can be transferred to other locations to guide the management of cellulosic feedstocks. Implementing beneficial practices in a context-specific way is consistent with the conservation strategies devised by the U.S. Environmental Protection Agency's Hypoxia Task Force to reduce nutrient loadings from the Mississippi River Basin to the Gulf of Mexico by 20% by 2025 (EPA 2015).

5.2 Scope of Assessment

Unlike other assessments in this report, this analysis focuses on two areas with unique cellulosic feedstocks: the switchgrass-dominated Arkansas White and Red (AWR) River basin in the southern Great Plains and the corn stover-dominated IRB in the upper midwestern United States (fig. 5.1). *BT16* projections suggest that the potential for cellulosic feedstock production is high both in the AWR and in the UMRB, where the IRB lies.

Figure 5.1 | Two major river basins with different projected cellulosic biomass-production profiles



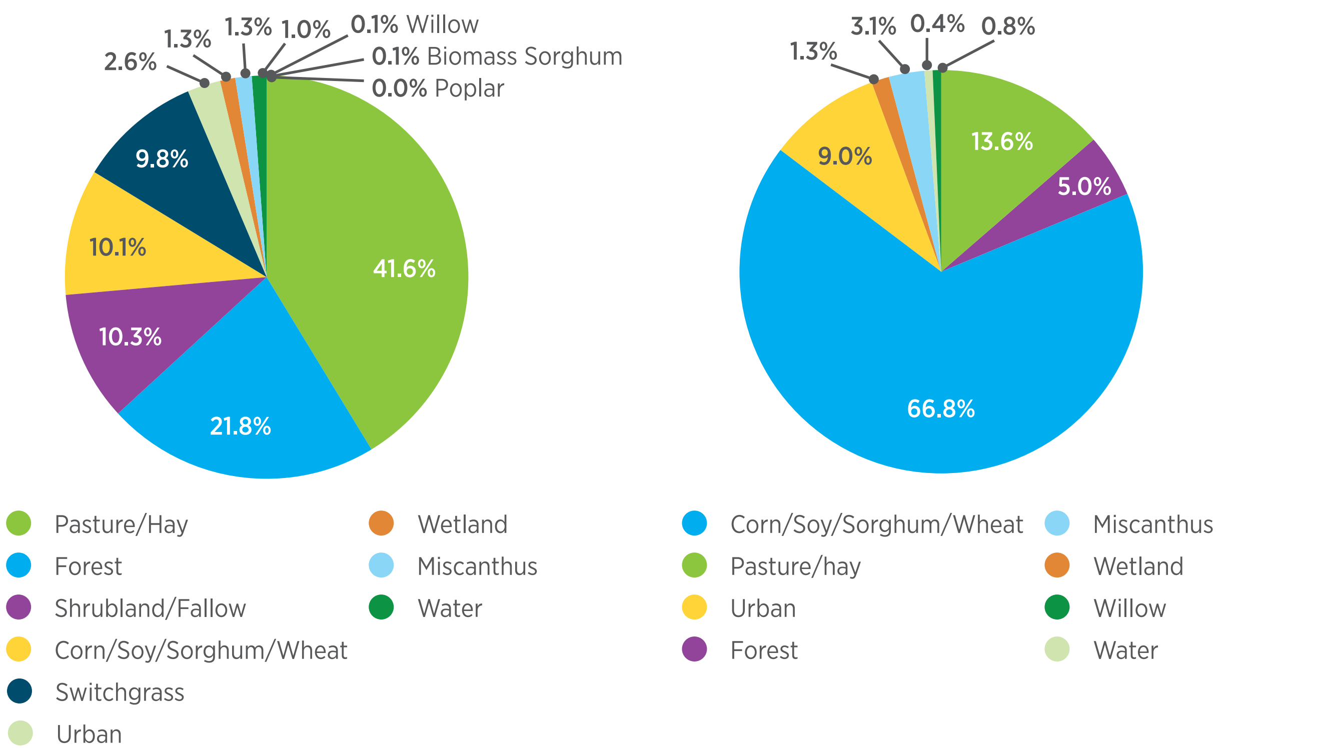
These basins are representative of two main agricultural systems that, according to *BT16* scenarios, would be dominated by distinct cellulosic feedstocks (chapter 3). In the UMRB, cellulosic-rich agriculture residue from corn stover is a promising near-term cellulosic feedstock (Graham et al. 2007). Located in the heart of the UMRB, the IRB resembles a typical landscape in the region with a corn grain- and soybean-production system. The BC1 2040 scenario estimates that farms growing corn and soybeans will continue to dominate the IRB (67% of the land area in the IRB) (fig. 5.2).

Farther south, the AWR is a promising region for sustainable biomass production and has high potential for reducing nutrient loadings into the Gulf of Mexico (Jager et al. 2015). The AWR is a large river basin with diverse land uses (fig. 5.2). Under the BC1 2040 scenario, the region will remain diverse, dominated by pasture (42%) and forest (22%). The dominant feedstock in the region, switchgrass, produces yields of 8 to 14 Mg/ha (~4 tons/acre) (Jager et al. 2010; Wullschleger et al. 2010).

5.3 Methods

Our task had four parts. First, for each river basin, we developed a Soil and Water Assessment Tool (SWAT) base model for the simulation area with at least 20 years of historical hydrology. Second, we downscaled the BC1 2040 scenario for each basin to produce a landscape for analysis. Third, we implemented SWAT with nominal conservation practices appropriate for respective production systems and with region-specific future energy crops and residues represented. Fourth, we simulated results for different conservation practices in SWAT. In our final step, we compared different conservation practices to understand

Figure 5.2 | Land use of BC1 2040 scenario in the (a) Arkansas-White-Red and (b) Iowa River Basins.



tradeoffs and complementarities among water quality and quantity indicators and biomass yields. This was done to promote the generalization of findings from these two regions to others with similar biomass feedstock profiles.

5.3.1 Environmental Indicators

Our analysis was designed to quantify environmental indicators (Dale et al. 2015) for different management practices associated with the BC1 2040-projected future landscape, which includes energy crops. To do this, we simulated a subset of the environmental indicators proposed by McBride et al. (2011). Our analysis focused on water quality and productivity indicators (table 5.1). Here, simulated annual values were averaged for the outlets of river basins.

Table 5.1 | Environmental Indicators of Water Quality, Quantity, and Productivity Are Average Annual Values over 20 Simulated Years.

Environmental Indicator	Units
Nitrate loadings	kg/ha
Total nitrogen loadings	kg/ha
Total phosphorus loadings	kg/ha
Total suspended sediment	t/ha
Productivity (biomass yield)	t/ha

Acronyms: kg/ha – kilograms per hectare; t/ha – tons per hectare.

5.3.2 SWAT Implementation

We implemented SWAT for a large river basin (AWR) dominated by switchgrass and a smaller watershed (IRB) dominated by production of cellulosic residues in a predominantly corn/soybean-growing region in the BC1 2040 scenario. SWAT is a physically based, semi-distributed hydrologic model to simulate changes in land management and the resulting changes in the hydrologic cycle and water quality (Gassman et al. 2007). We relied on models that have already been described in previous publications. The analyses reported here use SWAT to explore the effects of conservation practices on three classes of environmental indicators: feedstock production, water quality, and water quantity.

We used spatial data layers describing soils, slope (from elevation), and land cover to partition each sub-basin into areas with similar hydrologic response units (HRUs) to climate. Input data sources for SWAT include soil properties, stream network topology, land topography via a digital elevation model, meteorological data, and stream-monitoring data. Soil properties were obtained from the Soil Survey Geographic Database, using the State Soil Geographic dataset in the larger basin and the Soil Survey Geographic Database in the smaller one. Historical calibrations were performed independently for the two basins. For the IRB, climate data were obtained over a historical period from 1994 to 2013 from the National Oceanic and Atmospheric Administration's National Climatic Data Center. For the AWR, daily climate variables were obtained over the historical period from 1980 to 2011 from Daymet (Thornton, Running, and White 1997). Other climate variables, including wind speed, relative humidity, and potential evaporation, were simulated by SWAT's climate generator. Land cover data for 2014 were obtained from the Crop Data Layer generated by the U.S. Department of Agriculture's (USDA's) National Agricultural Statistics Service (NASS 2013). Simulations reported here were performed by using SWAT model version 2012, revision 622.

Soil units that comprised more than 10% of a sub-basin were represented as separate HRUs in SWAT. Maloney and Feminella (2006) showed that disturbances had greater impacts on sediment loadings in streams for watersheds with slopes greater than 5%. Therefore, we discretized slope into four categories: <1%; 1%–2%; 2%–5%; and >5%. Because a small amount of steep land can have large effects on sediment losses, we included all slope categories, regardless of area.

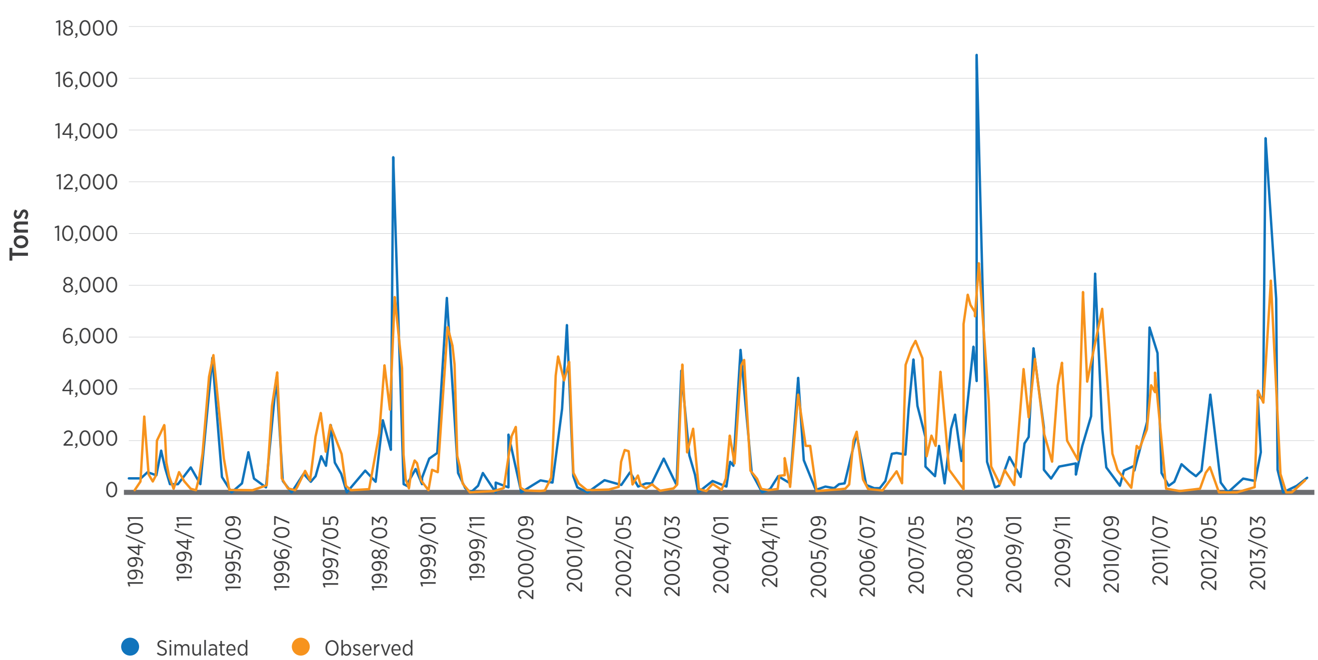
Defining land-management categories for HRU construction required that we cross-reference SWAT land-use classes with Crop Data Layer classes and manage agricultural classes modeled by the Policy Analysis System, the economic model. Land management in the BC1 2040 landscape was downscaled to USDA Common Land Unit parcels from county-level categories in the Policy Analysis System as described in the biodiversity chapter (chapter 10). In the AWR, we retained land-use classes that comprised more than 5% of the sub-basin. However, HRUs planted in dedicated energy crops were included, regardless of area. We represented a total of 15,437 HRUs across the AWR region and 3,346 HRUs in the IRB.

5.3.2.1 Sensitivity Analysis, Calibration, and Validation

Validation is more feasible in smaller, rather than larger, river basins. To illustrate, the IRB model was calibrated and validated for stream flow, sediment, nitrate, organic nitrogen, and total phosphorus at the U.S. Geological Survey's (USGS's) gauging station #05453100, which is located on the Iowa River at Marengo, Iowa, by using 20 years (1994–2013) of meteorological and monitoring data from the USDA Conservation Effects Assessment Project. The model calibration period is 1994–2003, and the validation period is 2004–2013. The calibrated parameters include the Soil Conservation Service runoff curve number; Universal Soil Loss Equation support practice factor; tile-drainage parameters; soil evaporation-compensation factor; plant uptake-compensation factor; surface-runoff coefficient; and parameters for channel flows, calculating sediment, nitrogen, and groundwater parameters, among others. Nash-Sutcliffe efficiencies (NSE) are commonly used for hydrologic modeling to explain its performance (∞ to 1; 1 is perfect matching). NSE values were 0.89, 0.69, 0.62, 0.40, and 0.85 (calibration) and 0.85, 0.73, 0.41, 0.66, and 0.86 (validation) for flow, suspended sediment, nitrate, organic nitrogen, and phosphorus, respectively. Coefficients of determination, R^2, ranged from 0.52 to 0.90 for flow, suspended sediment, nitrate, organic nitrogen, and phosphorus. Figure 5.3 presents calibration results for nitrate for the IRB model. SWAT-model calibration/validation evaluation values for monthly water quantity and quality parameters for IRB were well above the acceptable ranges reported by other researchers (Engel et al. 2007; Moriasi et al. 2007).

In the AWR basin, we used historical data in sensitivity analysis, calibration, and validation at two scales as described by Baskaran et al. (2010). We conducted parameter-sensitivity analysis and calibration for two smaller basins, the Current River watershed (Hydrologic Unit Code [HUC] #11010008) and Southern Beaver watershed (HUC #11130207). These produced NSE values of 0.74 and 0.78 (calibration, 1985–1996) and 0.75 and 0.65 (validation, 1997–2003). For the larger AWR region, we compared predictions for outlet gauges at 86 of the 173 sub-basins with long-term data. A strong relationship was observed between area-weighted USGS- and SWAT-predicted flow (adjusted R^2 = 0.83; root-mean-square-error = 90.48 cubic meters per second, 16,589 degrees of freedom), with a slope near 1 (0.91). In addition, we conducted sensitivity analysis focused on tradeoffs between switchgrass yield, nitrate export, and nitrogen fertilizer across the region (Baskaran et al. 2013). Because pasture was managed as switchgrass in the earlier Billion-Ton Update scenario, assumptions about fertilization or cattle density were important. This analysis sought to understand geographic patterns in the relationship between pas-

Figure 5.3 | Results of SWAT nitrate calibration for the IRB



ture intensification and to avoid densities that might lead to "breakthrough" of nitrate.

5.3.2.2 Biomass Crop / Residue Management

BC1 2040 future landscapes included several feedstocks, such as miscanthus and willow, that were not simulated in earlier resource assessments. Below, we summarize our implementation of these energy crops in the landscape. We also describe shared elements of crop management between the two basins, with individual refinements described in sections for each of the two basins.

A spin-up period is typically simulated before reporting results. This allows simulations to equilibrate away from the influence of initial conditions, and should be at least as long as the shortest crop rotation (4–10 years spin-up). The range of fertilizer values simulated for each crop bracketed those specified in the *BT16* volume 1 assessment.

Perennial grasses: Perennial grasses include multiyear crop rotations with planting in the first year and harvesting every year after planting. We assumed that new cultivars would be planted after 10 years for switchgrass or 15 years for miscanthus. Switchgrass and miscanthus were planted with no tillage. Results represent average yields over harvest years in the rotation. Perennial grasses require several years to become fully established, and no fertilizer was applied during the first 2 years of establishment to suppress weeds. In subsequent years, we compared simulations with different amounts of nitrogen fertilizer in the AWR. Miscanthus management in the AWR was based on the approach used by Cibin et al. (2016). In the IRB, region-specific crop-management practices and crop-growth parameters for miscanthus and switchgrass were derived from the Purdue Water Quality Field Station in Indiana (Trybula et al. 2015). The annual amount of nitrogen fertilizer applied in the Indiana study was 56 kilograms per hectare (kg/ha).

SRWCs: For willow, we assigned a 22-year rotation (Volk et al. 2006; Abrahamson et al. 2010). The plant is coppiced after the first year. Coppicing was simulated as a harvest-only operation with harvest index of 96% (Abrahamson et al. 2010). We simulated application of nitrogen after coppicing and applied a specified amount after every subsequent 3-year harvest cycle. For poplar, we simulated an 8-year rotation with growth parameters calibrated to match leaf area index and plant biomass (Guo et al. 2015). We varied the amounts of nitrogen depending on the conservation practice in the third and sixth years, as described in Section 5.3.3, and applied 17 kg/ha phosphorus in the third year.

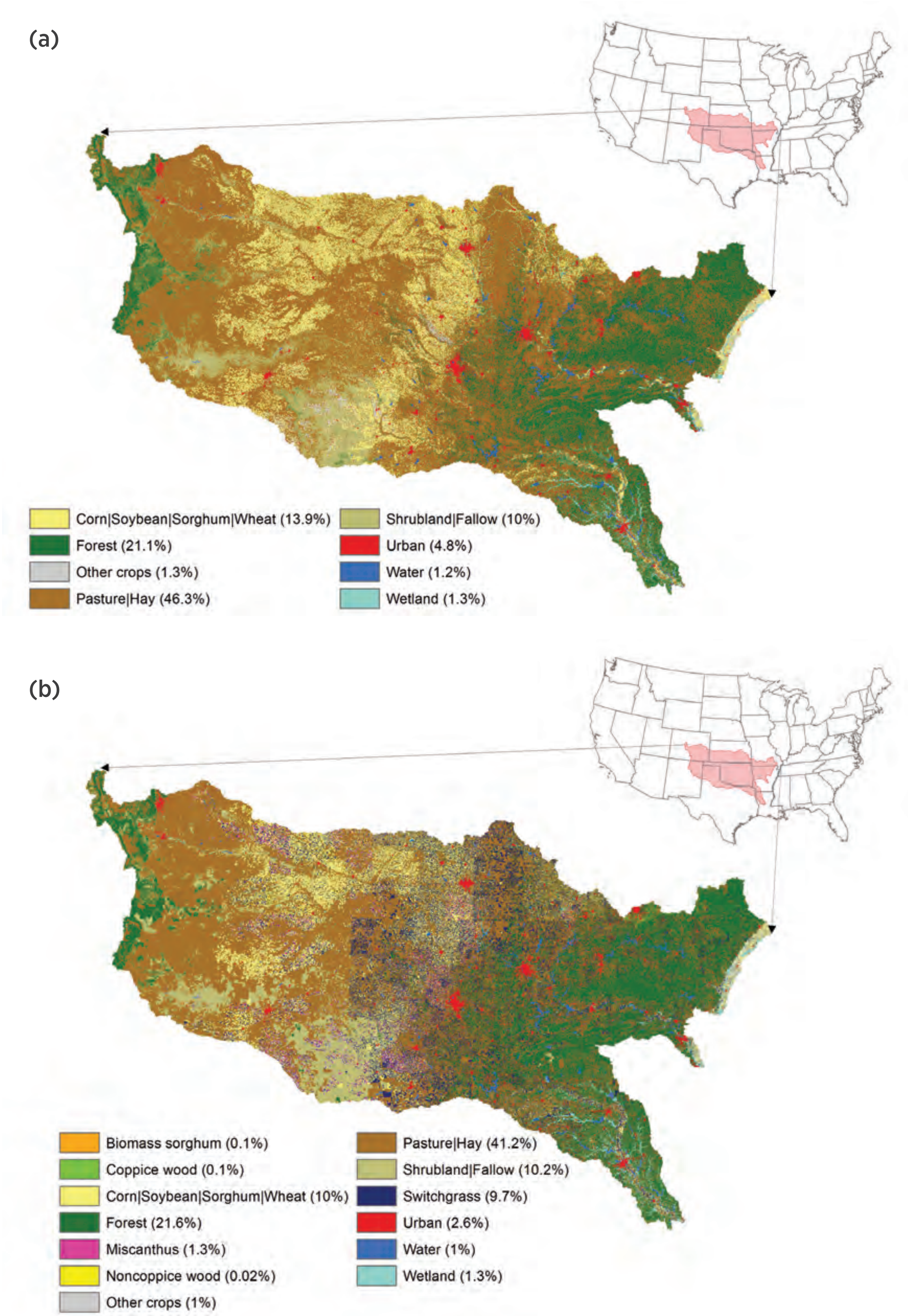
Energy sorghum: High-yield sorghum is an annual cellulosic feedstock (Venuto et al. 2008). We applied 67 kg/ha phosphorus each year and varied the amount of nitrogen applied. Our growth parameters for energy sorghum were derived from USDA values (White 2006).

Crop residues: We represented stover removal from annual crops in both regions. However, the IRB has a feedstock profile dominated by harvest of residues from corn. In both regions, we simulated split fertilizer application. In the IRB, fertilizer applications of nitrogen and phosphorus for corn, corn stover, and soybeans are consistent with BC1 2040 scenario presented in *BT16* volume I (table 5.3). Nitrogen fertilizer for corn grain was 142 kg/ha followed by 51 kg/ha after stover removal to account for nitrogen removed in the stover. In the AWR, we varied the application in fall for annual crops, corn, and sorghum.

5.3.2.3 AWR River Basin

We implemented SWAT for 173 sub-basins (USGS eight-digit HUCs) within the AWR drainage (fig. 5.4) (Jager et al. 2015). Details regarding the delineation of watersheds and hydrography is described in Baskaran et al. (2010) for the AWR.

Figure 5.4 | The (a) 2014 landscape based on cropland data layer and (b) spatial distribution of energy crops consistent with the BC1 2040 economic scenario



5.3.2.4 IRB

The location of the IRB in the UMRB and its cropland features are shown in figure 5.5. A SWAT base model was first constructed for the 2013 landscape. The terrain in the modeling area is relatively flat; 39.0% of the basin is <2% slope and 32.5% of the basin is with 2% to 5% slope. The model represented 90 sub-basins and 3,346 HRUs. Four-year corn and soybean rotations were simulated from 2010 to 2013. Sequences of the 4-year rotations were classified into 10 different rotation types. Land balance was conducted for each year of the rotation, with 99.6% accuracy in land accounting. The rotation sequence was applied to all 20 years of simulation. The model includes simulation of tile drainage.

Projected crop locations in the BC1 2040 scenario at the spatial resolution of counties were downscaled and simulated by using the IRB SWAT model. In the scenario, the watershed remains predominantly agricultural, with 66.9% corn and soybean rotation, 3.1% miscanthus, 0.8% willow, 13.6% pasture, 9.0% urban areas, 5.0% forest, 1.3% wetlands, and 0.4% water (fig. 5.2). In addition, its acreages for perennial grasses and SRWCs increase. We omitted poplar harvest, which represents a minimal resource (less than 0.01%).

Three different tillage operations were applied to corn and soybean areas in the IRB—for corn, operations included 9.5% conventional tillage, 27.4% no-tillage, and 63.1% reduced tillage, and for soybeans, operations included 4.2% conventional tillage, 40.8% no-tillage, and 55% reduced tillage. A land use/land cover map was created for the current year (2013) and for the future BC1 2040 scenario (fig. 5.6).

Figure 5.5 | The Iowa River Basin (IRB), a region dominated by annual agricultural crops (corn and soybean) located in the Upper Mississippi River Basin. Point sources of nitrogen (N) include waste-water treatment discharge (WWT) and industrial discharges.

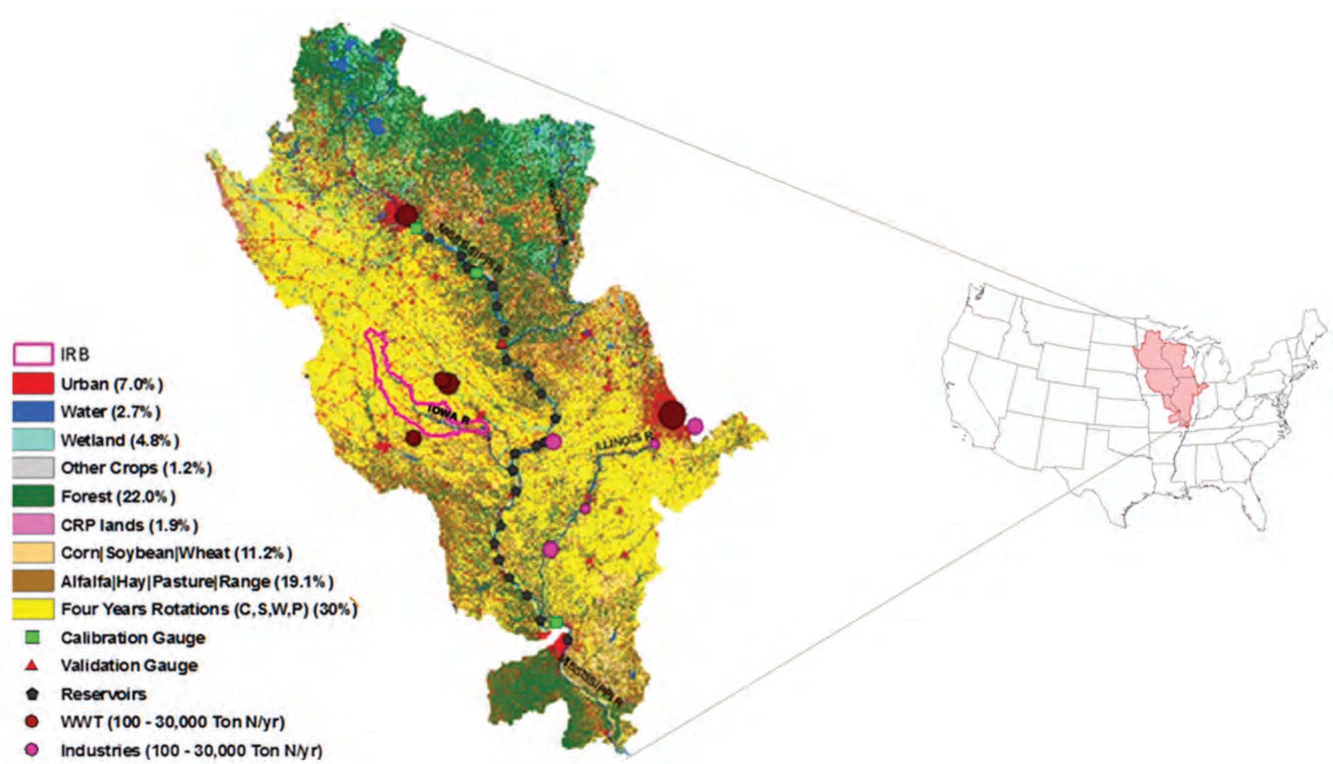
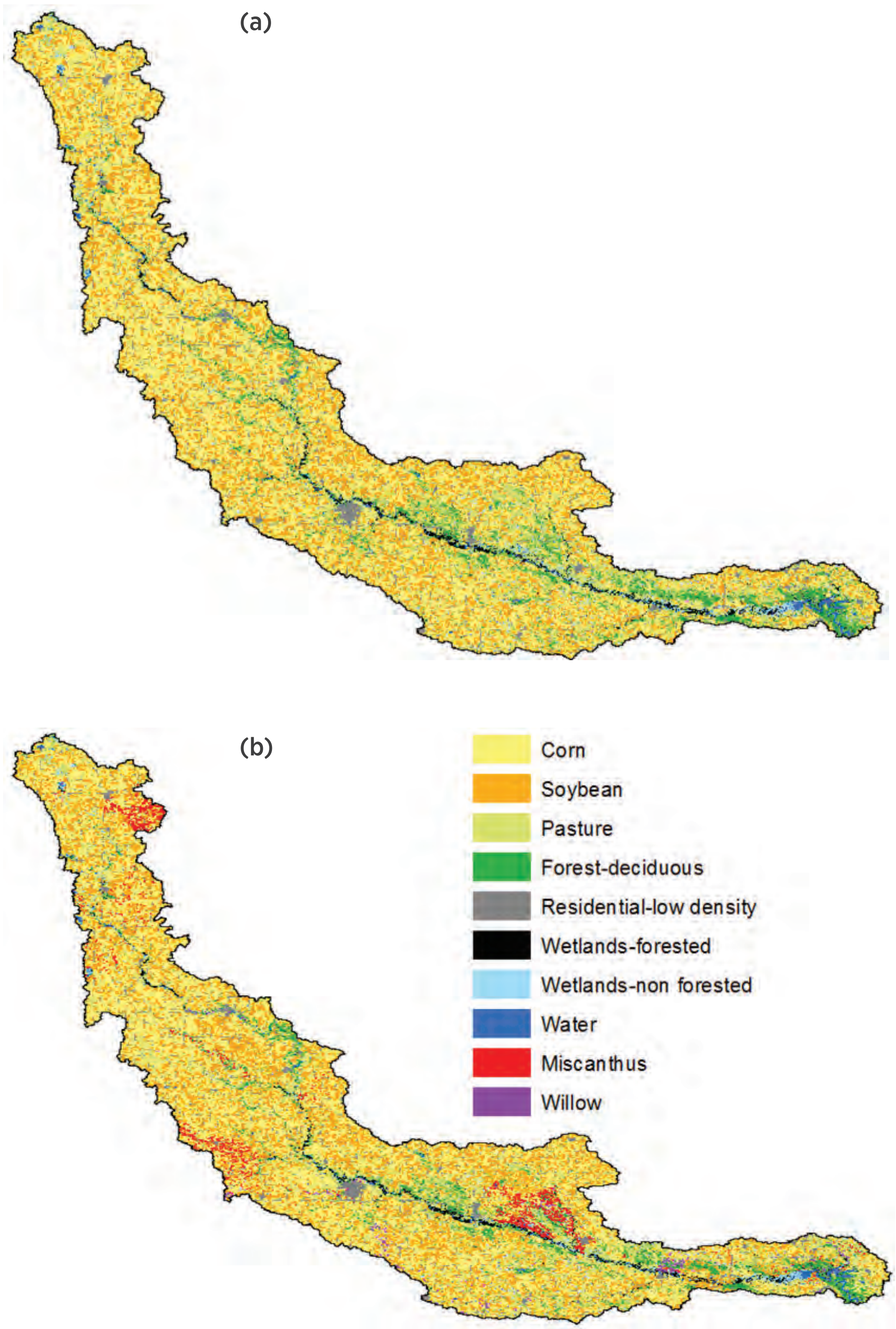


Figure 5.6 | Distribution of crops and other land use//land cover classes in the Iowa River Basin in (a) 2013 and (b) the future scenario BC1 2040



5.3.3 Conservation Practices

The primary objective of this research was to compare management practices and evaluate feedstock yields and water quality indicators. Below, we describe how this was done for the larger river basin and the smaller corn-soy-dominated watershed.

5.3.3.1 AWR Basin

After producing a SWAT setup for the BC1 2040 landscape using the ESRI© ArcGIS interface for SWAT, we used scripts to generate SWAT input files for simulations with different practices shown in table 5.2. We present results for all combinations of practices and what we refer to as "superlative" practices (i.e., those with the highest feedstock yield, those with the lowest nitrate loadings, those with the lowest total phosphorus (TP) loadings, and those the lowest total suspended sediment (TSS) loadings, respectively). Each set is optimized for a different indicator. In addition, we developed a visualization that allows stakeholders to set limits on water quality and yield indicator values. Stakeholders can evaluate the consequences of conservation practices capable of producing outcomes within specified limits, and the correlated responses of other indicators listed in table 5.1.

Filter strips: Filter strips were simulated by setting the ratio of the field area to the filter strip area to 40 to achieve 2.5% of the field area. It was assumed that 50% of the HRU drained to the most concentrated 10% of the filter strip. None of the concentrated flow was fully channelized such that it would bypass filtering effects of the filter strips (Kalcic, Frankenberger, and Chaubey 2015).

Fertilizer: Fertilization practices are described in section 5.3.3.2 for each feedstock. We varied these practices for each crop as described in table 5.2. In general, fertilizer was applied once in spring for perennial grasses. For residues, we varied only the second fertilizer application, which occurred in fall. Fertilizer amounts apply to the whole crop and not just residues.

Tile drainage: For annual crops, we simulated two alternative implementations of tile drainage controls to evaluate the potential for improving water quality outcomes. In one set of simulations, tile drains were simulated only for HRUs with low slopes <1% in all HRUs; in another, tile drains were simulated only for HRUs with slopes <2% (table 5.3). We assumed that perennial root systems can be used without tile drainage and that such drainage would be plugged.

Table 5.2 | Simulated Levels of Each Conservation Practice Applied in the AWR River Basin

Biomass feedstock	Filter strip	N fertilizer (kg/ha)	Tillage practice	Tile drainage
Switchgrass	None	0, 20, 60, 100	No-till	None
Poplar	With and without	0, 20, 60, 100	No-till	None
Miscanthus	Without	0, 20, 70, 120	No-till	None
Willow	With and without	0, 30, 70, 110	No-till	None
High-yield sorghum	None	101, 135, 168, 202, 235	No-till Conventional	Lands (HRUs) with <1% slope Lands (HRUs) with <2% slope
Sorghum stubble	None	105, 120, 135	No-till Conventional	Lands (HRUs) with <1% slope Lands (HRUs) with <2% slope
Corn stover	None	60, 85, 110	No-till Conventional	Lands (HRUs) with <1% slope Lands (HRUs) with <2% slope

5.3.3.2 IRB

Four different conservation practices were simulated and compared to the BC1 2040 scenario (table 5.3). They include cover crop, a riparian buffer of 30 m and 50 m, controlled-release nitrogen fertilizer, and controlled tile drainage. Neither buffers nor cover crops were harvested.

Riparian buffers: Riparian buffer installation is not mandatory in this region and is therefore rare. In simulations with a riparian buffer, the buffer was installed in sub-basins along the main stem of the Iowa River, in accordance with National Resources Conservation Service's guidelines for Iowa. The riparian buffer was planted in switchgrass. We compared two buffer widths: a 30-m (RB30) and 50-m (RB50) riparian buffer (table 5.3).

Cover crops: Rye is a common choice of cover crop in this region. For this scenario (CC in table 5.3), we assumed that the cover crop was killed in the spring but that residue remained on the soil.

Fertilizer: Corn grain, stover, and miscanthus receive nitrogen fertilizer. Nitrogen fertilizer is applied to corn at 142 kg/ha. When stover is harvested, a supplemental nitrogen fertilizer of 51 kg/ha is applied to compensate nitrogen loss due to removal of stover from the field. Miscanthus requires minimal nitrogen of 56kg/ha. Willow does not receive nitrogen fertilizer. Fertilizer is applied after harvest in fall and in the spring. In a controlled-release nitrogen fertilizer scenario (CR in table 5.3), the nitrogen fertilizer is applied after harvesting residue in fall and at spring planting. Simulated nitrogen release occurred within two months.

Tile drainage: Much cropland in the Midwest is tile drained, and this drainage aggravates downstream water quality problems by creating a bypass around potential nutrient uptake and conversion pathways within soils. Therefore, closing tiles when they are not needed could be an important practice. Three tile drainage options were simulated: no tile control (all tile drains are open [Open]), no tile (all tile drains

Table 5.3 | Simulated Conservation Practice Scenarios in the IRB. Conservation Practices Added to the BC1 2040 Scenario (BC40) Included a 30-m Riparian Buffer (RB30), a 50-m Riparian Buffer (RB50), a Cover Crop (CC), Controlled-Release of N Fertilizer (N CR), Closing of All Tile Drains (Tile), and Tile Drains Open for Land with <2% Slopes (Tile2%).

IRB conservation practice	Riparian buffer	Cover crop	N fertilizer	Tile drainage
BC40	No	No	Corn: 142 kg/ha Stover: 51 kg/ha Miscanthus: 56 kg/ha	Open
RB30	30 m, switchgrass	No	Same as above	Open
RB50	50 m, switchgrass	No	Same as above	Open
CC	No	Rye	Same as above	Open
N CR	No	No	Controlled release for corn: spring and fall, 2 months	Open
Tile	No	No	Corn: 142.3 kg/ha Stover: 51.1 kg/ha Miscanthus: 56 kg/ha	All plugged
Tile2%	No	No	Same as above	≥ 2% slope plugged

plugged [Tile]), and partial mitigation control (tile closed in areas where land slope is greater than 2% [Tile2%]) (table 5.3).

Historical climate data (1994–2013) were used to predict the long-term hydrology and the impact of the BC1 2040 scenario and conservation practices (case studies) on water quality. Modeled results are presented for water yield, TSS, nitrate, total nitrogen, organic phosphorus, and mineral phosphorus, and are compared among scenarios with different conservation practices.

5.4 Results

The two river basins differ in feedstock profiles. Below, we present and discuss the responses of feedstock yield and water quality indicators to conservation-management practices deemed most relevant to improve water quality in each river basin. In the AWR and IRB, we present results for three classes of feedstock: (1) perennial grasses, (2) SRWCs, and (3) crop residues, each with relevant conservation practices. Simulated conservation practices include (1) riparian buffers, (2) planting a cover crop, (3) tile-drainage control, and (4) use of slow-release nitrogen fertilizer.

5.4.1 Arkansas-White-Red River Basin

We represented the effects of conservation practices for each of three classes of feedstock in the AWR. All combinations of practices in table 5.2 were simulated, and our primary dataset includes the following information: 1) crop, 2) the HRU ID, 3) the value of each of the practices in table 5.2 (depending on crop), and 4) each of the simulated indicator values.

For each crop-HRU, we identified which combination of practices produced the best results in terms of each indicator (i.e., minimum nutrient and TSS or maximum biomass yield). We refer to these as 'superlative practices'. Thus, for a given crop, there is one practice with maximum yield for each HRU (i.e., slope-soil combination managed for the crop of interest within a subbasin). For each indicator, the total number of superlative practices associated with a given crop would be the number of HRUs in the crop. The set of superlative practices excludes combinations of practices that did not do best with respect to any indicator.

For each crop, we present two types of plots summarizing superlative practices. First, we produced a frequency histogram of HRU counts by practice combination. For crops for which we evaluated more than one practice, facet plots are used to display frequencies across multiple dimensions (practices). Second, the distribution of values for each of the four indicators is presented for the superlative subset of simulated data.

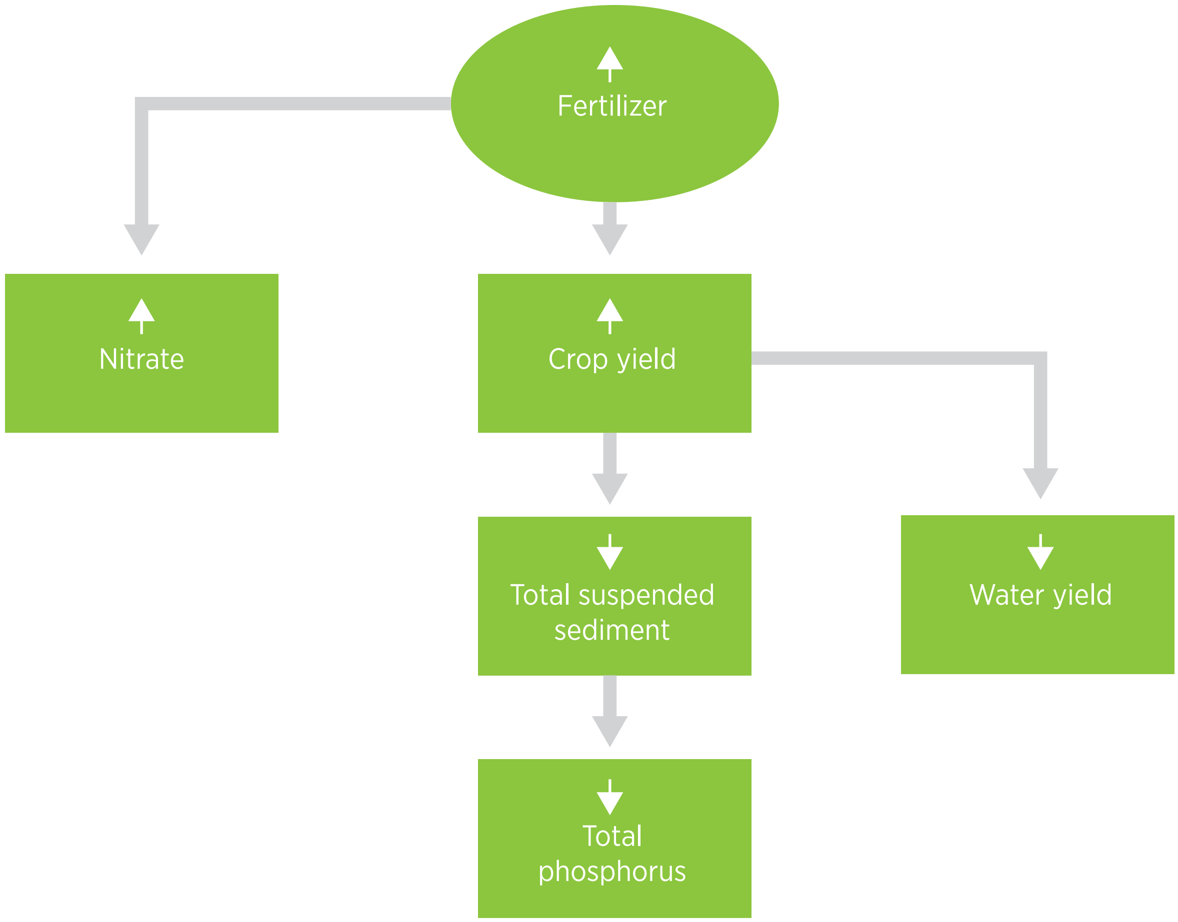
Generally, the associations observed for perennial grasses and SRWCs were described by the path diagram in figure 5.7. However, we did not observe this pattern for crop residues.

5.4.1.1 Perennial Grasses

SWAT-modeled responses of water quality and yield to switchgrass fertilizer were correlated in expected ways (fig. 5.8). For switchgrass, we observed a positive relationship between TSS and TP because TP is primarily bound to sediment. We observed negative relationships between TSS and switchgrass yield, and higher fertilizer amounts resulted in higher switchgrass yields and lower TSS.

For the grasses, the practices resulting in the highest yields were those with the highest levels of nitrogen fertilizer (fig. 5.8). For example, the light green bar shows that all HRUs with a maximum yield were managed by applying the highest fertilizer level. This was generally true for miscanthus as well. For both switchgrass and miscanthus, the practice resulting in the lowest nitrate level was the one with the lowest level of fertilizer. Patterns for TSS and TP were weaker, but both tended to be lower where yields were high (i.e., in the high-fertilizer scenario) (fig. 5.8).

Figure 5.7 | Path diagram describing the expected effects of nitrogen fertilizer on biomass yield and indicators of water quality and quantity



In general, miscanthus yields (fig. 5.9b) were significantly higher than for switchgrass (fig. 5.9a). Scenarios with minimum nitrate (no fertilizer) had very low yields. Yields in scenarios with minimum TP and TSS were not impacted as much as those in minimum nitrate scenarios (fig. 5.9). Practices that minimized nitrate (no fertilizer) produced much higher TP and TSS (fig. 5.9). This is consistent with the idea that more vegetative growth prevents runoff of sediment and sediment-bound nutrients. Counter to our initial intuition, this suggests that adding sufficient nitrogen fertilizer to grasses can help to lower export of sediment and sediment-bound TP by increasing vegetative cover. Nitrate loadings were considerably higher in scenarios with maximum yield. Fertilizer amounts that minimized TP and TSS were intermediate both in yield and nitrate (fig. 5.9).

For SRWCs, we compared scenarios with and without filter strips, in addition to the four levels of fertilizer. Figure 5.10 shows these results. HRUs with minimum nutrient and sediment loadings appeared with higher frequency when filter strips were simulated than when they were not. The majority of HRUs with maximum yield and minimum TSS were produced in simulations with high fertilizer amounts. Nearly all HRUs with minimum nitrate loadings occurred in simulations with no fertilizer and a filter strip. Because no practices without filter strips appeared among superlative practices for willow, we did not include this plot in figure 5.10. Note that

Figure 5.8 | Distribution of superlative practices (fertilizer amount) with respect to each indicator for (a) switchgrass and (b) miscanthus. The maximum possible frequency for a given fertilizer level is the number of HRUs with the crop.

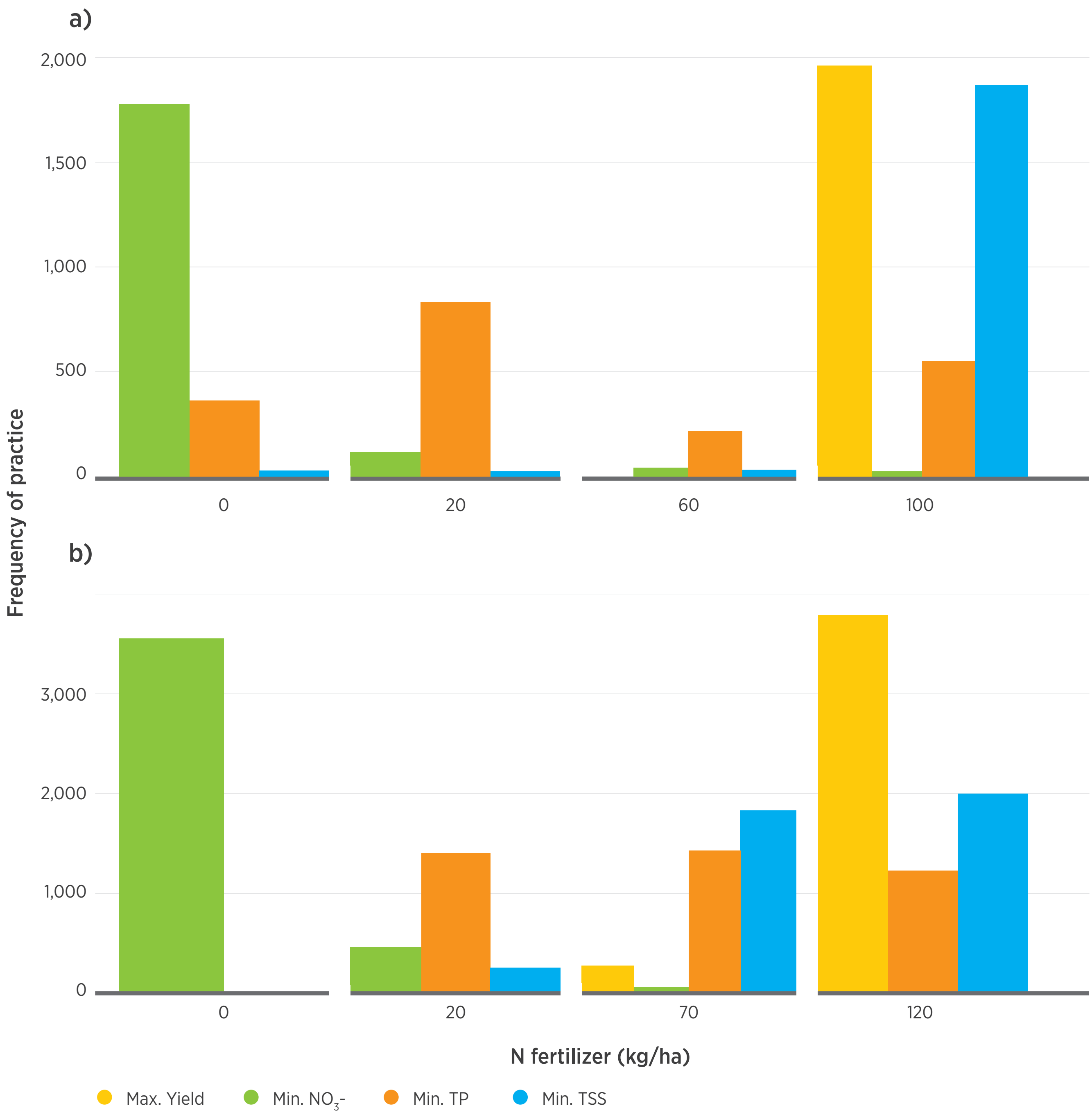


Figure 5.9 | Indicator values for the combination of practices (i.e., superlative practices) best able to meet the objective described by the x-axis for (a) switchgrass and (b) miscanthus. Indicators (y-axes) include feedstock yield, nitrate (NO_3^-), total phosphorus (TP), and total suspended sediment (TSS). Units for indicators are given in table 5.1.

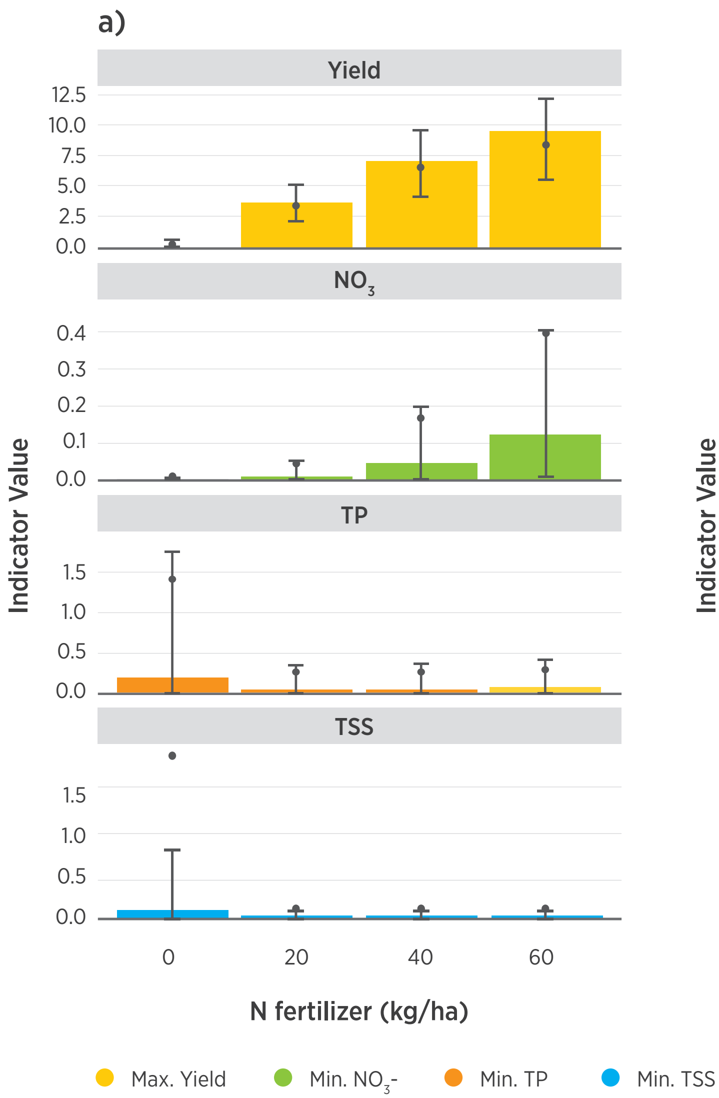
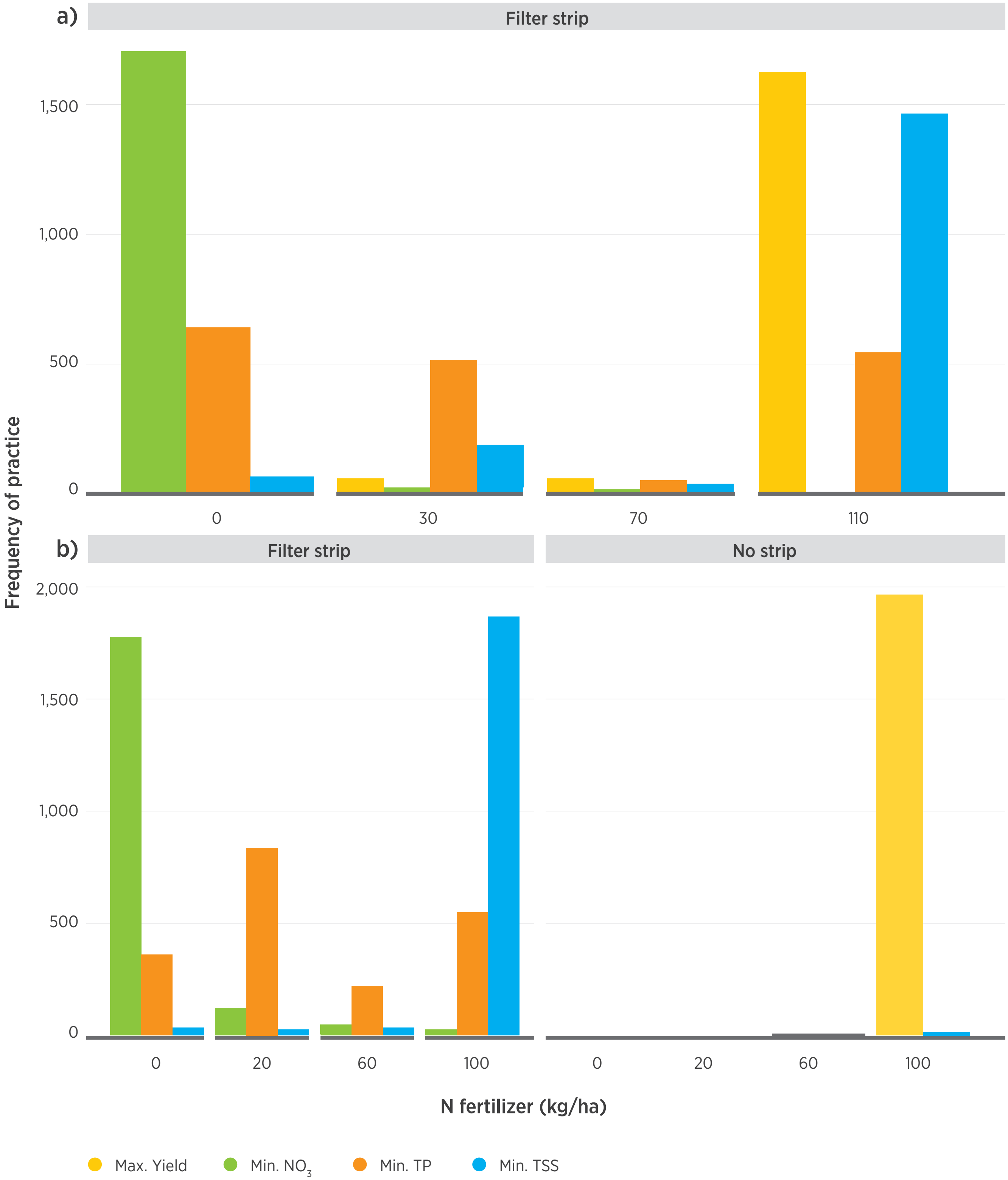


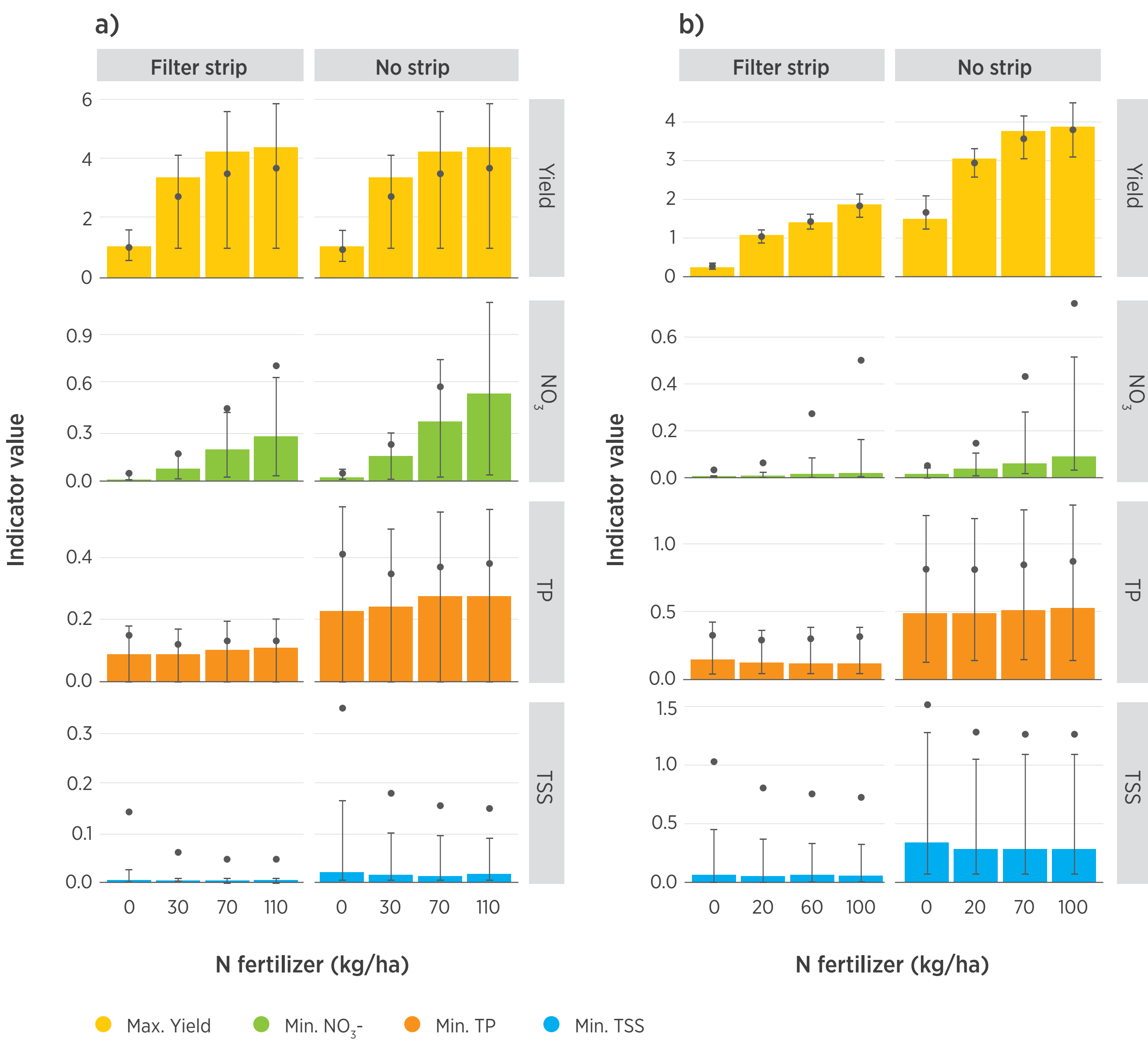
Figure 5.10 | Representation of superlative practices (filter strip versus none, fertilizer amount) with respect to each indicator for (a) willow and (b) poplar. The maximum possible frequency for a given combination of practices is the number of HRUs with the crop.



SWAT-simulated yield may be lower with a filter strip if the filter strip is not harvested, as it is not here.

Simulated filter strips were very effective at reducing nutrients and sediment for both willow (fig. 5.11a) and poplar (fig. 5.11b). However, there was a larger cost in terms of reduced yield for poplar. Likely, this is because we simulated willow as a coppice SRWC, and therefore did not harvest the whole tree, producing better water quality outcomes. However, simulated TP and TSS loadings were higher for willow than for poplar for each combination of practices (fig. 5.11).

Figure 5.11 | Indicator values for the best combination of practices per the objective described by the x-axis (i.e., "superlative practices") for (a) willow and (b) poplar. Indicators (y-axes) include feedstock yield, nitrate (NO_3^-), total phosphorus (TP), and total suspended sediment (TSS). Units for indicators are given in table 5.1.



5.4.1.2 Annual Energy Crops and Residues

Below, we present results comparing practices for annual crops and residues, including high-yield sorghum, corn stover, and sorghum stubble. As for perennial crops, we analyzed superlative practices (i.e., practices that are best with respect to each indicator).

High-yield energy sorghum: High-yield energy sorghum was the only dedicated annual crop that occurred in the AWR in BC1 2040. Frequencies for superlative practices with respect to each indicator are displayed in figure 5.12. Most maximum yields occurred in no-till scenarios and at the highest level of fertilizer simulated. No-till practice is well represented among minimum TP and TSS scenarios as well. However, nitrate followed a different pattern, with the lowest values under conventional tillage and low nitrogen fertilizer. Average nutrient and sediment values follow similar patterns, with no-till scenarios having lower average TP (fig. 5.13).

Corn stover: We simulated corn with tile drains implemented for slopes <1% and slopes <2%, each with conventional tillage and no-till and with three different levels of fall-applied nitrogen fertilizer. Results are shown in figure 5.14. Simulations with tile drains on lands with <2% slope were rarely among the superlative scenarios. Among scenarios with tile drains on lands <1% slope, simulations with the highest fertilizer consistently produced the highest yields. Conventional tillage produced maximum yields and minimum nitrate for more HRUs than did no-till. Minimum TSS values occurred most frequently for HRUs managed with no-till and less fertilizer. Minimum TP also occurred more frequently at low levels of fertilizer, but more often in simulations with conventional till (fig. 5.14 and 5.15).

Grain sorghum stubble: We simulated grain sorghum with tile drains implemented for slopes <1% and slopes <2%, each with conventional tillage and no-till and with three different levels of fall-applied nitrogen fertilizer. We observed better outcomes with tile drainage on lands with <1% slope (fig. 5.16 and 5.17). Frequencies (HRUs) for superlative practices with respect to each indicator are displayed in figure 5.16. We consistently observed the highest yields in HRUs with high fertilizer and no-till management. TSS was minimized most frequently for HRUs managed with no-till. Minimum TP included HRUs managed with either no till and high fertilizer or conventional till with low fertilizer.

5.4.1.3 AWR Summary of TradeOffs and Complementarities

The AWR analysis was designed to quantify tradeoffs among indicators, especially between feedstock yield and water quality indicators. First, we calculated the percentage improvement between the best and worst conservation practices for each crop. In general, conservation practices (reduced fertilizer) produced large decreases in sediment and nutrients for perennial grasses and for the two SRWCs (fig. 5.18). The smallest improvements were realized for TSS and TP loadings by sorghum stubble. Note that these differences may simply reflect the range of practices simulated, rather than potential for growing each of these crops with more environmentally favorable outcomes.

Tradeoffs and complementarities differed among the four perennial crops (fig. 5.19 a–d). For poplar (fig. 5.19c), practices showed strong tradeoffs that maximized yield (yellow bar) and produced the highest nutrient and sediment loadings. Conversely, practices with the lowest nitrate produced the lowest yield as well. One commonality across perennials (fig. 5.19 a–d) is that the practice that minimized TSS (blue in fig. 5.19c) performed reasonably well in maximizing yields and minimizing nitrate and TP, suggesting a complementarity between TSS and other indicators. Tradeoffs were strongest between nitrate and yield, with very low yields in simulations with practices that resulted in low nitrate, probably due to low fertilizer levels (fig. 5.19 a–d).

Figure 5.12 | Superlative practices for high-yield energy sorghum managed under all combinations of three practices: (1) conventional tillage and no-till; (2) tile drainage for two slope classes; and (3) five levels of fertilizer application. The maximum possible frequency for a given combination of practices is the number of HRUs with the crop

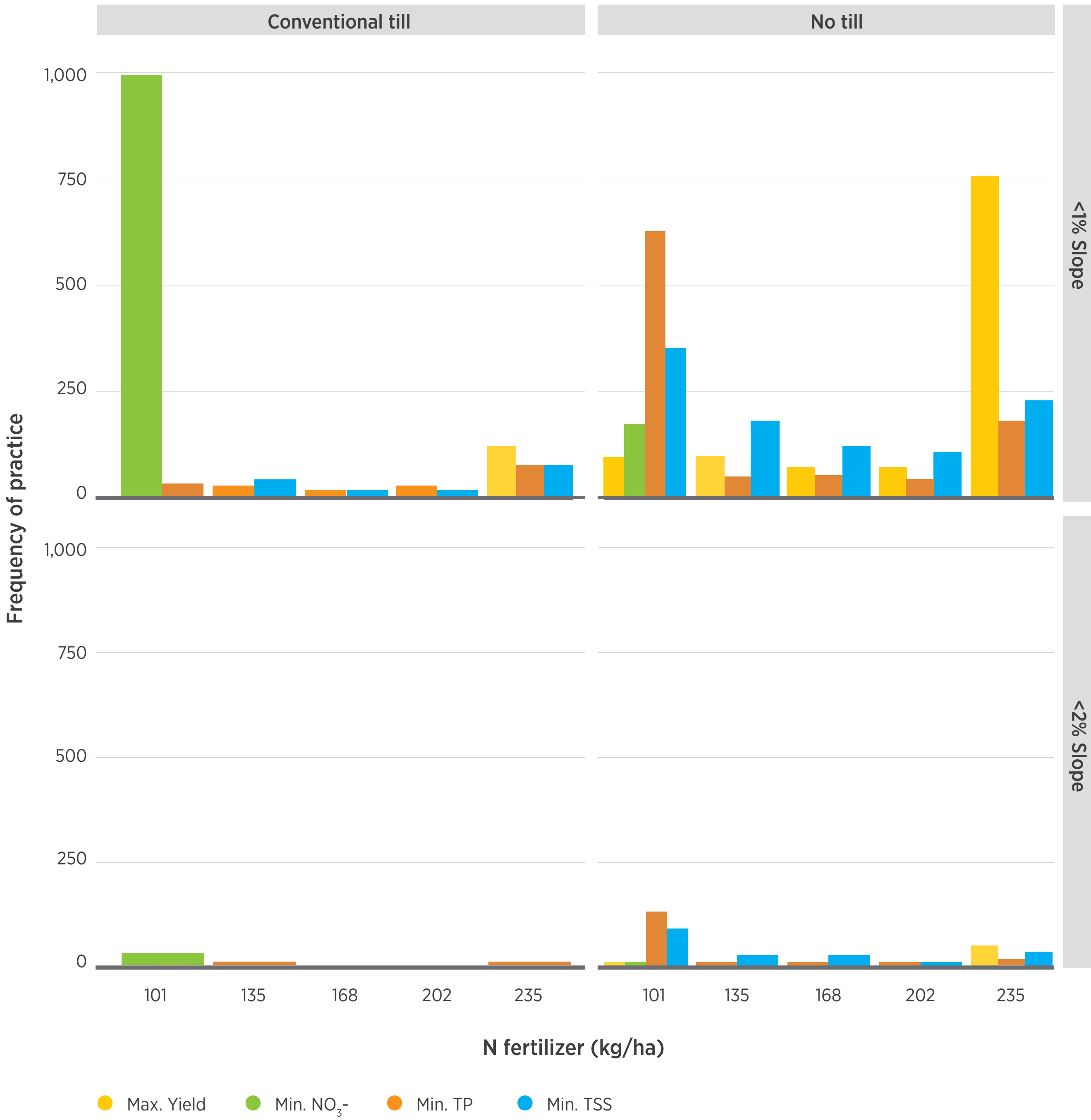


Figure 5.13 | Distributions of indicator values for high-yield sorghum scenarios. Whiskers indicate minimum and maximum values and the box encloses the 25th and 75th percentile with a horizontal line at the median. Indicators (y-axes) include log_{10}-transformed nitrate (NO_3^-), total phosphorus (TP), total suspended sediment (TSS), and feedstock (grain) yield. Units for indicators are given in table 5.1.

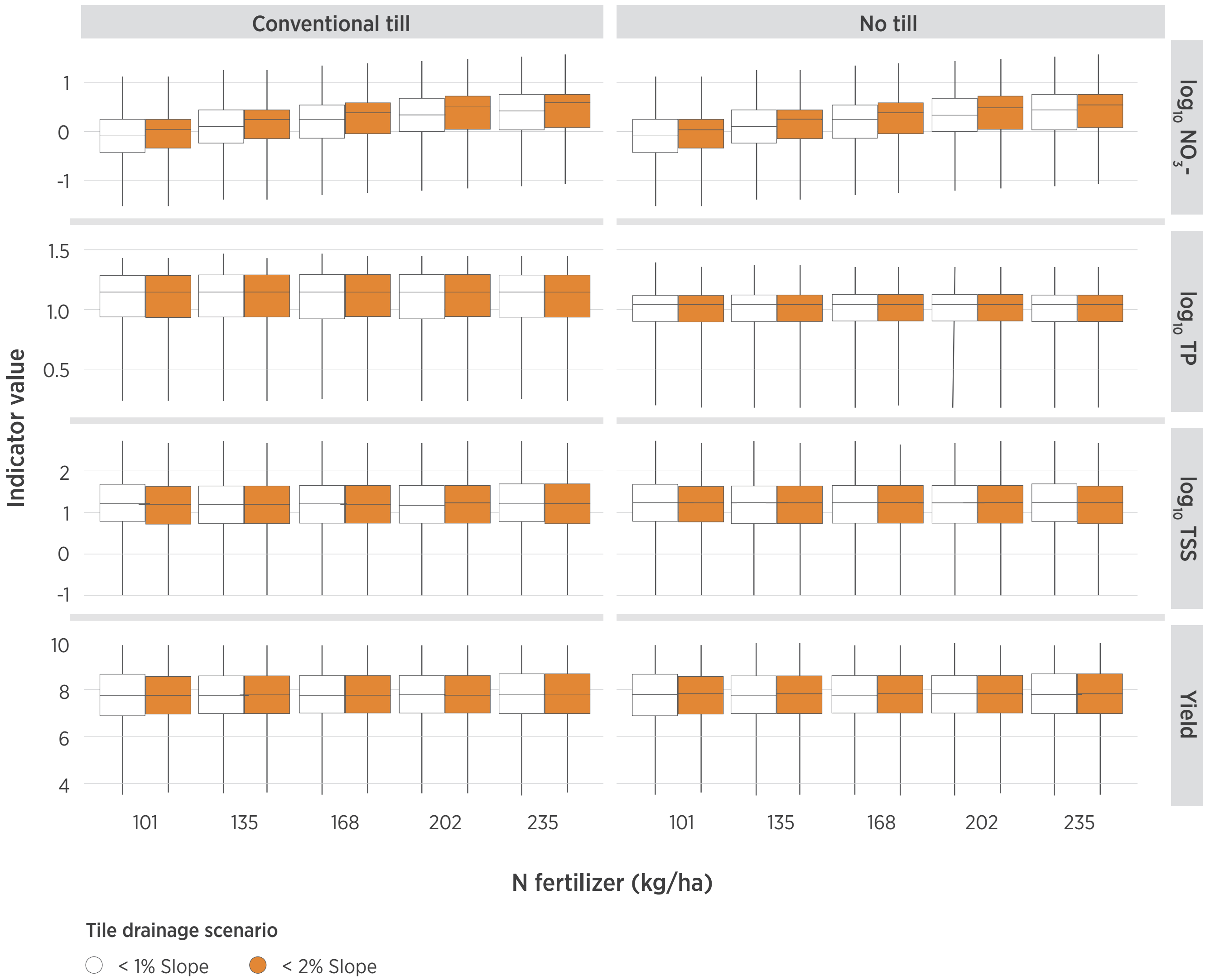


Figure 5.14 | Superlative scenarios for corn stover managed under all combinations of three practices: (1) conventional tillage and no-till; (2) tile drainage for two slope classes; and (3) three levels of fertilizer application. The maximum possible frequency for a given combination of practices is the number of HRUs with the crop.

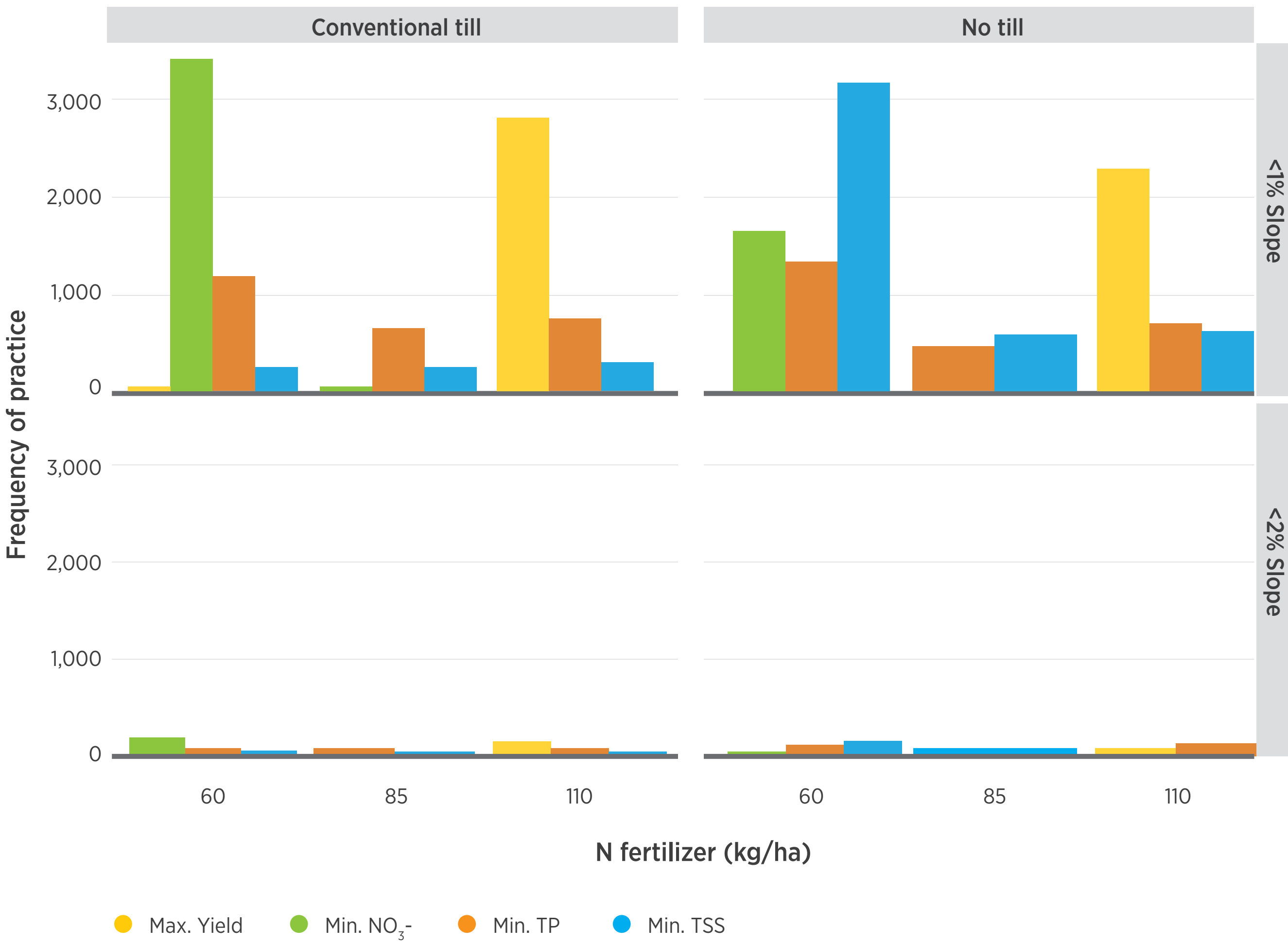


Figure 5.15 | Distributions of indicator values for corn-stover scenarios. Whiskers indicate minimum and maximum values and the box encloses the 25th and 75th percentile with a horizontal line at the median. Indicators (y-axes) include log_{10}-transformed nitrate (NO_3^-), total phosphorus (TP), total suspended sediment (TSS), and feedstock (residue) yield. Units for indicators are given in table 5.1.

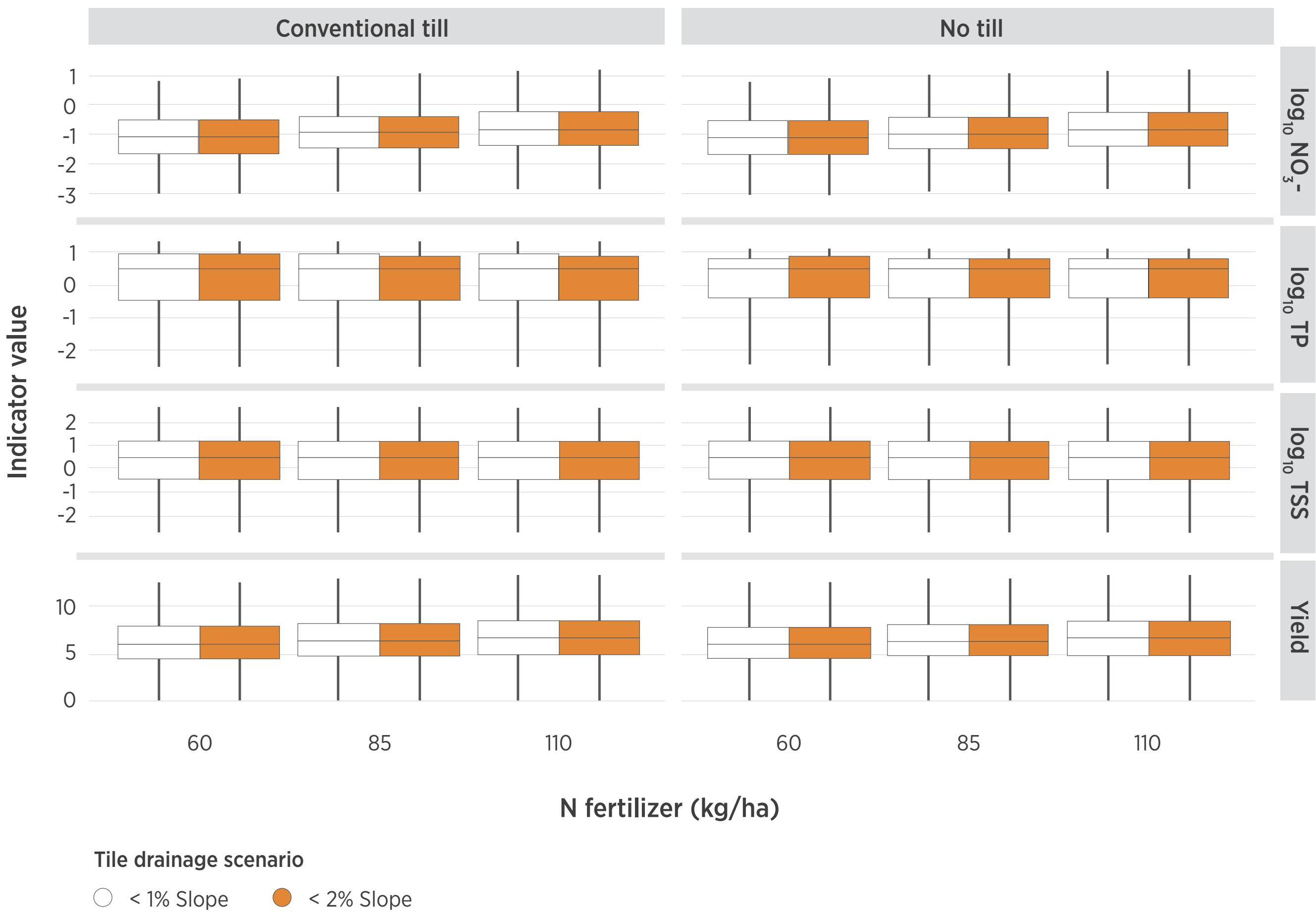


Figure 5.16 | Superlative scenarios for grain sorghum stubble managed under all combinations of three practices: (1) conventional tillage and no-till; (2) tile drainage for two slope classes, and (3) three levels of fertilizer application. The maximum possible frequency for a given combination of practices is the number of HRUs with the crop.

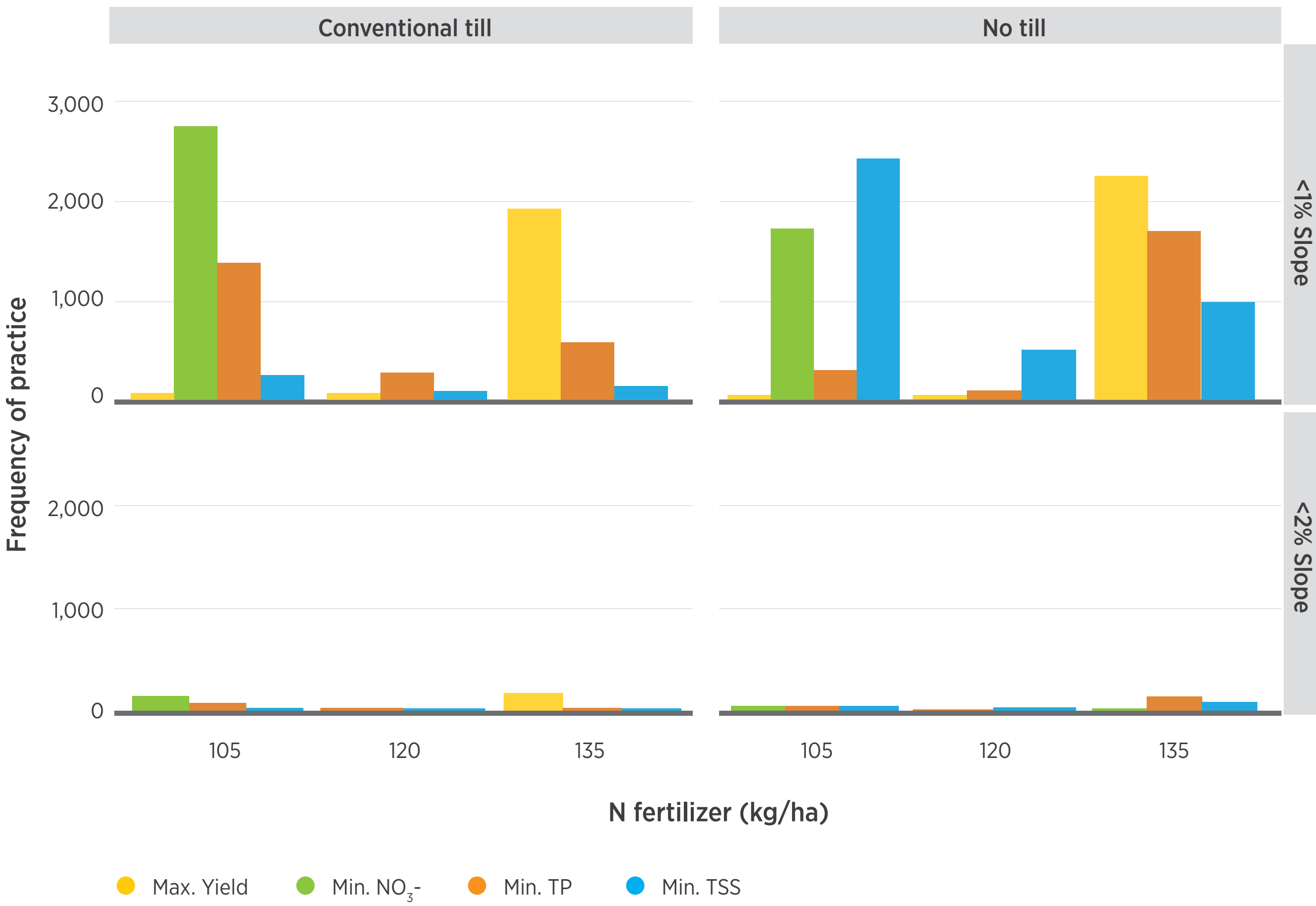


Figure 5.17 | Distributions of indicator values for grain sorghum scenarios. Whiskers indicate minimum and maximum values and the box encloses the 25th and 75th percentile with a horizontal line at the median. Indicators (y-axes) include log_{10}-transformed nitrate (NO_3^-), total phosphorus (TP), total suspended sediment (TSS), and feedstock (grain) yield. Units for indicators are given in table 5.1.

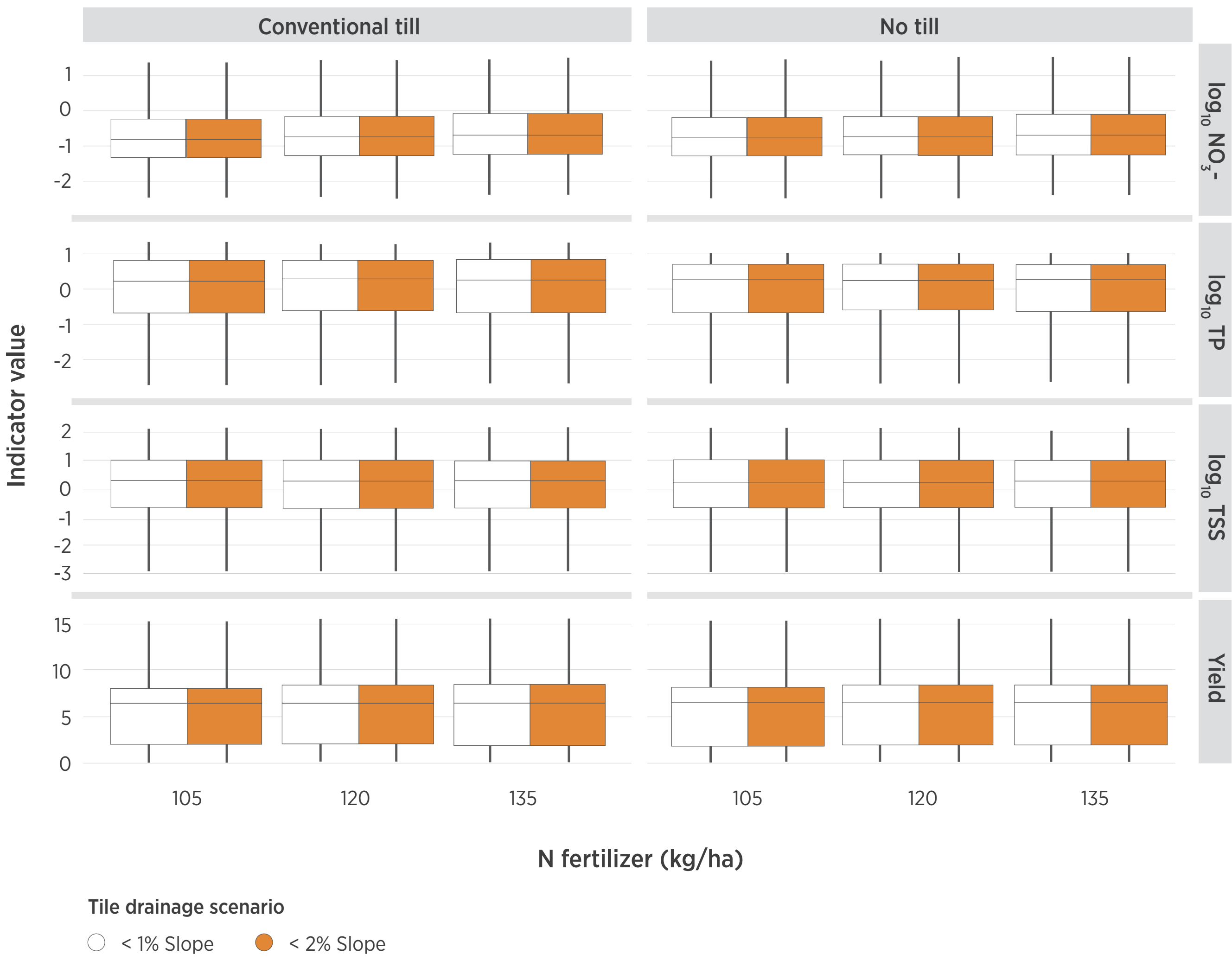


Figure 5.18 | Change in indicators among practices leading to best outcomes for each of four indicators: total suspended sediment (TSS), total phosphorus (TP), nitrate (NO_3^-), and feedstock yield

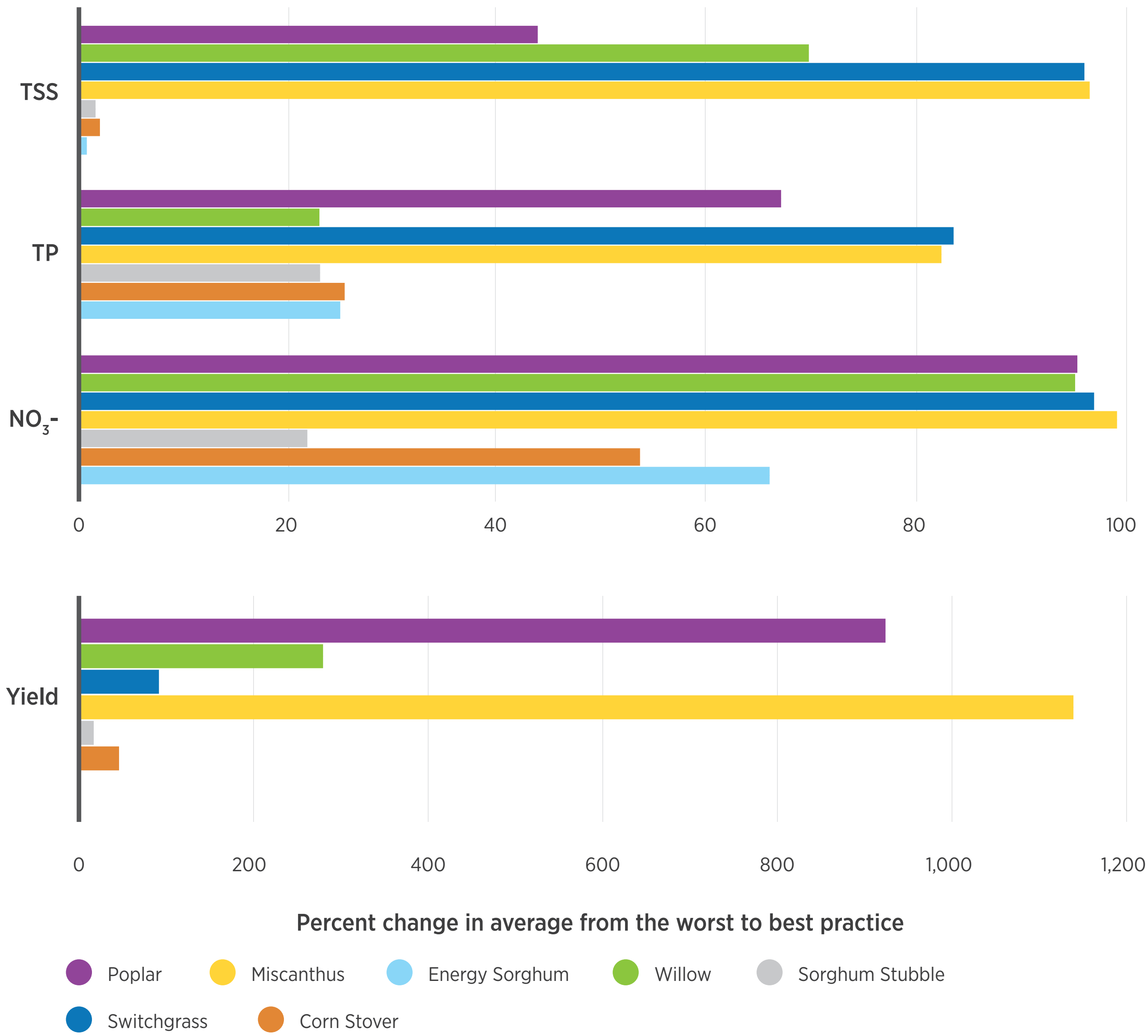


Figure 5.19 | Indicator values for practices leading to best outcomes for each of four indicators: total suspended sediment (TSS), total phosphorus (TP), nitrate (NO_3^-), and feedstock yield. Units for indicators are given in table 5.1.

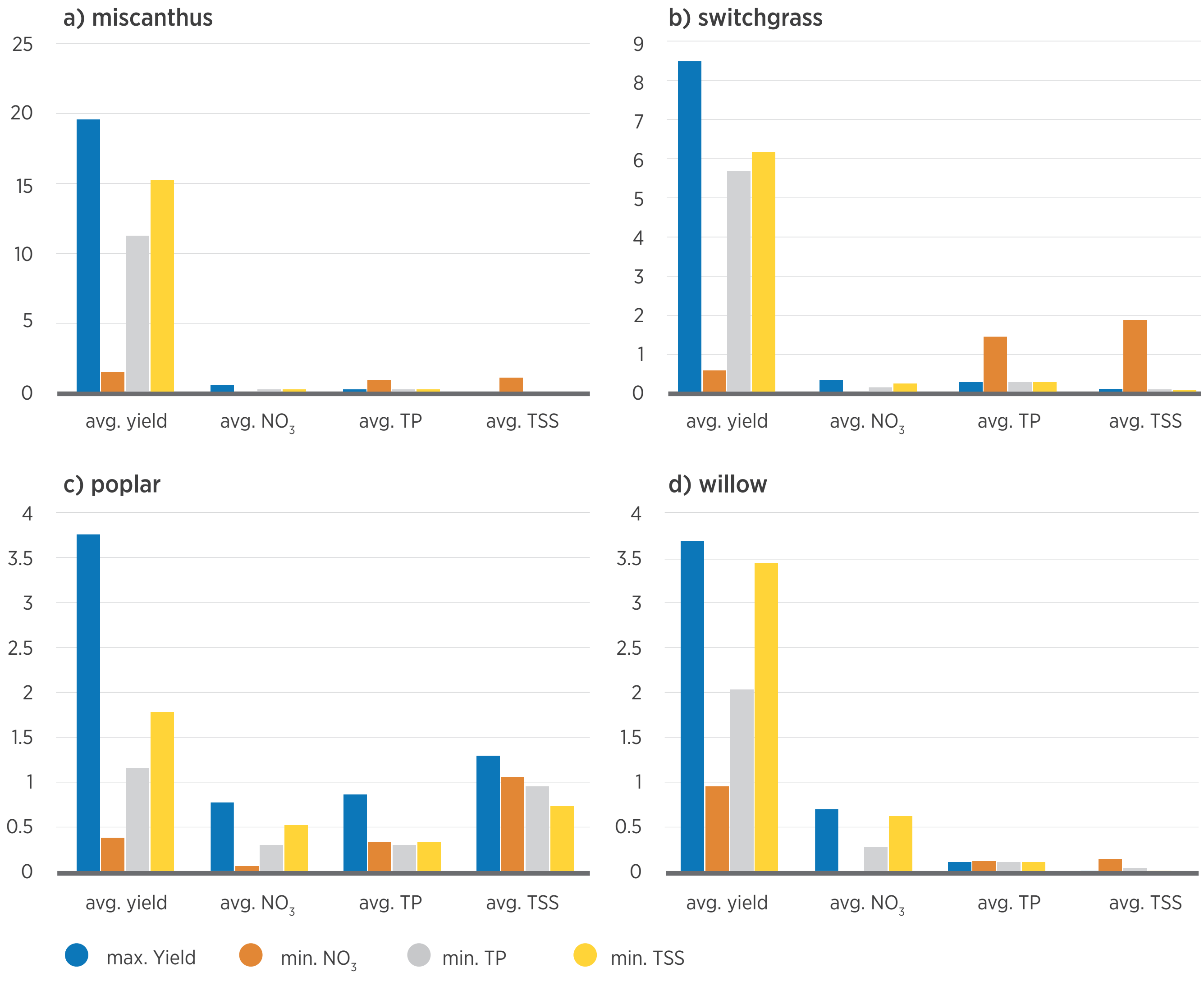
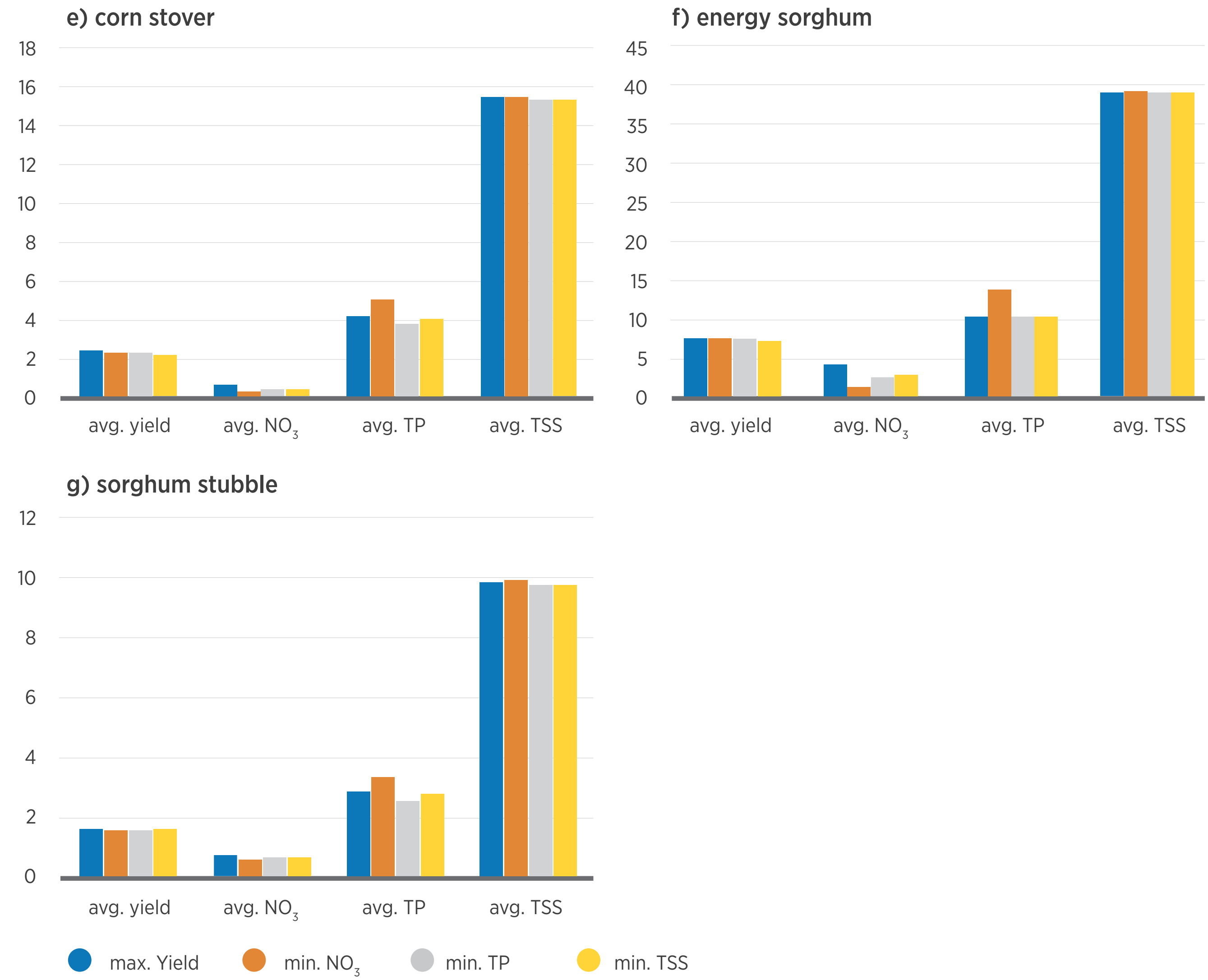


Figure 5.19 | *continued*



For annuals, adding tile drainage only on lands with <1% slope improved all indicators including yield. A tradeoff is evident between nitrate and TP (i.e., the practice leading to minimum nitrate generated high TP), but tradeoffs between yield and TSS were less apparent for residues (fig. 5.19e, g) and energy sorghum (fig. 5.19f). As expected, tillage generally produced lower TSS and TP for all annual crops. Whereas energy sorghum yield was consistently higher under no-till (fig. 5.12), the annuals produced higher whole-crop yields under conventional tillage in some sub-basins (fig. 5.14, 5.16).

As expected, yields were generally higher for the highest fertilizer application for all annuals. A positive yield response to fertilizer was most evident for energy sorghum (fig. 5.13). Therefore, it is unclear why lower fertilizer levels produced lower TSS loadings (and to a lesser extent TP) from lands growing annual crops under no-till management (fig. 5.12, 5.14, and 5.16). This differs from the pattern observed in simulations for perennial crops and illustrated by the conceptual diagram in figure 5.7.

Simulated nutrient and sediment loadings (fig. 5.19e, g) are attributed to the whole crop and not just the effect of residue removal. To attribute indicator values to residue removal, we subtracted nutrient and sediment loadings for simulations without residue removed from the values shown in figure 5.19. Interestingly, residue removal decreased nitrate and TP but increased TSS. This is because we simulated fixed fertilizer inputs rather than restoring the amount removed in residue. For example, harvest of sorghum stubble decreased average nitrate loadings by 15% to 20% and average TP loadings by 6% to 16%. However, harvesting residues increased average TSS by around 13%. For corn stover, decreases in nutrient loadings were quite variable (nitrate ranged from 13% to 40%; TP from 0.16% to 10%), whereas increases in TSS were similar among sub-basins (between 10% and 12.5%). The practice associated with the highest yield also produced the lowest (i.e., largest negative) "residual" contribution to nutrient loadings. Presumably, this is because more residue was harvested (i.e., a constant 80% percentage of yield).

5.4.2 Iowa River Basin

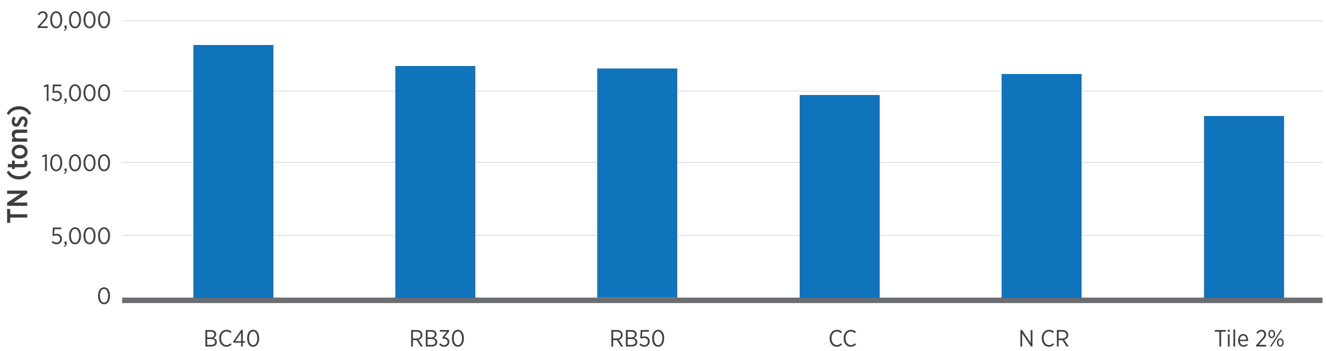
In the IRB study, instead of simulating responses of individual conservation practices we compared scenarios where a combination of conservation practices are implemented under the BC1 2040 scenario (table 5.3). The nutrient (nitrogen and phosphorus) and sediment loadings simulated at the IRB outlet and sub-basins are discussed in this section.

Implementing the conservation practices evaluated in this study—riparian buffer, controlled release, slow-release nitrogen fertilizer, and tile-drain control—substantially reduced watershed nitrogen loading (fig. 5.20). The reduction in total nitrogen because of these practices (compared with that from the baseline BC1 2040 scenario) ranged from 8% to 28%. Nitrate decreased from 6% to 29%. Tile-drain control and cover crops appeared most effective at reducing nitrate, at 28.6% and 19% respectively, followed by slow-release fertilizer (11.4%) (table 5.4).

Several conservation practices resulted in a significant reduction of phosphorus and sediments (fig. 5.21). Most noticeably, suspended sediments in the surface stream decreased by 70%, or 466,000 metric tons, when riparian buffers were installed in the main stem of the Iowa River. Cover crop ranked second with 37% reduction. These values are consistent with literature (Fischer and Fischenich 2000). For scenarios with a cover crop grown after stover is harvested, phosphorus loadings decreased by 27% (fig. 5.21) in the outlet of the basin.

Sensitivity of the annual crops to the nitrogen fertilizer-input rate in this region has been reported elsewhere (Demissie, Yan, and Wu 2012). Nitrate is the main component (>90%) of total nitrogen in this region. Consequently, a decrease in nitrate leads to a comparable reduction in total nitrogen in the watershed.

Figure 5.20 | Export of total nitrogen (TN) and nitrate (NO_3^-) loadings at the outlet of the IRB under various conservation practice scenarios for the BC1 2040 scenario



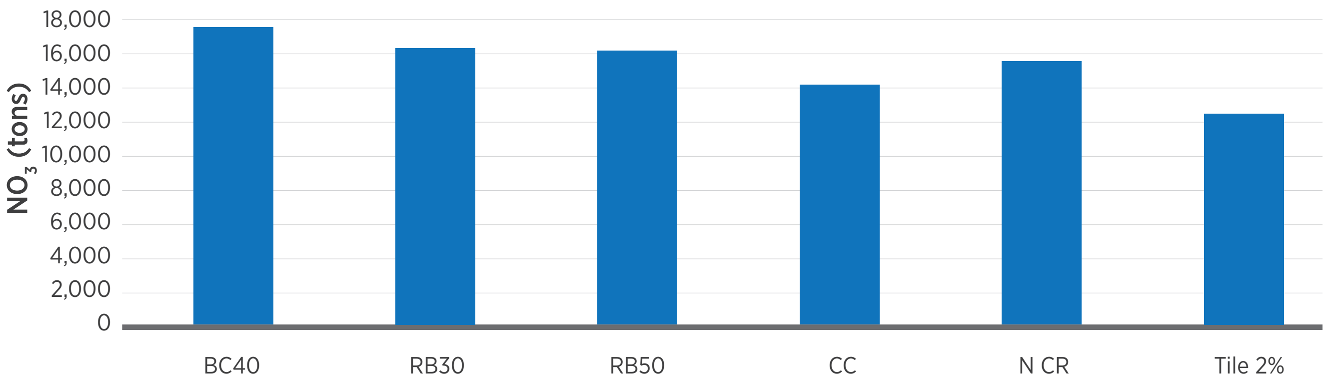


Table 5.4 | Comparison of Suspended Sediments, Phosphorus, and Nitrogen Removal under Conservation Practice Scenarios in the IRB

	Removals Relative to BC1 2040 Scenario (%)			
Conservation Practice Scenarios	**Suspended sediment**	**Total phosphorus**	**Total nitrogen**	**Nitrate**
RB30, main stem Iowa River	70.5%	7.9%	8.2%	6.2%
RB50, main stem Iowa River	70.8%	8.6%	8.9%	6.9%
RB50, entire Iowa River stream network	80.3%	22.7%	12.9 %	10.8%
CC	37.0%	27.4%	18.5%	19.0%
N CR	5.6%	9.9%	10.9%	11.4%
Tile2%	1.8%	1.7%	27.5%	28.6%

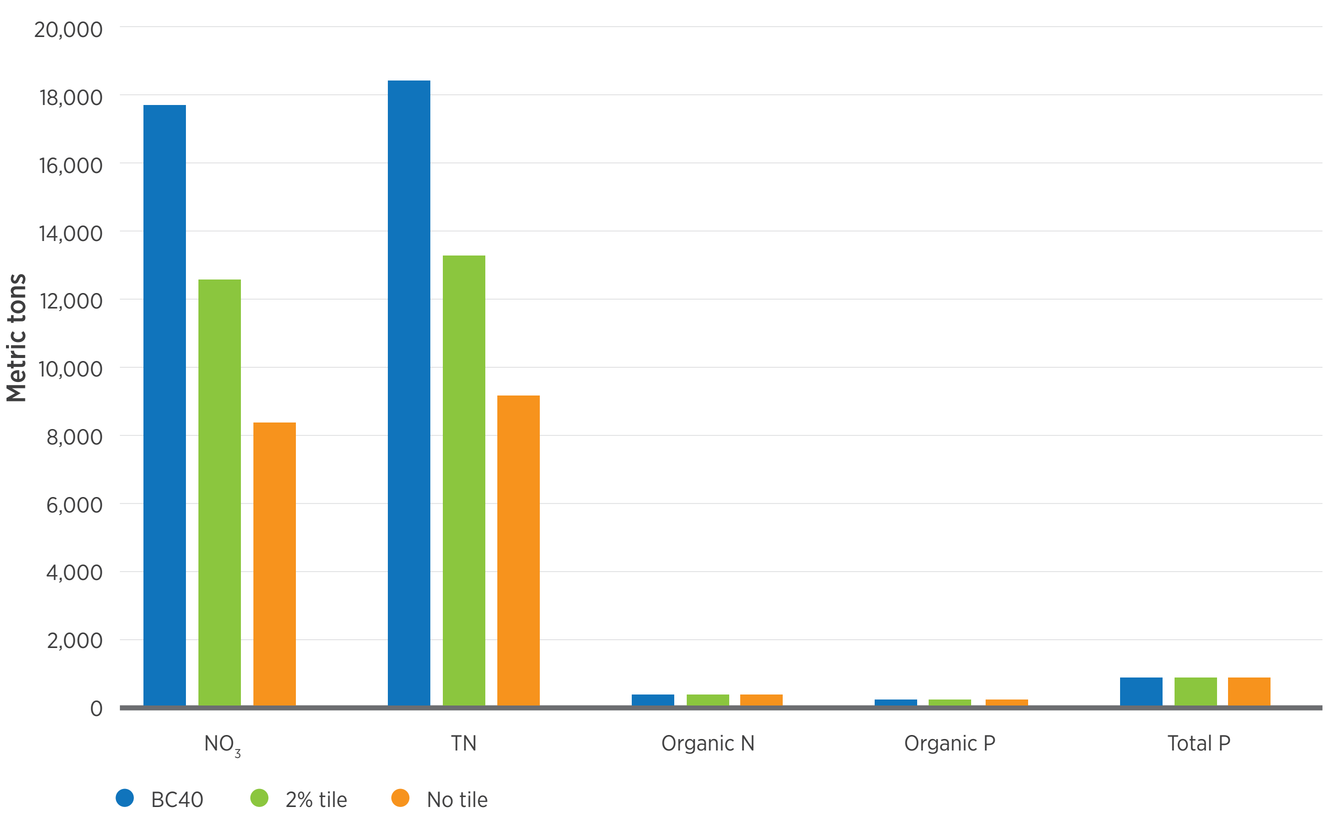
5.4.2.1 Tile Drains

The Iowa agriculture region is one of the most intensively tile-drained regions in the United States. Nitrate is water-soluble and is often transported with water through soil. Drainage tiles create pipelines to carry nitrate from crop root zones to the surface water by short-circuiting the natural flow and, thus, speeding up conveyance of nitrate from the landscape to surface streams (Dinnes et al. 2002).

Our results show that plugging a fraction of the tile drains could result in substantial nitrogen reductions (28% to 29%) in the surface water in the IRB study area (fig. 5.21). In the Tile2% case, most areas simulated without tile drains occurred in the lower portion of the study area. The resulting reduction in nitrogen of 5,000 metric tons is significant for downstream communities. Unlike nitrate, our results suggest that tile drains are not a major pathway for the loss of organic nitrogen, organic phosphorus, or total phosphorus in this basin (fig. 5.21). This is because phosphorus and organic nitrogen are far less mobile in soils than nitrate. These results corroborate previous findings (Dinnes et al. 2002; Brouder et al. 2005).

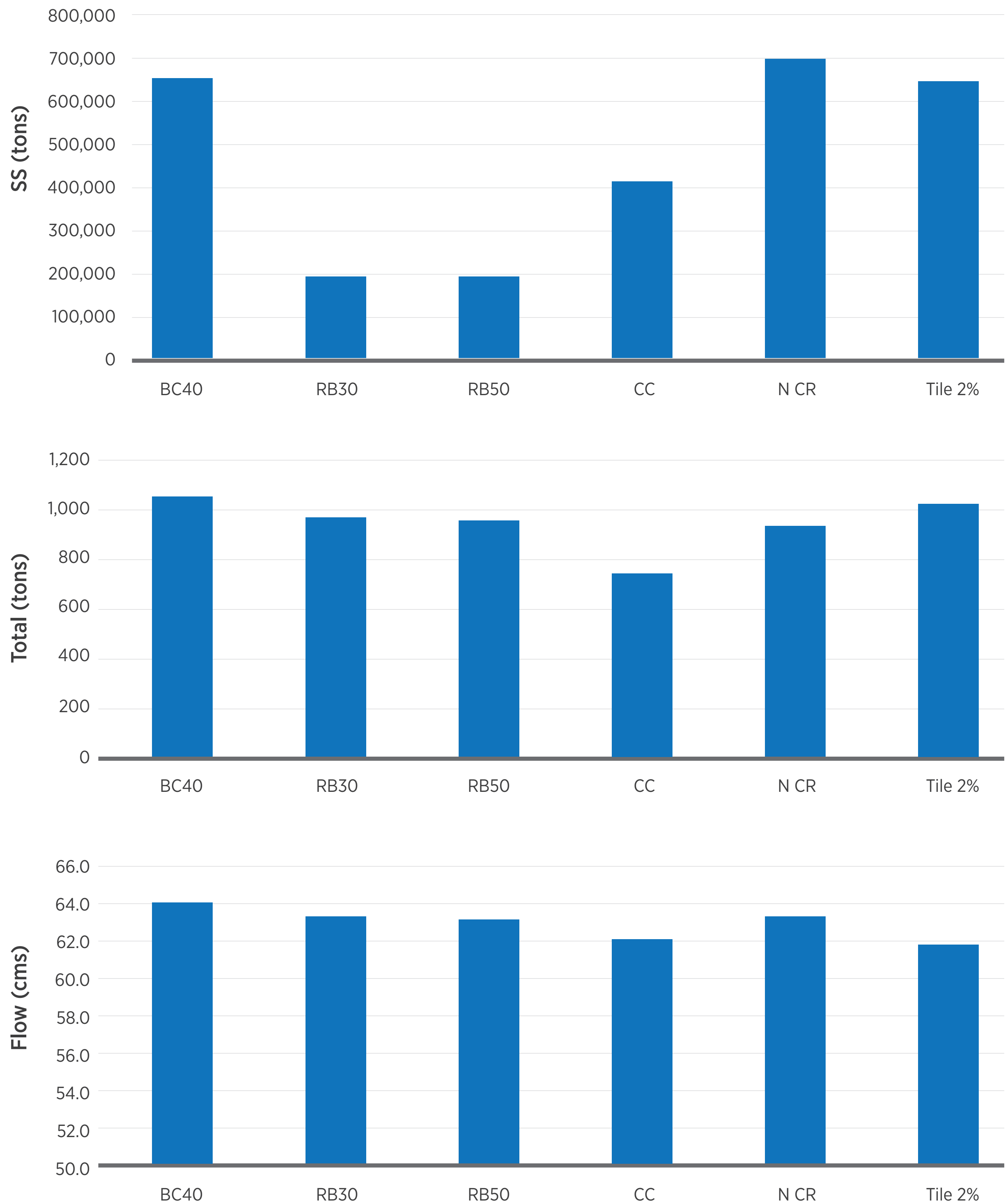
Sediments and phosphorus were not responsive to tile drain control. Reasons for the reduction of phosphorus under slow-release nitrogen fertilizer are unclear. Compared with nutrients and sediments, the impact of conservation practices on water yield or water

Figure 5.21 | Nutrient losses from tile drains at the outlet of the IRB for the BC1 2040 production scenario with tile drains on all lands, on lands with <2% slope, and no lands



Acronyms: N – nitrogen; P – phosphorus.

Figure 5.22 | Loadings of total phosphorus, sediment, and flow under various conservation practices for the BC1 2040 economic future scenario



Acronyms: N – nitrogen; P – phosphorus.

flow in the watershed was minimal (fig. 5.21). There was a slight decrease of flow (3.5%) because of tile-drainage control. This decrease could be caused by a diversion of the flow path—from direct transport via tile to seeping through soil naturally at a slower rate. Thus, it takes longer to reach the surface stream. When a cover crop is in place in a humid region, soil moisture would be expected to increase; thus, the surface runoff decreases.

5.4.2.2 Riparian Buffers

Riparian buffers have long been recognized as one of the most effective measures to trap sediments and reduce runoff. We simulated herbaceous riparian buffers along the main stem of the Iowa River bank adjacent to water. In HRUs with riparian buffers planted in miscanthus or willow, simulated suspended sediments were lower (fig. 5.21). The level of nitrogen removal by the buffer is likely affected by the buffer coverage in watersheds (Fischer and Fischenich 2000). The main stem of the Iowa River in the watershed boundary constitutes 125 km (77.6 miles) of stream, which is 13.7% of the total stream length in the IRB. In this study, the land area covered by riparian buffers (BC30 and BC50) totaled 19,202 acres and 30,591 acres and account for only 0.96% and 1.54% of the entire IRB area, respectively. The area planted in riparian buffer in the watershed is currently 502 acres, about 0.02% of the total IRB, of which most are lands enrolled in the USDA Conservation Reserve Program (Hubbert 2016, personal communication).[1] Excluding existing riparian buffers, the simulated buffers would still be 0.94% and 1.51% of the total IRB area. If the buffer were applied to the entire stream network in the IRB, which would increase coverage to 11.3% (RB50) of the total land, total nitrogen removal would increase substantially. We estimated that up to 22.7% total nitrogen (2,370 metric tons) and 10.8% nitrate (1,900 metric tons) can be avoided in the surface stream (table 5.4). Similarly, 80% of the sediment loadings can be removed (table 5.4), translating to a decrease in transport of sediment of 529,800 metric tons to the downstream Mississippi River. Removal efficiency can further increase if a mixture of trees and grasses is installed (Dosskey, Schultz, and Isenhart 1997).

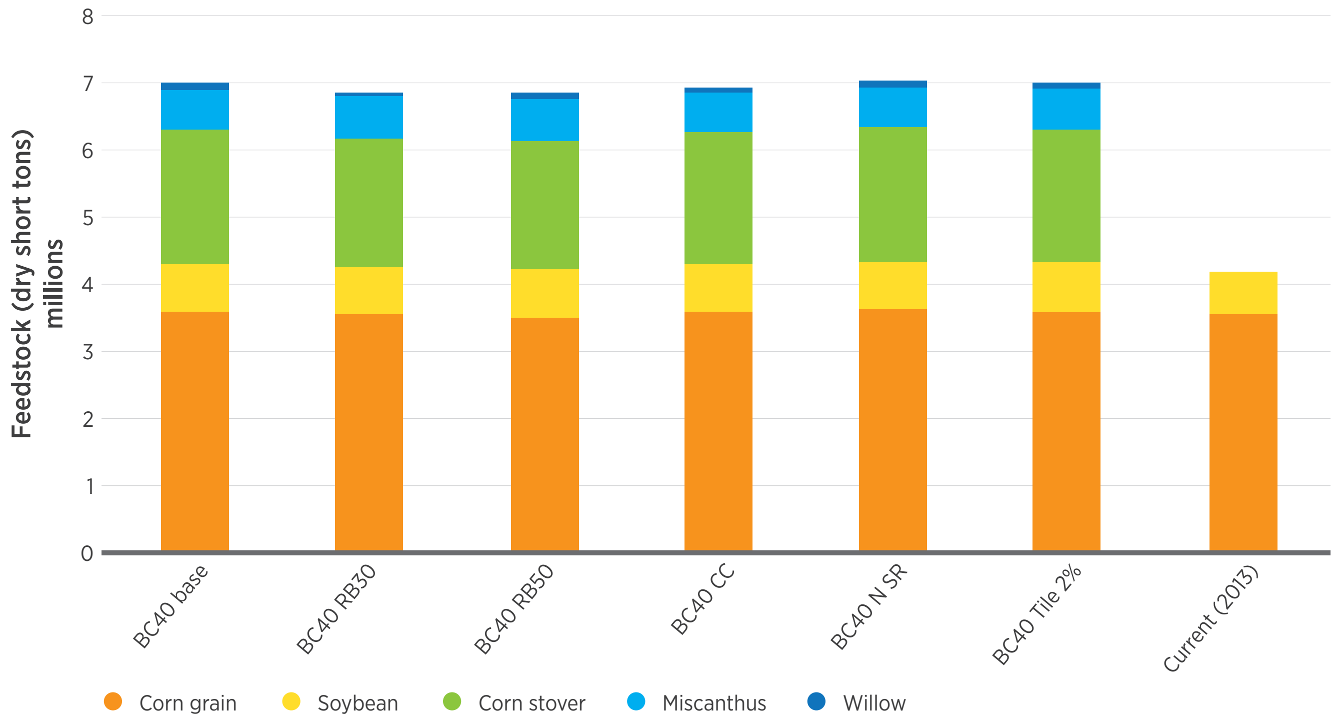
For scenarios with a cover crop grown after stover is harvested, phosphorus loadings decreased by 27% (fig. 5.22) in the outlet of the basin. Riparian buffer, again, can reduce phosphorus loadings by 7.9% (RB30) and 8.6% (RB50). Sediment and phosphorus loadings were not responsive to tile drain control. Reasons for the reduction of phosphorus under slow-release nitrogen fertilizer are unclear. Compared with nutrients and sediments, the impact of conservation practices on water yield or water flow in the watershed was minimal (fig. 5.22). There was a slight decrease of flow (3.5%) because of tile-drainage control. This decrease could be caused by a diversion of the flow path—from direct transport via tile to seeping through soil naturally at a slower rate. Thus, it takes longer to reach the surface stream. When a cover crop is in place in a humid region, soil moisture would be expected to increase; thus, the surface runoff decreases.

5.4.2.3 Biomass Production

The BC1 2040 scenario estimated that the feedstock production in the IRB from corn stover, willow, and miscanthus would total 2.68 million dt. Including corn grain and soybeans, total crop production (including all end uses) in the IRB would be 6.97 million dt (fig. 5.23). We found that implementing conservation practices would have minimal impacts on corn and soybean production under the future BC1 2040 scenario. Results indicate that annual production of corn grain would vary from -2.1% to 1.2% depending on the conservation practice, and soybean

[1] Hubbert, J. 2016. USDA-NRCS record of riparian buffer installation in the Iowa River watersheds. Personal communication between Hubbert, J. and Ha, M. May 26, 2016.

Figure 5.23 | Annual feedstock production in BC1 2040 base case and in BC1 2040 with various conservation practices in the IRB



production would vary from -2% to 0% compared to the BC1 2040 base scenario. The BC1 2040 scenario produces 1% more corn and 10% more soybean compared with a 2013 reference.

In this study, we assumed that the riparian buffer and cover crop were not harvested for biofuel production. However, with care to protect the adjacent stream, both could potentially provide feedstock. By a rough estimate, if 50% of the switchgrass grown on riparian buffer were harvested, an additional 73,000 and 121,000 dt of biomass could be obtained from RB30 and RB50, respectively. In addition, if 40% of the cover crop were harvested, an additional 351,000 dt of biomass could be obtained from rye, a 12.8% increase from the BC1 2040 base scenario. By harvesting rye and switchgrass from a 50-m riparian buffer, the cellulosic biomass production could potentially increase by 16%.

5.4.2.4 Regional Distribution of Cost and Benefits

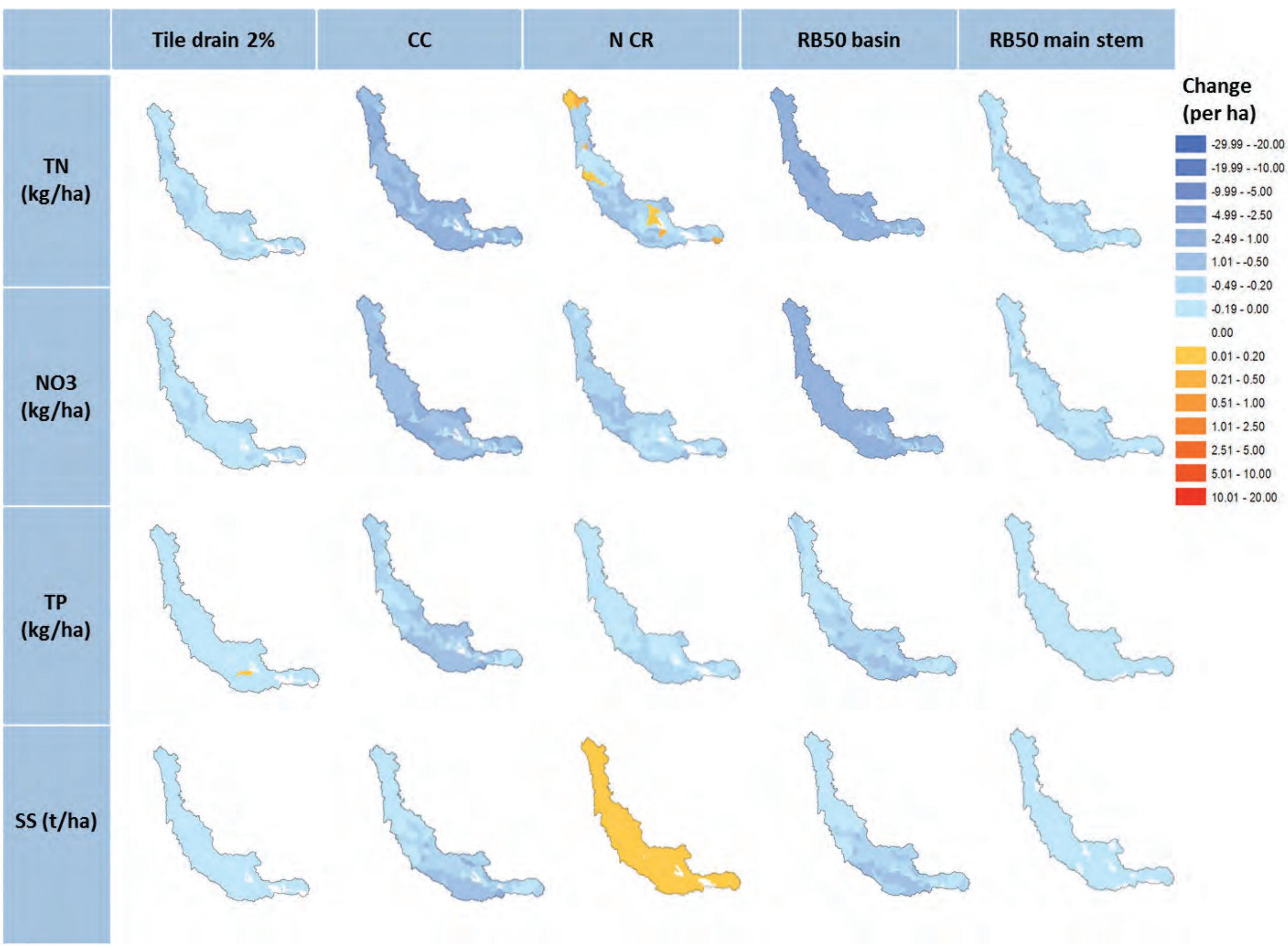
At the sub-basin level, loadings of nutrients and sediments exhibit strong heterogeneity across the landscape (fig. 5.24). As expected, riparian buffer scenario for the entire IRB stream network resulted in nitrogen reductions across the watershed. In the cover crop scenario, reduction of nitrogen loadings appears aggregated because the basin is predominantly planted with corn/soybean rotation system (66.9%; figure 5.6) and residue is harvested from most cornfields. Similarly, we observed a reduction of nitrogen by using slow-release fertilizer in corn HRUs. The largest nitrogen reduction occurred in the middle of the basin where annual crops were grown in highest acreages.

These results suggest that basin-wide effective nitrogen, phosphorous, and sediments removal could be achieved by installing a buffer in the riparian zone in the IRB, combined with planting a cover crop and using slow-release nitrogen for acreage planted in corn. Geographically, reductions in phosphorus and sediments occurred consistently in the lower sub-basins of the IRB, which has a larger flow and a denser stream network than upstream.

5.4.2.5 Uncertainties

Several factors contribute to uncertainties in this analysis. The simulation is based on historical 20 year climate data. Future potential climate change and its regional impacts for 2040 were not available at the time of this study. Climate issues and potential effects on biomass production are discussed in chapter 13. Riparian buffers can effectively trap sediments and nutrients; however, the scale of buffer implementation in the watershed would depend on land use and other economic considerations, as well. At present, most riparian buffers occur along streams on Conservation Reserve Program land. Increases in riparian buffers could affect the amount of land available for production. On the other hand, the use of riparian buffers could create a land-use change from annual to perennial cropping systems. Thus, a systems

Figure 5.24 | Changes of nitrogen, phosphorus, and suspended sediment under evaluated conservation practices relative to BC1 2040 scenario across the IRB



Acronyms: CC – cover crop scenario added; NCR – controlled release of nitrogen fertilizer added; RB30 – 30-m riparian buffer added; RB50 – 50-m Riparian Buffer Added.

approach in land management, conservation, and feedstock production, with careful planning under a multiagency joint effort, will be a critical step toward water quality improvement.

Finally, this chapter addresses region-specific issues in two regions, the AWR and IRB, by evaluating conservation practices that are suitable for region-specific feedstocks at the production scale estimated by *BT16* volume 1. The differences in the choice of conservation practices selected are largely due to distinct regional environments and feedstock requirements. Due to regional heterogeneity, results may not be applicable to other regions. Nevertheless, this study provides valuable information for regions with similar characteristics.

5.5 Summary

In this chapter, we asked the question, "How can we manage future biomass production to protect water quality with minimal cost to feedstock supply?" We identified two tributary river basins of the Mississippi River Basin with contrasting future biomass feedstock profiles under the BC1 2040 scenario, each set against a different agricultural backdrop. Our analysis of these two regions identified the swing potential of different management practices.

For the modeled scenario, we found complementarities between simulated potential biomass yield, TSS and TP in the AWR basin, and tradeoffs between biomass yield and nitrate for perennial grasses and SRWCs. Higher fertilizer levels produced higher yields and lower TSS and TP, but higher nitrate. We note that if we had simulated even higher levels of fertilizer, we would have reached a point beyond which no improvement was observed in yield (and thus, TSS and TP). Thus, the challenge is to avoid nitrate runoff by using conservation practices such as filter strips, as we demonstrated for SRWCs.

In addition, our analysis revealed water quality benefits of coppiced willow, which minimized tradeoffs between nutrient and sediment reduction and biomass yield in the scenario. Filter strips also provided water quality benefits for both SRWC crops.

Among annual crops and residues in the AWR, implementing tile drainage only on the flattest lands produced the best outcomes in terms of productivity, as well as water quality. No-till reduced sediment loadings, but in some cases, it came with a small cost to productivity. Fertilizer practices did not produce much variation in any of the indicators in this analysis. When subtracting the effect of the annual crop without residue removal (corn and sorghum grain), we observed sizable percentage decreases in nutrient loadings and increases in sediment loadings. This is likely because we simulated variable residue removal (and associated nutrients), but applied the same amount of fertilizer. In other words, harvested nutrients were not specifically replenished.

For the AWR, a visualization tool allows users to explore simulated data. By selecting thresholds for each of the water quantity and quality indicators, users can evaluate (1) the 'sustainable' supply (thus defined) and (2) the set of conservation management practices that, according to SWAT simulations, lead to user-defined sustainable production. The visualization shows the relative benefits of different practices for each crop. The visualization can be found here: www.bioenergykdf.net.

In the IRB, we demonstrated the benefits of four conservation practices (riparian buffer, cover crop, slow-release nitrogen fertilizer, and tile-drain control) in the annual crop corn/soy dominant flat terrain. These practices could effectively reduce nitrogen up to 29%, phosphorus 27%, and suspended sediments 80%. Riparian buffer implementation on the entire IRB stream network could lead to the highest reduction of suspended sediments and phosphorous loadings in the watershed while partial control of tile drainage could bring the most benefits to nitrogen reduction in the practices evaluated. Reductions of sediments and phosphorus in IRB under the conservation practices were consistently concentrated in the middle and lower portions of the river basin while that of nitrogen could be extended to the entire IRB.

This study suggests that basin-wide effective nutrient removal and sediment reduction in the biomass development could be achieved by implementing a combination of the practices - installing a buffer in the riparian zone, control tile drainage and using slow-release nitrogen fertilizer in the crop growing area, and planting a cover crop in the area stover is harvested. If the effects of the four practices were additive, by adopting tile drain control (Tile 2%) and cover crop, together nitrogen could be reduced by nearly 50% and sediments reduced by more than a third compare to BC1 2040 scenario. We also highlight the potential benefits, both for production and water quality, of developing protocols for harvesting riparian buffers.

The Gulf of Mexico, which receives nutrient inputs from upstream agriculture in the Mississippi-Atchafalaya River Basin, has a large hypoxic zone that is deadly to aquatic life during summer. The river basins simulated here are both tributaries of the Mississippi River (fig. 5.1). By choosing perennial feedstocks (Jager et al. 2015) and implementing conservation practices (Hu and Wu 2015), we envision a win-win situation in which biomass production helps to reduce downstream nutrient loadings to the Gulf of Mexico. Done right, biomass production can decrease the environmental impacts of conventional crops.

5.6 References

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06 | Water Quality Response to Forest Biomass Utilization

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6.1 Introduction

Forested watersheds provide approximately 80% of freshwater drinking resources in the United States (Fox et al. 2007). The water originating from forested watersheds is typically of high quality when compared to agricultural watersheds, and concentrations of nitrogen and phosphorus are nine times higher, on average, in agricultural watersheds when compared to forested watersheds (Fox et al. 2007). Silvicultural activities typically occur on a low percentage of forested lands in any one year, and effects on water quality from silvicultural operations are typically localized and short-lived (Bethea 1985; Dissmeyer 2000).

The effects of silvicultural activities on water quality have been reviewed on several occasions, and the findings are remarkably consistent. Throughout the United States, silvicultural activities have minimal effects on water quality, and potential effects from harvest operations are largely mitigated by the widespread adoption of best management practices (BMPs) (Binkley and Brown 1993; Fulton and West 2002; Grace III 2005; Stednick 2010; Ice et al. 2010). Silvicultural activities that may compromise water quality are typically nonpoint source and include road construction, ground disturbance from whole-tree skidding, mechanical site-preparation activities, herbicide application, and fertilizer application (Fulton and West 2002).

In this chapter, we briefly review the current effects of silvicultural activities on water quality and then assess the potential effects of increased demand for biomass, based on select scenarios from the 2*016 Billion-Ton Report* (*BT16*), on several water-quality indicators including sediment, nitrate (NO_3^-), and total phosphorus (TP) load. The literature documenting the specific effects of biomass removal from forests on water quality is sparse at best. However, the majority of biomass would be harvested using harvest systems that mimic current silvicultural practices. Therefore, it is reasonable to relate the potential effects of traditional forest-harvest operations to what we might expect from the removal of biomass.

6.1.1 Sediment, Nitrate, and TP

Perhaps the most widespread and deleterious water-quality-related effect of silvicultural operations comes from the displacement of sediment and its transport into stream channels, particularly due to road construction, harvesting, and site preparation (Grace III 2005). The extent of erosion and sediment transport is based on several factors, including the soil texture, organic matter content, slope angle, and application of BMPs (Fulton and West 2002). Sediment impairs aquatic habitats by reducing water and gas exchange between the stream and the groundwater below and adjacent to it. Sediment also fills in pools and covers stream-bed gravels, which are critical to salmonid survival and reproduction (Waters 1995). Harvest operations, including road construction, log skidding, and site preparation, often expose bare soil and increase the risk of erosion. It has been estimated that up to 90% of sediment delivered to streams following forest-harvest operations is road-related (Appelboom et al. 2002; Scoles et al. 1996). However, skidding logs across the soil surface exposes and compacts mineral soil, and may create furrows that channel overland flow (Fulton and West 2002). In addition to road construction and harvesting activities, mechanical site-preparation activities, such as shearing, disking, drum-chopping, and root-raking, cause significant soil disturbance, which can lead to further sediment transport after harvests (Fulton and West 2002). These activities were once common on pine plantations, which are widespread in the southeastern United States (Grace III 2005), but chemical herbicides have increasingly replaced mechanical site-preparation activities as a more economical way to reduce competition. Sedimentation effects from silviculture are typically short-lived, lasting 2–5 years (one example is from Amatya et al. 2006) or until understory vegetation has recovered in disturbed areas.

In healthy, undisturbed forest ecosystems, only a very small fraction of nutrients is lost to surface waters. Nutrient cycles in these systems are typically very tight, with most nutrients being bound and efficiently cycled through vegetation and soils (Bormann and Likens 1994; Scoles et al. 1996). The removal of trees and understory vegetation during harvest activities can cause nutrient transport to streams to occur via leaching and erosion (Scoles et al. 1996). Nitrogen and phosphorus are the primary nutrients that influence ecological processes and productivity in streams and lakes (Fulton and West 2002). Increased loading of nitrogen and phosphorus can cause increased biological activity, increased turbidity, limited light penetration, and increased biological oxygen demand (Fulton and West 2002). The "eutrophication" of surface waters by increased nutrient loading has significant effects on fish and other aquatic organisms.

Most forest-harvesting studies in the United States have demonstrated that stream-water nitrogen concentrations, including NO_3^-, increase after harvest, but stream-water concentrations of NO_3^- rarely exceed the U.S. Environmental Protection Agency's (EPA's) drinking water standard of 10 milligrams per liter (mg/L) (Binkley and Brown 1993). More commonly, nitrate nitrogen (NO_3^--N) increased up to 1 mg/L (Swank 1988; Askew and Williams 1986; Riekirk 1983; Hewlett, Post, and Doss 1984; Miller et al. 1988; Amatya et al. 2006). Within the literature, however, it has been documented that higher levels of stream-water nitrate may occur after harvest in areas prone to high levels of atmospheric nitrogen deposition, particularly in the northeastern United States (Likens et al. 1970; Bormann et al. 1968; Yanai 1998). Phosphorus has not been as thoroughly studied within the context of forest harvest, but several studies describe a significant increase in TP immediately following harvest (Blackburn and Wood 1990; Wynn et al. 2000; Amatya et al. 2006; McBroom et al. 2008). As with sediment, nitrogen and phosphorus typically increase the first year after harvesting but return to pre-harvest levels within 2–4 years following harvest (Shepard 1994; Amatya et al. 2006).

6.1.2 Management Intensity and BMPs

Although road building, road use, and related activities account for the majority of silviculture-related effects on water quality (Grace III 2005), the intensity of management may also determine the extent of effects. Silvicultural methods vary widely in the amount of material harvested and the mechanical disturbance created by harvesting. Single-tree selection or group-selection harvests often remove significantly less biomass than clearcutting. The research findings that relate biomass removal to sediment and nutrient loads (loadings) are straightforward. For example, Beasley and Granillo (1985) demonstrated that selective harvests yielded significantly less sediment than clearcuts. However, in a study comparing four harvesting methods, including selective and clearcutting, Eschner and Larmoyeux (1963) determined that neither the number/mass of trees removed nor the harvesting method utilized was the primary factor influencing water quality; rather, it was skid trail and logging-road design.

Mechanical site preparation after clearcutting has been demonstrated to increase sediment and phosphorus loads, but less evidence supports significant increases in nitrate or total nitrogen loads (Amatya et al. 2006; Muwamba et al. 2015). Shearing, root raking, disking, and windrowing expose bare soil, decrease soil stability, and increase erosion rates. For example, Douglass (1977) determined that the amount of sediment lost from sites that were cleared and disked was twice that from sites that were cleared only. Because phosphorus is often transported along with sediment as particulates, it may increase after site preparation as well. Blackburn and Wood (1990) observed that when shearing was used to remove stumps and windrow debris, phosphate and TP increased significantly compared to treatments in which debris was chopped in place. With use of herbicides replacing mechanical methods for competition control, these effects should be reduced.

Intensive management of pine and Douglas-fir plantations increasingly involves herbicide and fertilizer application. The effect of herbicide application on sediment, nitrogen, and phosphorus in streams and lakes is likely minimal; however, it has been demonstrated that fertilization can temporarily increase ammonium, total nitrogen, ortho-phosphate, and TP in streams draining plantations (Fulton and West 2002; Beltran et al. 2010). The Binkley, Burnham, and Allen (1999) review of forest fertilization concluded that in the absence of BMPs, nitrate and phosphorus levels increased in receiving waters, drinking water standards were not exceeded, and the increase in nutrient levels was short-lived.

It has been widely demonstrated that BMPs are very effective at mitigating the effects of silvicultural operations on water quality (Grace III 2005). The most common and effective BMPs typically involve aspects of road design and utilization of riparian buffers. Because the majority of sediment introduced to stream channels from silvicultural activities is road-related, significant improvements in water quality can be made by employing road-design BMPs (Appelboom et al. 2002). Appelboom et al. (2002) showed that a continuous berm maintained along the edge of a forest road can reduce total sediment loss by an average of 99% compared to the same type of road without the presence of a continuous berm. When a continuous berm is not present, graveling the road surface can reduce the total loss of sediment from roads by an average of 61% compared to a non-graveled road surface. An experiment at the Coweeta Watershed in western North Carolina demonstrated that sediment delivery to a stream channel can be reduced by up to 50% with proper planning and layout of roads and skid trails (Swift 1988). Similarly, Mostaghimi et al. (1999) reported that harvesting and intensive site preparation increased nitrogen and phosphorus loading where BMPs were not applied. Mostaghimi et al. (1999) also reported that use of BMPs mitigated the effects of harvesting and site preparation, while Vowell

(2001) reported that following state BMPs in Florida resulted in no significant increases in stream water nitrogen or phosphorus. An increasing body of evidence shows that silvicultural effects on water quality are relatively small and short-lived (Shepard 1994) when compared to agricultural practices, and proper implementation of BMPs can effectively mitigate most water-quality effects. In a recent survey of BMP implementation, Ice et al. (2010) estimated that BMP compliance in forestry is approaching 90% nationally.

The objective of the analysis that follows is to estimate the effects of forest-biomass removal on surface-water quality (sediment, NO_3^--N, and TP) for select scenarios of *BT16* (described in chapter 2) in the conterminous United States. The analysis focuses on three commonly reported harvest types: thinning operations, clearcuts with natural regeneration, and clearcuts with site preparation and planting (plantations). Water-quality estimates for the potential biomass supply are produced at the county level and aggregated to three regions having relatively unique climates and vegetation (DOE 2016).

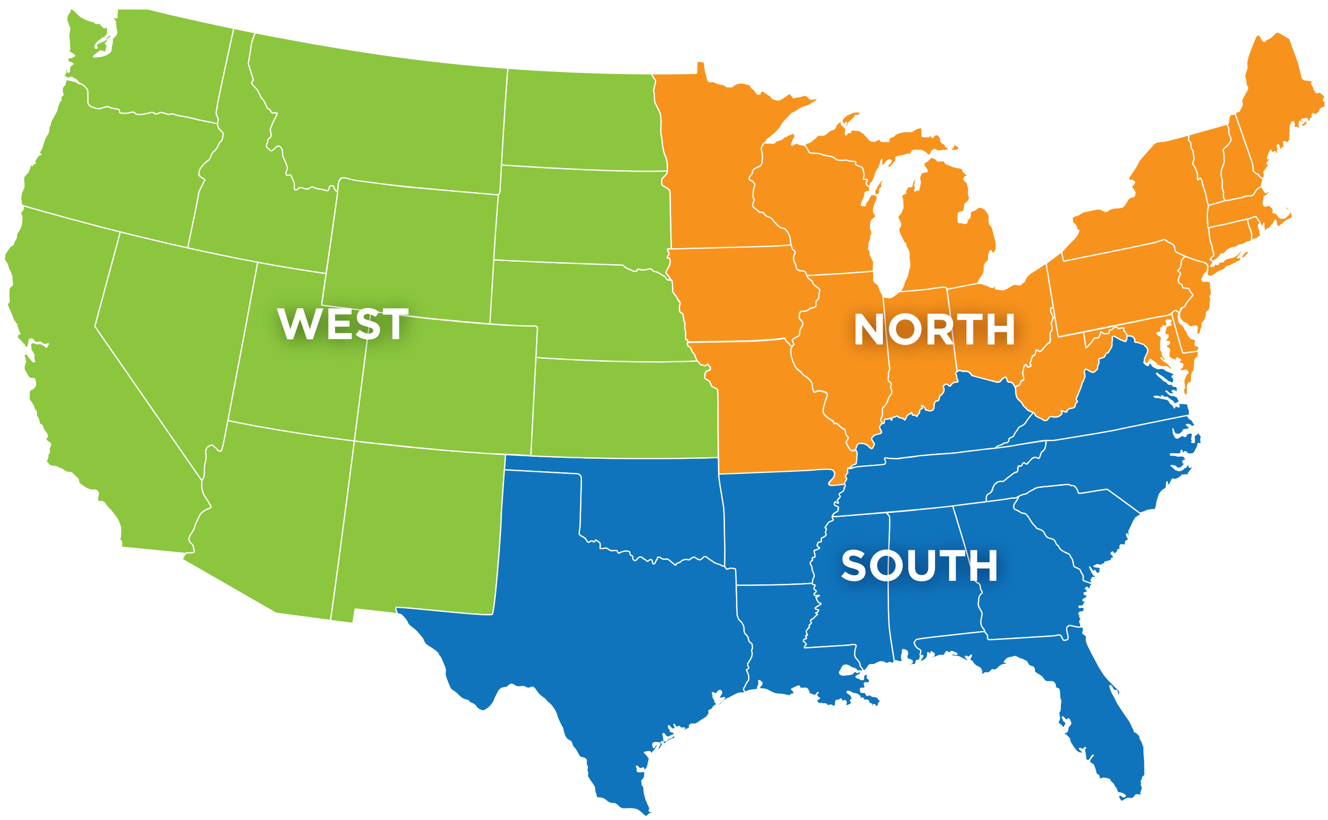
6.2 Methods

To assess the possible effects of potential forest-biomass removal on water quality, we searched the peer-reviewed literature for studies that either directly reported effects on water quality from biomass removal or reported effects on water quality from traditional silvicultural operations. Within each paper, data were extracted detailing the forest vegetation type, stand age, basal area, geographic location, climate, topography, soil, harvest operations, the mass of material removed, pre-harvest water-quality parameters, and changes in water quality following silvicultural operations. The initial goal was to develop a series of regionally specific models that would relate potential mass of biomass removed to changes in water quality; then, we could apply those models to the output derived from the Forest Sustainable and Economic Analysis Model (ForSEAM) for select biomass-removal scenarios proposed in the years 2017 and 2040 (DOE 2016). However, developing significant, predictable relationships between biomass removal and water quality that could be applied regionally proved impossible given the lack of detailed information and relatively small number of studies available in the literature. Since it was not possible to develop a full suite of harvest-type and region-specific models relating biomass removal to water quality, we adopted an approach that relates the acres harvested to changes in water quality, and we developed regional or harvest-type-specific models when possible. All other models are general for the conterminous United States.

6.2.1 Scope of Assessment

The scope of this assessment covers the incremental effects of biomass harvest activities on water quality for select scenarios described in *BT16*, volume 1 (DOE 2016). The scenarios include the baseline moderate housing–low wood energy demand (ML) scenario in 2017 and 2040, and an alternative high housing–high wood energy demand (HH) scenario in 2040. The scenarios and assumptions are described in chapter 2. For this assessment, results from ForSEAM are analyzed at the county level and then aggregated to three regions (North, South, and West) of the conterminous United States (fig. 6.1).

Figure 6.1 | Map showing states in the northern, southern, and western regions of the United States. Separate forest water quality analyses were undertaken for these regions.



6.2.2 Description of Water Quality Response Modeling

We searched peer-reviewed literature and identified 38 papers containing quantitative data describing the effects of forest harvest on water quality (table 6.1). Studies were separated into three categories for analysis: thinning operations; clearcuts with natural regeneration; and plantations where extensive site preparation, fertilization, and herbicide applications were used in conjunction with replanting trees. Table 6.2 details the silvicultural activities common to each harvest type. Sediment, NO_3^--N, and TP were the three water-quality parameters selected for this assessment. When pre-harvest or control data were available, they were considered the reference condition. All data recorded after harvest treatments were considered the response to harvest. Units for reference and response conditions were expressed as kilograms of response variable delivered to a water body per hectare per measurement year (kg/ha/year). A generalized, linear mixed-effects model (Proc GLIMMIX, SAS 9.4, SAS Institute, Cary, North Carolina) was used to determine harvest type and regional differences in NO_3^--N, TP, and sediment response to biomass removal. Because not all studies reported data for the same number of years post-harvest, only the initial response year was used for statistical comparisons.

Table 6.1 | Peer-Reviewed Publications Used To Extract Water-Quality Parameters

Citation	Region	State	NO_3^--N	TP	Sediment
1. Bormann et al. (1968)	North	NH	•		
2. Bormann et al. (1974)	North	NH	•		
3. Briggs et al. (2000)	North	ME	•		
4. Hornbeck et al. (1987)	North	NH	•		
5. Hornbeck et al. (1990)	North	NH,ME,CT	•		•
6. Likens et al. (1970)	North	NH	•		
7. Martin and Hornbeck (1994)	North	NH			•
8. Wang et al. (2006)	North	NY	•		
9. Yanai (1998)	North	NH	•	•	
10. Amatya et al. (2006)	South	NC	•		•
11. Amatya and Skaggs (2008)	South	NC	•		•
12. Arthur, Coltharp, and Brown (1998)	South	KY			•
13. Aubertin and Patric (1974)	South	VA			•
14. Beasley (1979)	South	MS			•
15. Beasley and Granillo (1988)	South	AR			•
16. Beasley, Granillo, and Zillmer (1986)	South	AR			•
17. Blackburn, Wood, and Dehaven (1986)	South	TX			•
18. Blackburn and Wood (1990)	South	TX	•	•	
19. Chang, Roth, and Hunt (1982)	South	TX			•
20. Fox, Burger, and Kreh (1986)	South	VA	•		
21. Grace III (2004)	South	AL			•
22. Grace III and Carter (2000)	South	AL			•
23. Grace III and Carter (2001)	South	AL			•
24. Grace III, Skaggs, and Chescheir (2006)	South	NC	•	•	
25. McBroom, Chang, and Sayok (2002)	South	TX	•		•

Citation	Region	State	NO_3^--N	TP	Sediment
26. McBroom et al. (2008)	South	TX	•	•	•
27. Miller (1984)	South	OK			•
28. Muwamba et al. (2015)	South	NC			
29. Sanders and McBroom (2013)	South	TX			•
30. Swank, Vose, and Elliott (2001)	South	NC	•		•
31. Van Lear et al. (1985)	South	SC	•		•
32. Wynn et al. (2000)	South	VA		•	•
33. Brown and Krigier (1971)	West	OR			•
34. Gravelle et al. (2009)	West	ID	•		
35. Heede and King (1990)	West	AZ			•
36. Karwan, Gravelle, and Hubbart (2007)	West	ID			•
37. Martin and Harr (1988)	West	OR	•		
38. Tiedemann, Quigley, and Anderson (1988)	West	OR	•		

The length of time required for water quality in the treated units to return to pre-harvest levels, or levels similar to controls, was defined as the response period. In most cases, the experiments were not of sufficient length to capture the full response period as many studies only reported 1–3 years of post-harvest data. Post-harvest measurement periods ranged from 1 to 13 years in the literature searched (table 6.1). The total loading of sediment, NO_3^--N, or TP delivered to a water body over the response period in excess of the reference condition was defined as the response load. To characterize the response load for each water-quality variable, the mean and 90% confidence intervals were calculated for each harvest type, region, and year post-harvest, where appropriate, as indicated by the results of the mixed model. The mean response load and confidence intervals for each year (kilograms/hectare [kg/ha]) were plotted against the year after harvest, and a curve was fit to each data set. The resultant family of response curves was best represented by an exponential function of the form:

Equation 6.1:

$$y = a^{-b*x}$$

In this equation, y is the water quality response (kg/ha), a is a constant representing the y-intercept, b is the exponential decay rate, and x represents the year after harvest. Solving each equation for x, where the response curve intersects the pre-harvest condition, gives the modeled response period for each variable. Integrating each curve on the interval from 0 to the end of the modeled response period generates the total modeled response after harvest in kg/ha. The modeled response to harvest could then be applied to the biomass output from ForSEAM. First, the number of hectares where whole-tree harvests for biomass occurred was summed within each county by harvest type for each scenario and year. Next, the appropriate response load was applied to each harvest type, and the total water-quality response load for each county was calculated as the sum of all harvested acres (kg). Finally, the regional water-quality response to biomass harvest was calculated as the sum of all county-level response loads within each region and expressed in gigagrams (Gg).

Table 6.2 | Common Silvicultural Operations Conducted during Three Different Harvest Types

Harvest type	Road building/ improvement	BMPs	Log skidding	Residue removal	Mechanical site preparation	Herbicide	Fertilizer
Thin	•	•	•	•			
Clearcut with natural regeneration	•	•	•	•			
Plantation clearcut	•	•	•	•	•	•	•

For comparative purposes, reference estimates of sediment, NO_3^--N, and TP load were also produced using pre-harvest conditions for each region. We applied the pre-harvest water quality values to all forested acres within a county based on data from the National Land Cover Database and the U. S. Forest Service's Forest Inventory and Analysis (FIA) data, and then calculated the sum of all water-quality values delivered to a water body within each geographic region. This load is referred to as the regional reference load. Similarly, we applied the pre-harvest water quality values to only the harvested acres within a county. This load is referred to as the pre-harvest reference load.

6.3 Results

Results from the mixed model comparing regional and harvest-type differences indicate that sediment load was consistent across regions, but sediment load was significantly ($\alpha = 0.05$) greater from plantations when compared to naturally re-generated stands ($P < 0.05$). Conversely, NO_3^--N loads (loadings) were greater in the North than in any other region ($P < 0.05$), and there were no significant ($\alpha = 0.05$) differences between harvest types. There were no significant region or harvest-type effects for TP.

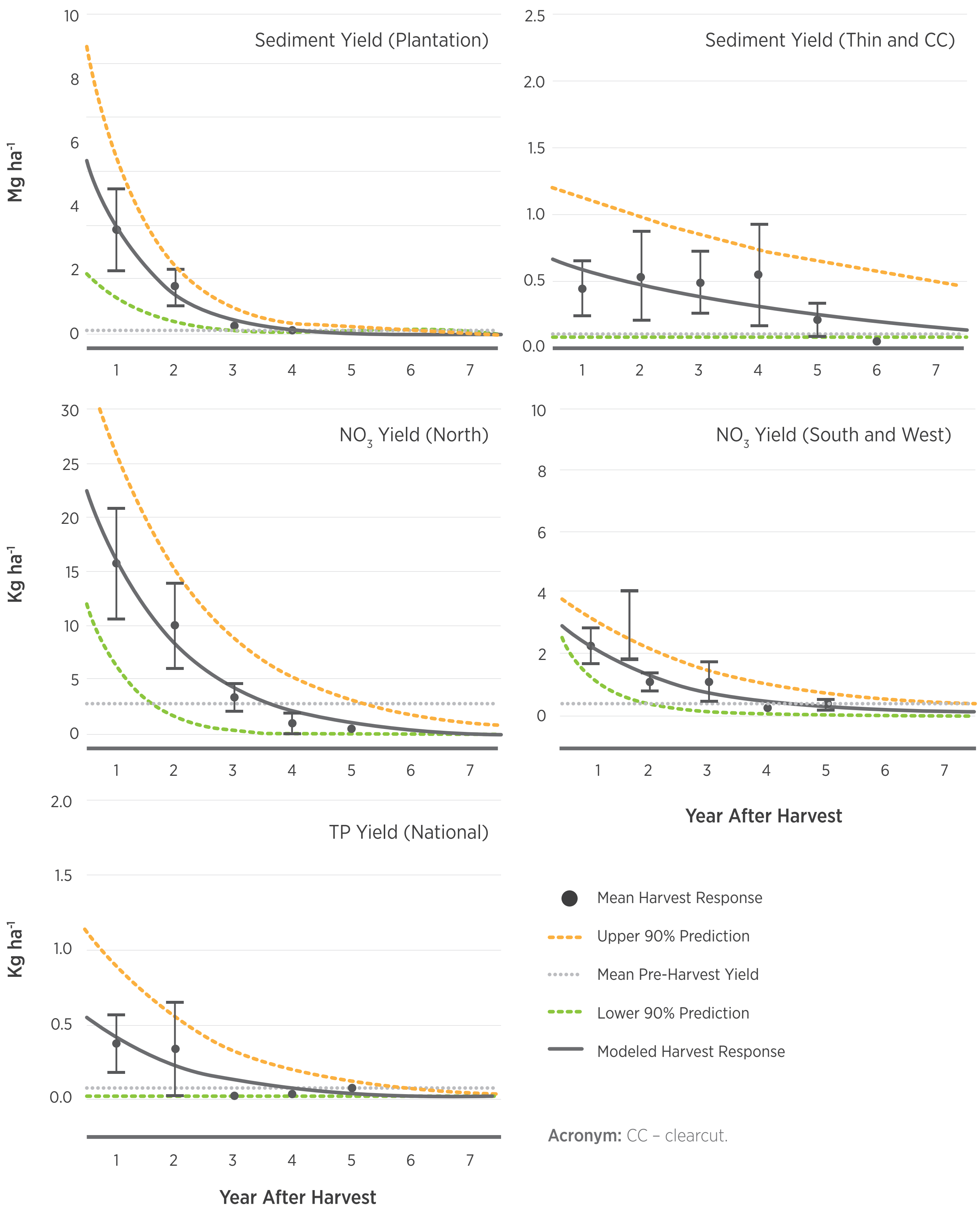
The load-response curves were generated from annual means, based on the results of the mixed model, and were best fit by the exponential decay function described in equation 6.1 (fig. 6.2). The modeled mean response period after harvest for sediment load from plantations across all regions was 4.4 years with an integrated response load of 8,798 kg/ha. By comparison, the mean response period for sediment from non-plantation harvests across all regions was 8.8 years, but with a response load of only 2,881 kg/ha. Over the life of the rotation, typical average annual rates of sediment yield from agriculture are typically much higher.[1] The mean response period for NO_3^--N

[1] Over a typical 30-year pine plantation rotation, the average sediment load delivered to a water body is 520 kg/ha/year if BMPs are utilized. Over a similar 30-year period, agricultural production with and without BMPs applied may produce 2,700 kg/ha/year and 18,000 kg/ha/year of sediment loading respectively (Hill 1991).

in the northern region for all harvest types was 3.7 years, with a mean response load of 43 kg/ha. The mean response period for NO_3^--N across all harvest types for the rest of the United States was 4.3 years with a mean response load of 6 kg/ha. For TP, the mean response time was 3.9 years, and the response load was 1.0 kg/ha across all regions and harvest types. Table 6.3 provides the full suite of coefficients of the fitted model, as well as related statistics for means and 90% confidence intervals of response loads and periods.

Non-aggregated, county-scale graphical depictions of sediment, NO_3^--N, and TP increases due to biomass harvest can be found in figures 6.3–6.5. The complete series of regional reference estimates, pre-harvest estimates, and increases due to biomass harvesting can be found in table 6.4.

Figure 6.2 | Sediment, NO_3^--N, and TP load response curves and the 90% prediction intervals generated from the results of the mixed model comparing regions and harvest types. Bars represent the upper and lower 90% confidence limit for each mean in each year after harvest.



Acronym: CC – clearcut.

Figure 6.3 | Graphical depiction of sediment load (in megagrams, Mg) due to potential biomass harvest under the select demand scenarios. Data are presented for individual counties where biomass harvest occurs.

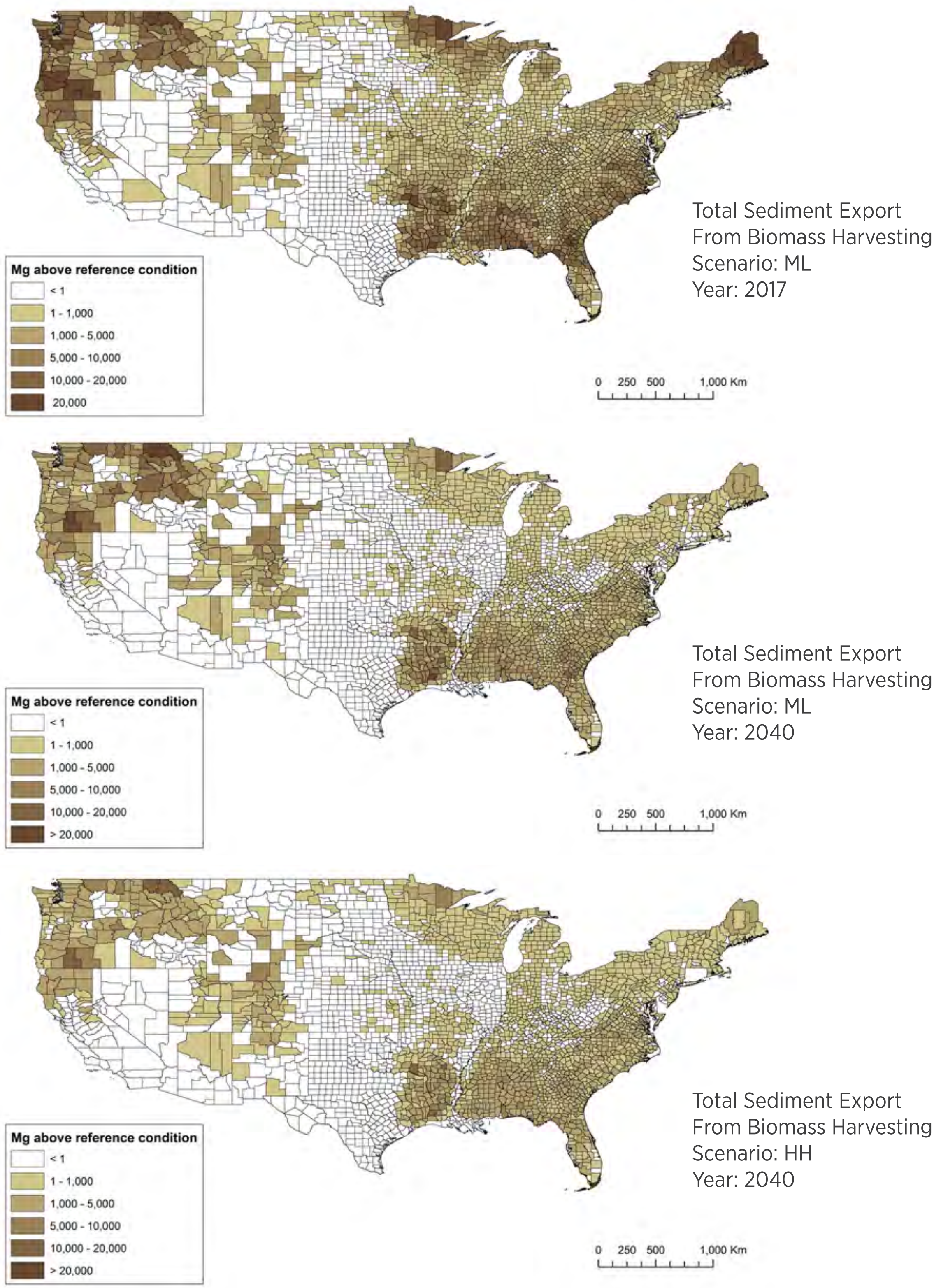
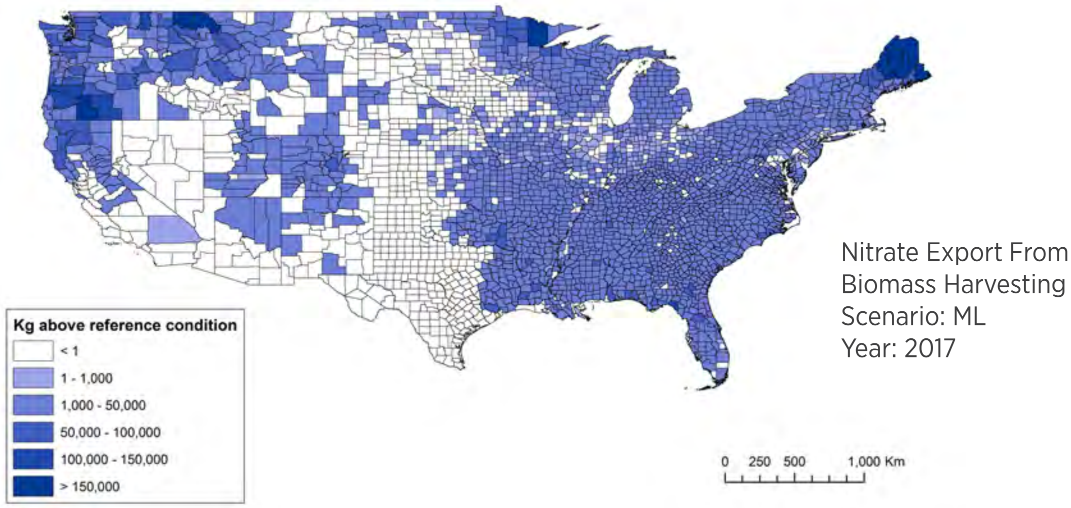
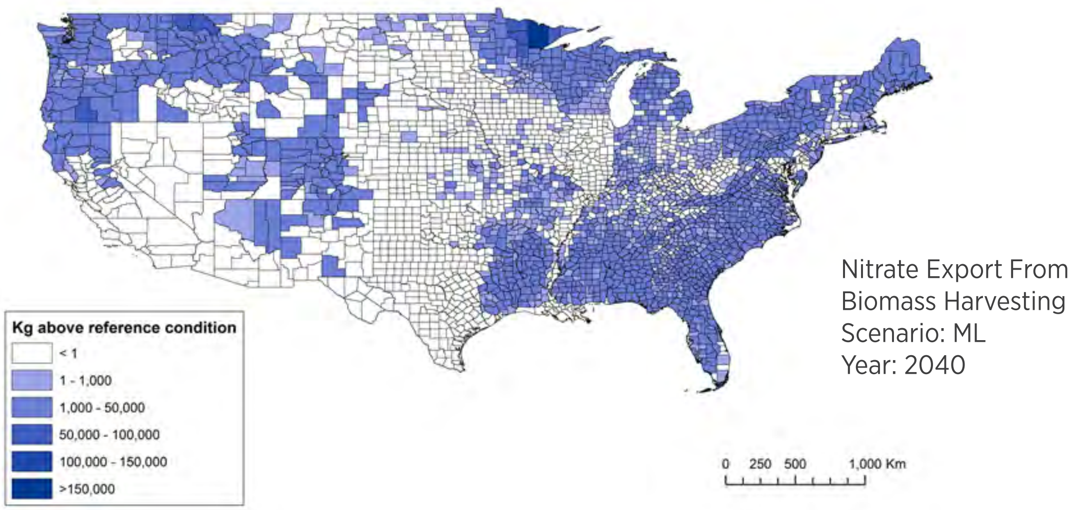


Figure 6.4 | Graphical depiction of nitrate load (in kg) due to potential biomass harvest under the select demand scenarios. Data are presented for individual counties where biomass harvest occurs.





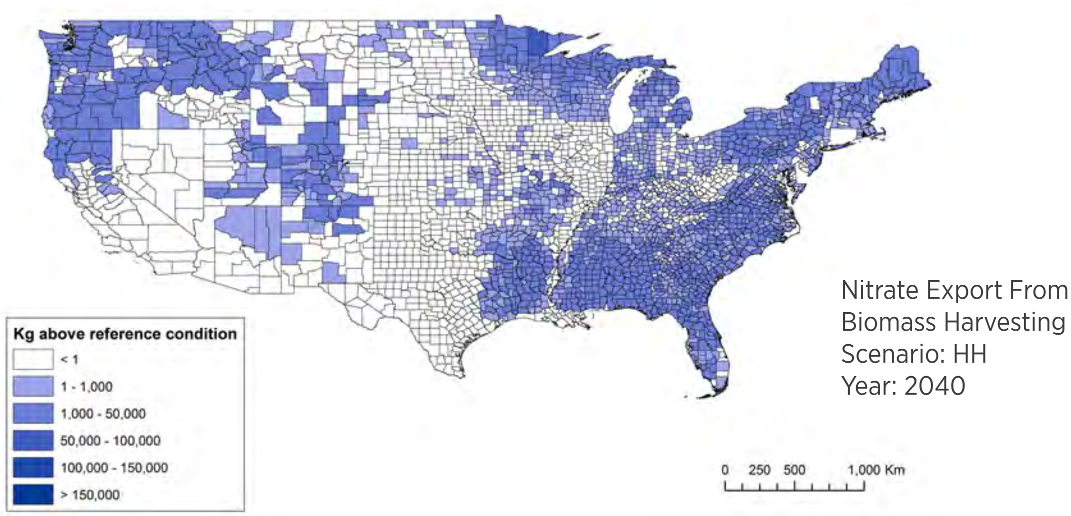


Figure 6.5 | Graphical depiction of total phosphorus load (in kg) due to potential biomass harvest under the select demand scenarios. Data are presented for individual counties where biomass harvest occurs.

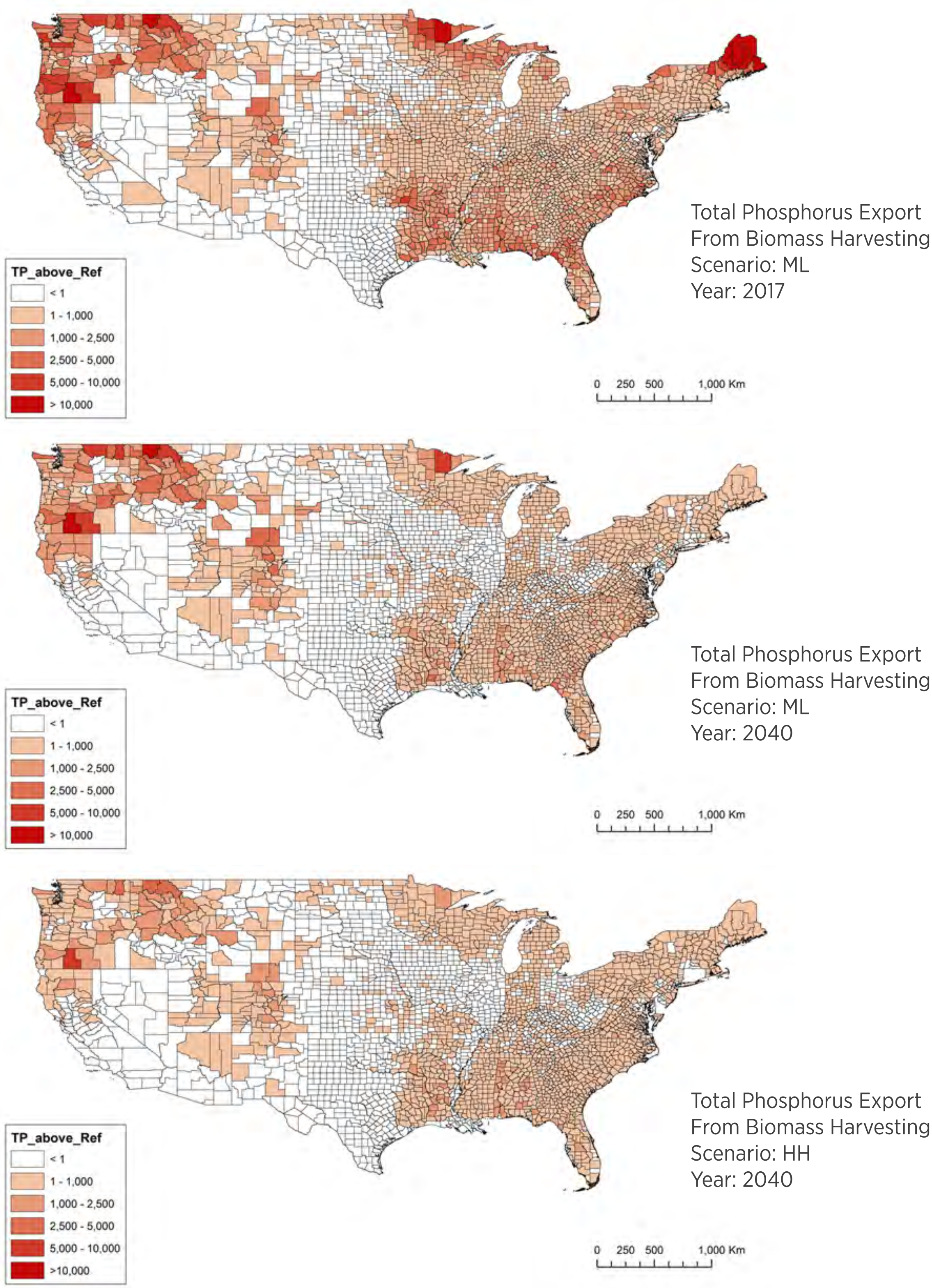


Table 6.3 | Parameters for the Mean and 90% Prediction Interval, Water-Quality Response Curves

Response variable	Region	Coefficient a	Coefficient b	R^2	P-value	Response period (years)	Response load integral (kg/ha)
Sediment plantation 90% LPL	National	3308.46	-1.17	0.98	0.0096	2.9	2,738
Sediment plantation mean	National	8971.63	-1.01	0.79	0.0104	4.4	8,798
Sediment plantation 90% UPL	National	14754.48	-0.98	0.98	0.0110	5.0	14,972
Sediment 90% LPL	National	104.54	-0.01	0.00	0.9488	0.0	0
Sediment mean	National	742.00	-0.22	0.66	0.0137	8.8	2,881
Sediment 90% UPL	National	1288.38	-0.14	0.28	0.2850	18.2	8,686
NO_3^- 90% LPL	Northern	23.86	-1.34	0.61	0.1171	1.6	16
NO_3^- mean	Northern	31.77	-0.68	1.00	<0.0001	3.7	43
NO_3^- 90% UPL	Northern	44.26	-0.54	0.95	0.0054	5.2	77
NO_3^- 90% LPL	National	5.33	-1.47	0.89	0.0152	1.9	3
NO_3^- mean	National	3.86	-0.57	0.90	0.0245	4.3	6
NO_3^- 90% UPL	National	4.62	-0.39	0.73	0.0656	6.7	11
TP 90% LPL	National	0.01	-0.04	0.17	0.4849	0.0	0
TP mean	National	0.74	-0.61	0.78	0.0488	3.9	1
TP 90% UPL	National	1.46	-0.50	0.56	0.1461	6.1	3

Acronyms: LPL – lower prediction limit; UPL – upper prediction limit.

6.3.1 Baseline Scenario ML 2017

Under the baseline scenario, in the year 2017, it is estimated that sediment loading would be greatest from the southern region of the United States and that the mean sediment load of 4,300 Gg represents a 39% increase over the regional reference for sediment load from current forest management (table 6.4). Mean sediment load attributed to biomass harvest from the northern region (1,400 Gg) and the western region (1,300 Gg) would be considerably lower than in the southern region and would represent 19% and 12% increases, respectively, over regional reference conditions derived from current forest management (table 6.4). Under the baseline 2017 scenario, total NO_3^--N loading from biomass harvesting is estimated to be greatest from the northern region with an additional 15 Gg or 2% increase occurring on average over reference conditions (table 6.4). Total NO_3^--N load from the southern (4 Gg) and western regions (2 Gg) represent 3% and 1% increases, respectively, over reference conditions (table 6.4). Similar to sediment, TP load from biomass harvesting is estimated to be greatest in the southern region, where an additional 0.9 Gg of TP would represent a 13% increase over reference conditions (table 6.4). TP load from biomass harvest in the ML 2017 scenario is estimated to be 0.5 Gg from the northern region and 0.4 Gg from the western region, or 10% and 13% increases, respectively, over reference conditions (table 6.4).

6.3.2 ML 2040 and HH 2040

In 2040, under the ML scenario, sediment, NO_3^--N, and TP delivery to water bodies due to biomass harvesting all decrease below the 2017 baseline. For instance, mean sediment load decreases to 1,800 megagrams (Mg) in the South, and to 700 Mg and 200 Mg in the West and North, respectively (table 6.4). These mean sediment loads are approximately 16%, 7%, and 2% increases, respectively, over regional reference conditions (table 6.4). The main driver of this decrease in post-biomass-harvest load is the assumptions made in ForSEAM. The model assumes that no new land will be converted to plantation forestry in the southeastern United States—even if demand for wood products increases. Therefore, greater quantities of wood products are diverted to housing and other building supply chains rather than to biomass for energy. Under this scenario, NO_3^--N load decreases to 2 Gg in the North, 1.6 Gg in the South, and 1.2 Gg in the West. All the decreases are ≤1% above regional references (table 6.4). Similarly, total post-biomass-harvest loads for TP were obtained as 0.4 Gg in the South, 0.3 Gg in the West, and 0.1 Gg in the North (table 6.4).

The HH scenario results in a further reduction of biomass-harvest-related sediment, NO_3^--N, and TP loads in 2040 (fig. 6.6). Under this scenario, significant wood resources are diverted into housing, and demand for biomass cannot be met. The sediment load attributable to biomass harvest in the South falls to 1,100 Gg and to 300 Gg and 100 Gg in the West and North, respectively (table 6.4), which represent 10%, 3%, and 1% increases over regional reference conditions (table 6.4). Under the HH 2040 scenario, NO3--N is actually higher in the South compared to the North and West, but decreases to 1.3 Gg while loads in the North and West are 1.1 Gg and 0.5 Gg, respectively (table 6.4). All NO_3^--N loads in this scenario represent ≤1% increase over reference conditions (table 6.4). TP loads from biomass harvest under the HH 2040 scenario are highest in the South, but are well below 1 Gg in each region, as shown in figure 6.6. TP loads are 4% in excess of reference values in the South, 2% over reference in the West, and 1% over regional reference in the North (table 6.4).

6.4 Discussion

The water-quality estimates obtained using the empirical models derived from the peer-reviewed literature and applied to potential biomass utilization in select scenarios show there could be regional variation in how biomass harvest would influence water quality. Sediment loads often increase after intensive site preparation in plantations. Because these practices are most common in the South, our estimates indicate that absolute sediment loads and percent increases over reference conditions would be greatest in the South, with smaller increases in the West and North. Alternatively, estimates indicate that absolute NO_3^--N loads would increase most in the North, but when considered as an increase over regional reference, the highest increase occurs in the South, followed by the North and then the West in ML 2017. In the ML 2040 and HH 2040 scenarios, the largest percent increase is still estimated to be in the South, but the West surpasses the North (table 6.4). The pattern observed is likely due to two factors. The northern region of the United States, where many of the peer-reviewed studies of harvest effects on nutrient and sediment load were conducted, has a long legacy of atmospheric nitrogen deposition from industrial processes. This legacy has led to increased reference concentrations of NO_3^--N in much of the region. When vegetation is removed from forests in the region, temporary spikes in NO_3^--N are common due to reduced plant uptake. However, because the reference-load values are large, their increase after harvest may be relatively small when considered as a percentage of total load. In contrast, the South and West reference NO_3^--N loads are lower, so changes after harvest can be a larger percentage of total loads. The changes in regional NO_3^--N loads over time in alternative biomass-demand scenarios occur due to the dynamic nature of ForSEAM, which models supply and demand at the regional scale as well. Because a single model was applied for TP response to all biomass harvests, the estimated regional differences in TP response to biomass harvest and change over time, as well as intensity, are solely due to the forested acres within a region and the supply and demand for biomass.

The estimated response to biomass harvest indicates that sediment flux is the most dynamic water-quality parameter; sediment flux typically increases after harvests, particularly in areas where mechanical site preparation is common prior to planting. However, chemical herbicides are becoming economically viable and effective alternatives to mechanical site preparation for controlling competition during the early stages of plantation development. If this trend of increasing herbicide use continues, then sediment loads are likely to decrease below what has been estimated here. The estimated responses for NO_3^--N and TP tend to be less dynamic and typically result in <10% increase over reference loads. For all water-quality parameters, the load-response period is typically <5 years. Silvicultural activities generally occur on relatively few acres each year compared to the total forested acres within any given watershed, and activities typically only occur on the same tract of land once during a stand rotation. Therefore, the effects of silvicultural activities on water quality are typically small when compared to current agricultural activities involving annual crops (on a per-area basis); which typically occur multiple times each year on the same tract of land (Shepard 1994). Continued adherence to and increased adoption of BMPs on lands on which silviculture is practiced should minimize biomass-harvest effects.

Table 6.4 | Mean Region Reference Load, Pre-Harvest Load, and the Increase over Reference Load after Biomass Harvest Expressed as Total Regional Flux and a Percentage of Reference Load with Lower (LPL) and Upper (UPL) 90% Prediction Limits.

			Raw Values		Increase over Pre-Harvest			% Change over Regional Baseline		
Scenario	Year	Region	Sed. Region Baseline (Gg)	Sed. Pre-Harvest (Gg)	Sed. LPL (Gg)	Sed. Mean (Gg)	Sed. UPL (Gg)	Sed. LPL	Sed. Mean	Sed. UPL
ML	2017	North	7,400	50	60	1,400	4,000	0.8%	19%	54%
ML	2017	South	11,000	100	810	4,300	9,600	7%	39%	87%
ML	2017	West	10,200	40	130	1,300	3,300	1%	12%	32%
ML	2040	North	7,400	10	4	200	500	0.1%	2%	7%
ML	2040	South	11,000	40	360	1,800	3,800	3%	16%	34%
ML	2040	West	10,200	30	2	700	2,200	0.0%	7%	22%
HH	2040	North	7,400	4	3	100	300	0.0%	1%	4%
HH	2040	South	11,000	30	180	1,100	2,700	2%	10%	25%
HH	2040	West	10,200	10	0	300	1,000	0.0%	3%	10%
			Raw Values		Increase over Pre-Harvest			% Change over Regional Baseline		
Scenario	Year	Region	NO_3^- Region Baseline (Gg)	NO_3^- Pre-Harvest (Gg)	NO_3^- LPL (Gg)	NO_3^- Mean (Gg)	NO_3^- UPL (Gg)	NO_3^- LPL	NO_3^- Mean	NO_3^- UPL
ML	2017	North	670	4.4	2.7	15.1	30.4	0.4%	2%	5%
ML	2017	South	150	1.3	1.7	4.2	8.5	1.2%	3%	6%
ML	2017	West	140	0.5	0.7	1.6	3.3	0.5%	1%	2%
ML	2040	North	670	0.6	0.4	2.0	4.0	0.1%	0.3%	0.6%
ML	2040	South	150	0.5	0.7	1.6	3.3	0.5%	1.1%	2.2%
ML	2040	West	140	0.4	0.5	1.2	2.5	0.4%	0.9%	2%
HH	2040	North	670	0.3	0.2	1.1	2.3	0.0%	0.2%	0.3%
HH	2040	South	150	0.4	0.5	1.3	2.5	0.4%	1%	2%
HH	2040	West	140	0.2	0.2	0.5	1.1	0.2%	0.4%	0.8%

Acronym: Sed. = sediment

			Raw Values		Increase over Pre-Harvest			% Change over Regional Baseline		
Scenario	Year	Region	TP Region Baseline (Gg)	TP Pre-Harvest (Gg)	TP LPL (Gg)	TP Mean (Gg)	TP UPL (Gg)	TP LPL	TP Mean	TP UPL
ML	2017	North	4.7	0.03	0.0	0.5	1.2	0.0%	10%	26%
ML	2017	South	7.0	0.06	0.0	0.9	2.4	0.0%	13%	34%
ML	2017	West	6.5	0.02	0.0	0.4	0.9	0.0%	6%	14%
ML	2040	North	4.7	0.00	0.0	0.1	0.2	0.0%	1%	3%
ML	2040	South	7.0	0.02	0.0	0.4	0.9	0.0%	5%	13%
ML	2040	West	6.5	0.02	0.0	0.3	0.7	0.0%	4%	11%
HH	2040	North	4.7	0.00	0.0	0.03	0.1	0.0%	1%	2%
HH	2040	South	7.0	0.02	0.0	0.3	0.7	0.0%	4%	10%
HH	2040	West	6.5	0.01	0.0	0.1	0.3	0.0%	2%	5%

6.5 Uncertainties and Limitations

Within the vast body of silviculture-based literature reviewed, only 38 studies could be identified that reported sediment and nutrient loading to a body of water. Fewer than 10% of those studies identified sites monitored long enough to determine that sediment and nutrient loads after harvest had returned to pre-harvest levels, defined here as the response period. Therefore, the mean load responses for the response periods were modeled, and 90% prediction intervals were determined to illustrate the ranges of possible responses as uncertainties in estimates (fig. 6.2 and table 6.3). Within the literature selected for this study, not all publications measured all variables of interest. The number of publications reporting data for sediment, NO_3^--N, and TP were, 24, 20, and 9, respectively. Similarly, the number of studies found for each region was not equal, with 23 studies representing the South, 9 from the North, and 6 from the West. In addition, not all studies reported data for the same number of years post-harvest. Furthermore, harvest type was not represented evenly, and within each region, there were differences in stocking rates, harvest rates, soil type, slope, aspect, vegetation type, and climate between studies. This resulted in an uneven number of data points for each variable and statistical uncertainty in computed parameters. To test the applicability of the model for load response, mean absolute error (MAE) and root mean square error (RMSE) (Chai and Draxler 2014) were calculated using the data reported from the literature and estimated values (table 6.5). The magnitude of the MAE and RMSE values was found to be minimal for NO_3^--N and TP. However, the MAE and RMSE for sediment were relatively high, perhaps due to the variability of management operations used to manipulate surface soil. We acknowledge that there may be other studies that were not examined in this analysis that may influence the statistics and model estimates.

Table 6.5 | Mean Absolute Error (MAE) and Root Mean Square Error (RMSE) for Literature-Derived Data and Projected Load-Response Value Comparisons

Parameter	MAE-Reference	RMSE-Reference	MAE-Treatment	RMSE-Treatment
NO_3^--N	1.420	1.800	0.020	0.030
TN	0.002	0.002	0.002	0.002
TP	-0.001	0.001	-0.003	0.003
Sediment	-0.002	0.002	729.5	1,029.8

Acronym: TN – total nitrogen.

Because ForSEAM was used to generate the potential biomass and acres harvested under each scenario, our estimates of changes to water quality from biomass harvest are subject to the assumptions and limitations of ForSEAM as well. In particular, the assumption that no new plantations will be established in the southern United States drives the trend in decreasing sediment and nutrient load with increasing demand for wood products. As demand for wood products increases in the housing sector, less biomass is available for energy production, and therefore, less sediment and nutrient load is attributable to biomass harvests.

The values for sediment, NO_3^--N, and TP presented here are only meant to represent the additional response to harvesting biomass, and they do not include the effects of associated harvests for other wood products; therefore, the results are incremental. Similarly, the additional sediment and nutrient load produced by biomass harvest is compared to a reference considering pre-harvest forest watershed conditions and does not include any discharges due to concurrent silviculture, agriculture, or other activities.

6.6 Summary and Future Research

Our objectives were to utilize select scenarios from *BT16* to estimate the effects of potential forest biomass removal on water quality at regional scales. However, the data available from peer-reviewed literature were not sufficient to warrant multivariate models relating biomass harvested to changes in water quality. Therefore, a simple, empirical modeling approach was developed to estimate sediment and nutrient response to the total acres estimated to be harvested for biomass within a given county, and then, results were aggregated to three regions of the United States.

This simple modeling approach produces a wide range of potential outcomes because of high levels of uncertainty associated with both the derived models and each data point within the model. This is particularly true for sediment load. Despite this limitation, the results offer an initial estimate of the magnitude of possible effects relative to current forestry and agricultural practices. For example, sediment load for biomass harvesting from plantation forestry is estimated to be less than 9 Mg/ha over 4.4 years. On an average annual basis, this sediment loading rate is about 20% of rates associated with agriculture with BMPs[2] and about 3% of rates associated with agriculture without BMPs (Hill 1991).

A process-based modeling approach would likely be most appropriate for this task, because there are nearly infinite combinations of soil type, topography, climate, vegetation, and harvest systems involved in estimating water-quality response to biomass harvests. However, at this time, very limited process-based modeling platforms are available to conduct large-scale distributed modeling of silvicultural activities (Amatya et al. 2013). It is imperative that forest-sector field researchers collaborate with engineers and modelers to develop, parameterize, and test process-based models for silvicultural activities. Rather than starting from scratch, it may be worthwhile to utilize platforms from the agricultural sector as Amatya et al. (2013) did when modeling the fate of nitrogen in forest ecosystems.

Often, silviculture is not the only use of land within a watershed, and silvicultural effects on water quality are not isolated. It is critical that we begin to model watersheds with multiple land uses so that silviculture, agriculture, urban, and other land uses can all be integrated to estimate cumulative effects while assessing their individual effects as well.

Additional research is also needed to fill in the gaps in the existing literature. Where possible, long-term watershed-scale research should continue to determine the effects of traditional and emerging silvicultural practices on water quality. Based on findings from this study, additional studies from the West, Intermountain West, Upper Midwest, North, and South states would fill in gaps in the knowledge base. There are several established experimental forests and watersheds throughout the United States. Many of these sites have been monitored for extended periods of time (Amatya et al. 2016). To maximize the value of these research installations, a coordinated series of experiments could be implemented to determine how emerging silvicultural practices, including biomass utilization, interact with variable climate and soils to influence water quality. These experiments could be modeled after the Long-Term Soil Productivity Experiment or the Long-Term Agricultural Research Network and could incorporate periodic herbicide application, fertilization, and thinning, or multiple rotations.

[2] BMPs commonly utilized in agriculture include cover cropping, no-till or reduced tillage practices, contour cropping, crop rotations, perennial grass or forested riparian filter strips, grass swales, sediment detention basins, retention ponds, wetland basins, as well as manufactured media filters and porous pavement.

6.7 References

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7 | Impacts of Forest Biomass Removal on Water Yield across the United States

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7.1 Introduction

Water is essential to all forms of life on earth and is a powerful, integrated indicator of environmental health and ecosystem sustainability (Asbjornsen et al. 2015). In some areas of the United States, water availability and water quality are declining as a result of urbanization, climate change, and increased water demand for agricultural irrigation, power generation, and domestic water use (Sun et al. 2008). Forest hydrological studies across the United States and around the world in the past century (Vose et al. 2011) show that forests greatly influence water quantity and quality. Forests play an important role in regulating the quantity, quality, and timing of water yield from watersheds—and, thus, in maintaining the ecosystems that depend on water (Edwards, Williard, and Schoonover 2015). It is estimated that over half of the water supply from the United States is provided by domestic forestlands (Brown, Hobbins, and Ramirez 2008; Sun, Caldwell, and McNulty 2015); therefore, forest management—such as reforestation/afforestation, tree harvesting, stand thinning, and other forest management practices—can influence watershed water yield (i.e., outflow from a drainage basin) by altering the terrestrial hydrological cycle. This cycle involves precipitation, evapotranspiration (ET), infiltration, soil moisture dynamics, and streamflow (Sun, Caldwell, and McNulty 2015; Stednick 1996; Christopher, Schoenholtz, and Nettles 2015). For example, deforestation generally elevates total streamflow and peak flow rates due to the reduction of ET caused by the removal of forest canopies (Brown et al. 2013), decrease in soil infiltration capacity as a result of soil compaction (Bruijnzeel 2004), and forest road construction (Edwards and Williard 2010). In contrast, afforestation or reforestation generally decreases watershed water yield because ET increases as a result of increase in water use by trees that have greater biomass both above- and belowground than vegetation in previous land uses (Sun et al. 2010; Brown et al. 2005).

Harvesting biomass from forests is one potential approach to both meeting increasing bioenergy demand and contributing to energy security in the United States (Evans 2016; Caputo et al. 2016; Holland et al. 2015). It is important to evaluate the environmental effects of various biomass harvesting methods and removal fractions to make sure that the harvesting of biomass does not harm aspects of the environment, such as water quality and water supply (King et al. 2013; Bonsch et al. 2016; Caputo et al. 2016; Christopher, Schoenholtz, and Nettles 2015). Supply constraints applied in *BT16* dictate that biomass removal is excluded from environmentally sensitive areas and is limited to a fraction of the total biomass available. Although these constraints are intended to reduce potentially negative environmental impacts, more thorough analyses are required for better planning of harvesting biomass, as well as better understanding of how these effects differ across locations, biomass types, and management practices (Lin, Anar, and Zheng 2015; Christopher, Schoenholtz, and Nettles 2015).

In addition, water quality is intrinsically linked to water quantity. As such, it is important to examine water quantity consequences in addition to impacts on water quality as a result of biomass removal (Binkley, Burnham, and Allen 1999). Changes in water quantity due to forestry activities are likely to affect water quality because water quantity affects both concentrations of stream water nutrients and other chemicals and total loading of chemicals and sediment. For example, forest harvesting may increase streamflow in forested watersheds and, therefore, may increase overland flow, peak flow rates, stormflow volume, which results in stream bank and channel erosion and increased sediment loading (Boggs, Sun, and McNulty 2015; Cristan et al. 2016).

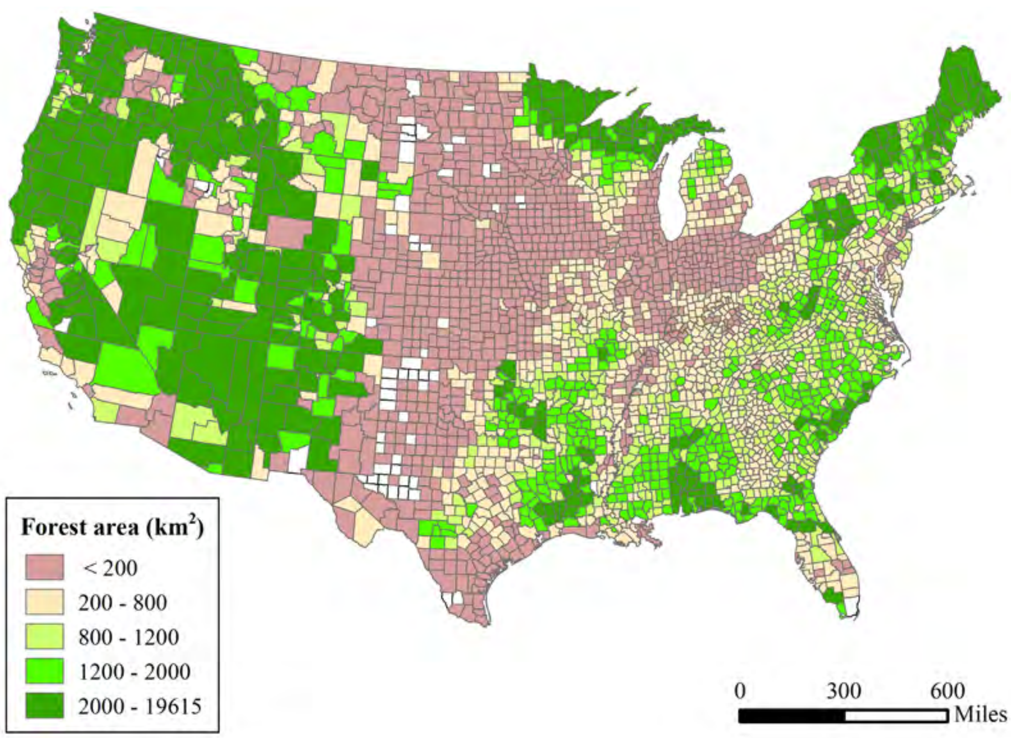
The overall goal of this chapter is to evaluate the potential effects of select *BT16* scenarios of forest-biomass harvesting on water quantity. The specific objective of the study is to quantify the water yield at both watershed (12-digit hydrologic unit code [HUC 12]) and county levels across the lower 48 states. The study focuses on the effects of potential forest removals on the seasonal and annual total water yield at watershed and county scales. Counties that are sensitive to biomass removal are identified to help reduce the risk of environmental degradation and to maximize the positive effects of biomass production on watershed functions.

The following hypotheses have been used to guide this analysis: (1) forest removals decrease water use by trees and canopy interception of precipitation, and thus cause an increase in water yield and water availability for human and aquatic ecosystems; (2) the magnitude of streamflow increase depends on the level of biomass removal per unit area (e.g., thinning intensity), the total amount of forest removed (e.g., the acreage cut) and the local background climate (i.e., dry or wet environment as indicated by climate dryness index); and (3) effects of biomass removal on water quantity have a large spatial and temporal (i.e., seasonal) variability because of differences in biophysical characteristics.

7.2 Methods

We applied a watershed-scale hydrological modeling approach with biomass harvesting scenarios as the driving forces of hydrologic disturbances under a mean climatic condition (1991–2001). Water-yield responses to complete tree harvesting (100% clear-cutting) or thinning (70% reduction in leaf area index [LAI]) are first examined to quantify the maximum potential impacts per unit of land area at the watershed scale (HUC 12), and then at the county level, by scaling up watershed-scale data. Then, the area of harvesting (clearcutting or thinning) by county from BT16 volume 1 is applied to the complete-harvesting datasets to quantify the projected effects due to potential forest biomass removal at the county level from scenarios in *BT16* volume 1. The forestland area is estimated from the National Land Cover Database 2011 (NLCD 2011) and has a spatial resolution of 30 meters (m) (Homer et al. 2015).

Figure 7.1 | Forestland area by county as determined by the NLCD (2011). The data are from 2006.



7.2.1 Scope of Assessment

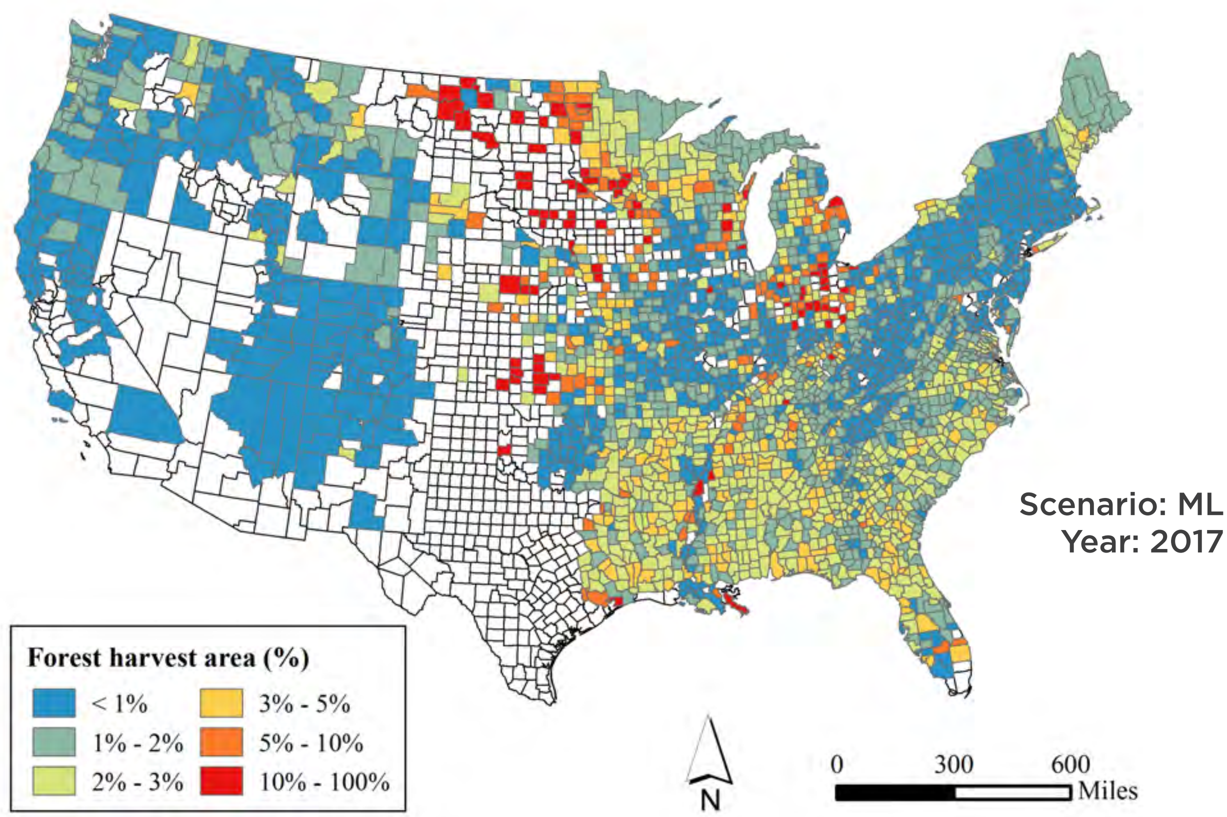
This analysis evaluates water-yield responses to select harvesting scenarios: ML 2017, ML 2040, and HH 2040. These three scenarios represent two levels of biomass demand and two time periods. The ML scenarios represent the baselines while the HH scenario represents the forestry high-housing, high-biomass demand scenario. Areas of harvesting from thinning and clearcutting are compared to total forest areas from NLCD 2011 data (fig. 7.1) in each county to derive harvesting area ratios (percent) for estimating the likely change in water yield from the potential maximum water yield response if the entire forest area in the county were harvested. A majority of counties have a harvesting area, either clearcutting or thinning, that encompasses less than 2% of the land area by county (fig 7.2). In addition, the baseline ML 2017 scenario has the highest potential biomass removal of the three scenarios that were examined. Areas showing high percentage harvesting (>5%) are located at the forest-grassland or forest-cropland transition zones with limited forest biomass potential. Data errors for these areas may exist since the harvesting area data are derived from models and Forestry Inventory Analysis (Nelson and Vissage 2007), but the forestland areas (fig. 7.1) are determined from remote sensing imagery.

The projected hydrological response to forest harvesting is estimated based on the maximum potential response in each county if the entire forest were harvested, with an assumption that the response is proportional to forest removal:

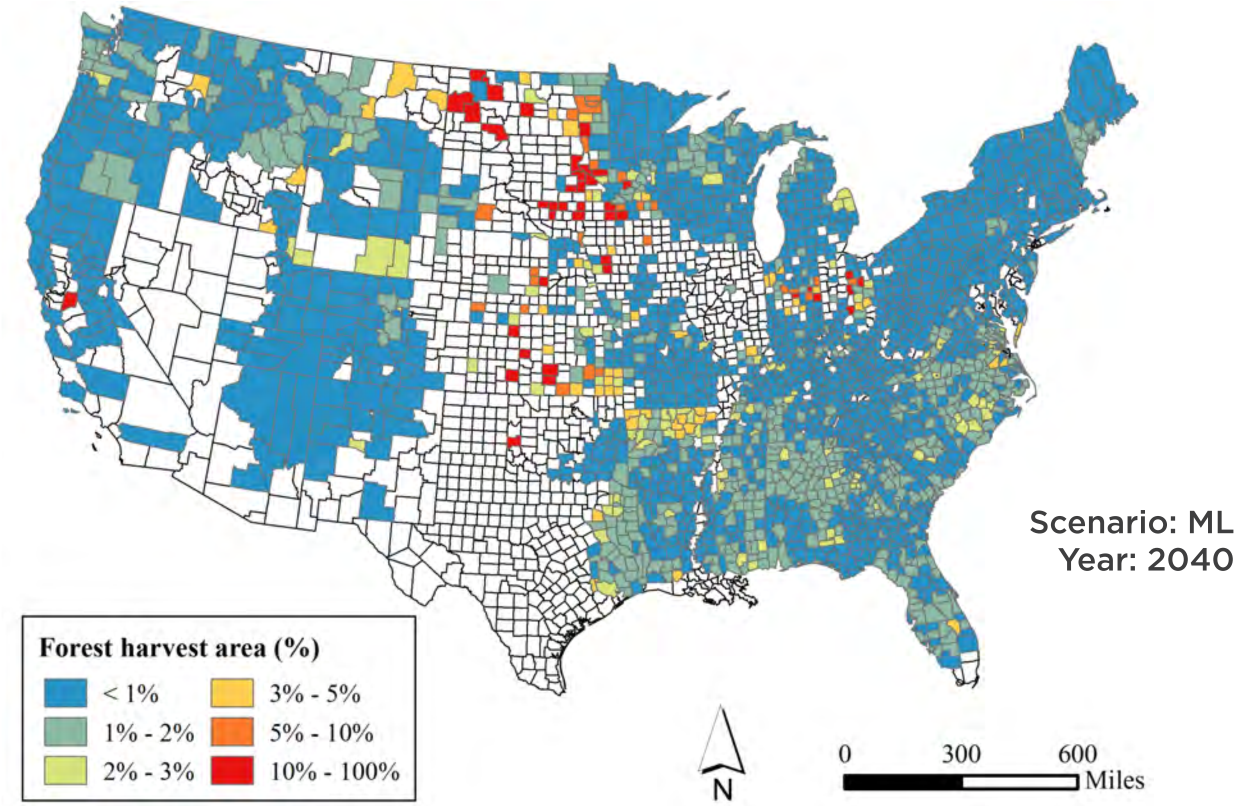
Projected hydrological response = maximum potential hydrological response × percentage of forest harvest area

Figure 7.2 | Forest harvesting percentage (clearcutting plus thinning) under three biomass scenarios in 2017 and 2040

A.



B.


Forest harvest area (%)
< 1%
1% - 2%
2% - 3%
3% - 5%
5% - 10%
10% - 100%
N
0
300
600
Miles
Scenario: ML
Year: 2040

C.

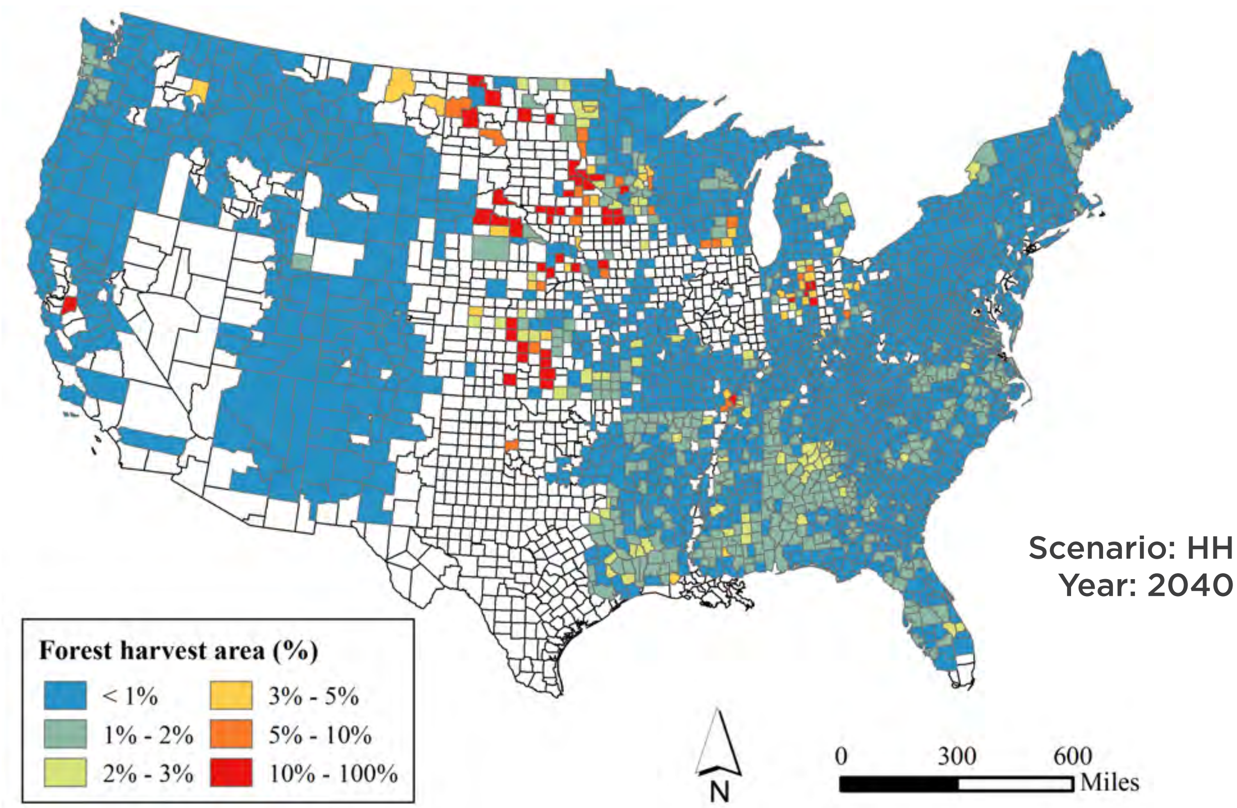
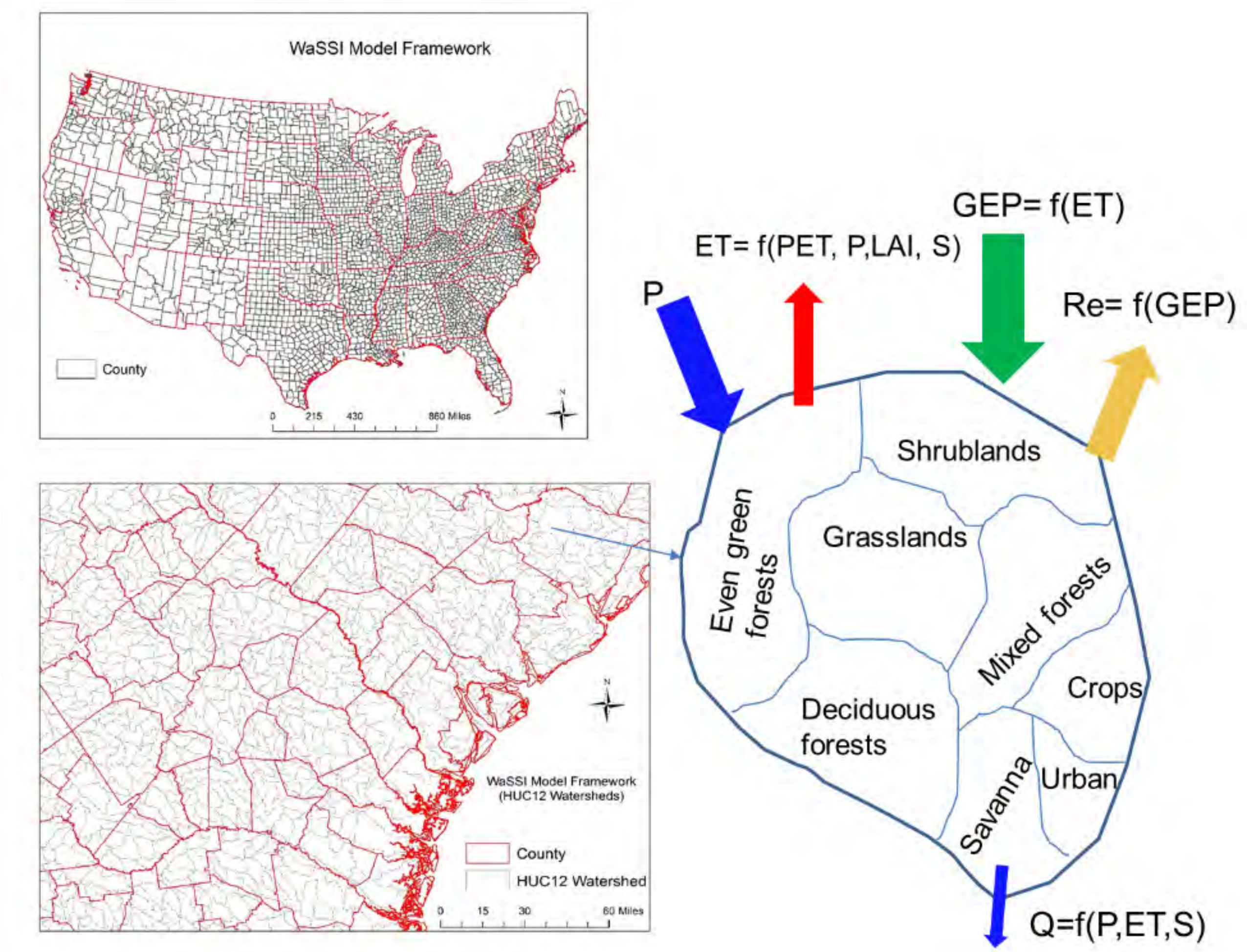

Forest harvest area (%)
< 1%
1% - 2%
2% - 3%
3% - 5%
5% - 10%
10% - 100%
N
0
300
600
Miles
Scenario: HH
Year: 2040

Figure 7.3 | Structure of the ecohydrological model (WaSSI) that simulates the full water and carbon balances at the HUC 12 watershed scale



7.2.2 Description of the Ecohydrological Model (WaSSI)

The WaSSI (Water Supply Stress Index) ecohydrological model (Sun et al. 2011b; Sun et al. 2008; Caldwell et al. 2012) was developed to examine the broad impacts of climate change, land cover/land use change, and population growth on water and carbon budgets and on water stresses at monthly and watershed scales (see fig. 7.3). WaSSI has been tested, validated, and applied at the 8-digit HUC (HUC 8) and HUC 12 watershed scales across the conterminous United States (Caldwell et al. 2015; Caldwell et al. 2012; Sun et al. 2015b; Sun et al. 2015a). The model simulates all monthly water fluxes (i.e., ET, infiltration, soil water storage, snow accumulation and melt, surface runoff, and base flow) for each of the land cover categories in a watershed with mixed land uses, as well as aggregates to the entire basin using an area weighted averaging method. Infiltration, soil storage, and runoff were estimated based on the algorithms from the Sacramento Soil Moisture Accounting Model and the 11 soil parameters derived from State Soil Geographic Data Base (STATSGO). The monthly ET model embedded in WaSSI was derived empirically using eddy flux and sap flow measurements at multiple sites from grassland to subtropical conifer forests (Sun et al. 2011a). ET was calculated as a function of potential ET (PET), which is calculated by a temperature-based PET equation, LAI, precipitation, and soil water content. Forest LAI data are derived from the Moderate Resolution Imaging Spectroradiometer (MODIS) product at a 1 kilometer (km) resolution. The WaSSI model has been applied to quantify the effects of introducing exotic tree species (e.g., Eucalyptus) on regional water budgets in the United States (Vose et al. 2015). Details of the WaSSI model can be found in the program's user guide: [http://www.forestthreats.org/research/tools/WaSSI/WaSSIUserGuide_english_v1.1.pdfforestthreats.org/research/tools/WaSSI/WaSSIUserGuide_english_v1.1.pdf.](http://www.forestthreats.org/research/tools/WaSSI/WaSSIUserGuide_english_v1.1.pdf)

In this study, it is assumed that the magnitude of biomass removals corresponds to change in LAI, the key parameter in the WaSSI model linking vegetation dynamics, water use (ET), and water yield (table 7.1). The total water yield response is the sum of the response to thinning and clearcutting activities. Water yield is modeled first at the HUC 12 scale and then is scaled to the county level using a weighted average approach. Water yield is expressed in both depth in millimeters (mm) and volume units (million cubic meters or million gallons).

Input data to the WaSSI model mainly include soil properties, land covers, LAI, precipitation, and air temperature. Monthly mean (2000–2006) LAI data were used in this modeling study that focus on sensitivity to LAI change. The 1 km STATSGO soil data were used to derive the 11 soil parameters. The watershed-level land cover compositions were scaled from 30 m using NLDC 2011 data for the year 2006 for the conterminous United States. The mean monthly 1 km LAI over 2000–2011 was derived from MODIS LAI products. The multi-year mean monthly LAI by land-cover type was computed by overlaying the land-cover data with MODIS LAI products. The monthly 4 km-scale precipitation and temperature data over the 1991–2001 averaging period were obtained from PRISM Climate Group data.

Model outputs from WaSSI include monthly and annual ET, water yield, and gross primary productivity by watershed. These variables at the watershed level are scaled to the county level using a weighted average approach. Water yield in a unit per land area (mm) is recalculated to convert to quantity in a volume unit (million cubic meters or gallons of water) at the county level by multiplying county land area with water yield in depth (mm).

Table 7.1 | Modeling Experiment Design That Includes Two Types of Biomass Removals (Thinning and Clearcutting) for 2 Years (2017, 2040) as Simulated by ForSEAM

Forest Biomass Harvesting	Effects on LAI
Reference	Mean LAI with land use in 2006; mean climate (1991–2001)
Thinning Three Harvesting Scenarios	Forest LAI decreased by 70%; mean climate
Clearcutting Three Harvesting Scenarios	Forest LAI decreased to 0.5; mean climate

7.3 Results

7.3.1 Potential Maximum Impacts of Forest Removal on Water Yield by County

Mean long-term annual water yield for each county (i.e., for the 1991-2001 reference period) varies greatly from less than 100 mm per year to as high as 2,012 mm because of the large differences in climate (e.g., precipitation and air temperature) across the United States (fig. 7.4). Water yield at the watershed and county level is also influenced by vegetation composition, soil characteristics, and precipitation forms (e.g., snow or rain). For example, forests have higher ET than non-irrigated croplands or grasslands and thus have lower water yield under the same climatic regime. High-elevation watersheds generally receive high precipitation and have low PET, and therefore produce high water yield.

Clearcutting forests can increase county-scale water yield from less than 10 mm per year in the dry areas to as high as 151 mm per year in the wet areas in coastal counties in the Pacific Northwest and the Appalachian region of the eastern United States (fig. 7.5A). These values represent the maximum hydrological response to clearing all forests in a county when comparing to current reference water yield. Thinning forests (reducing 70% of forest LAI) results in relatively lower impacts when compared to the clearcutting options (fig. 7.5B).

Figure 7.4 | WaSSI modeled reference long-term mean annual water yield by county across the United States

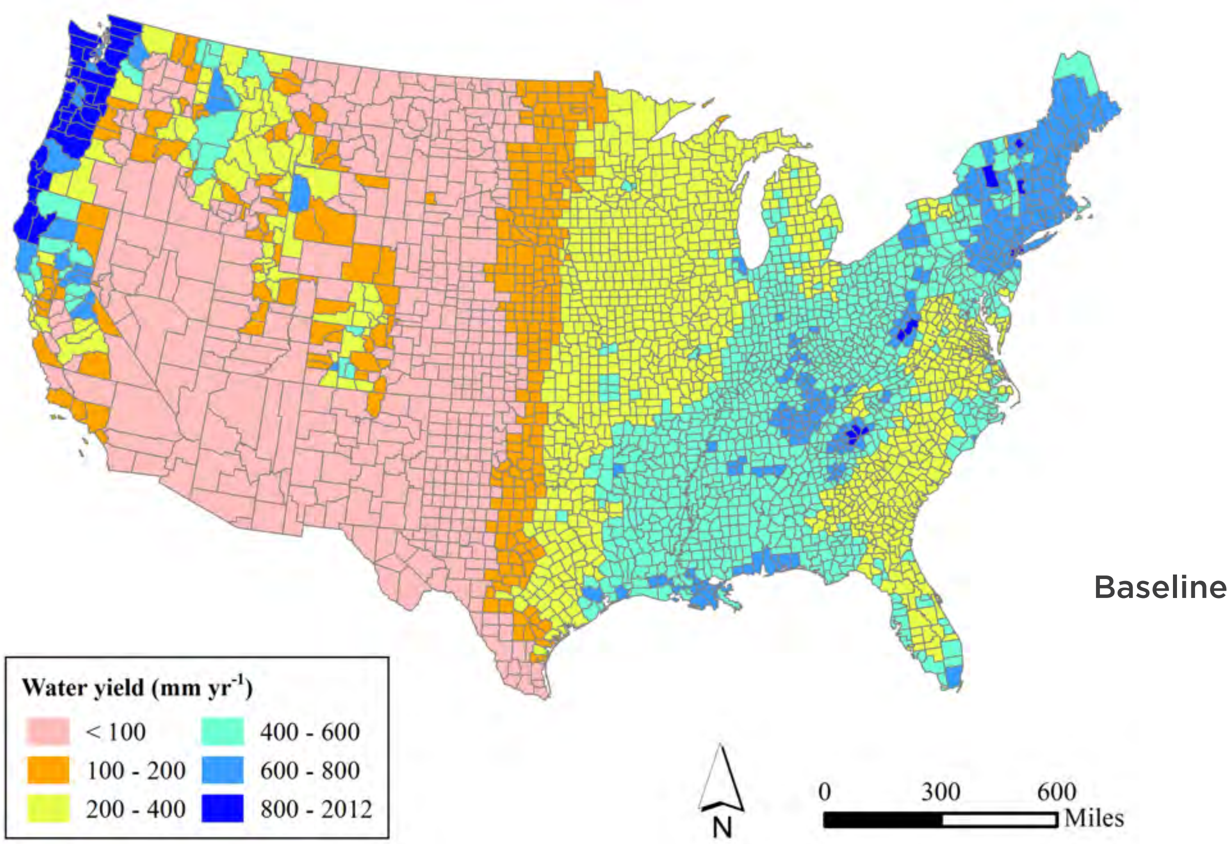
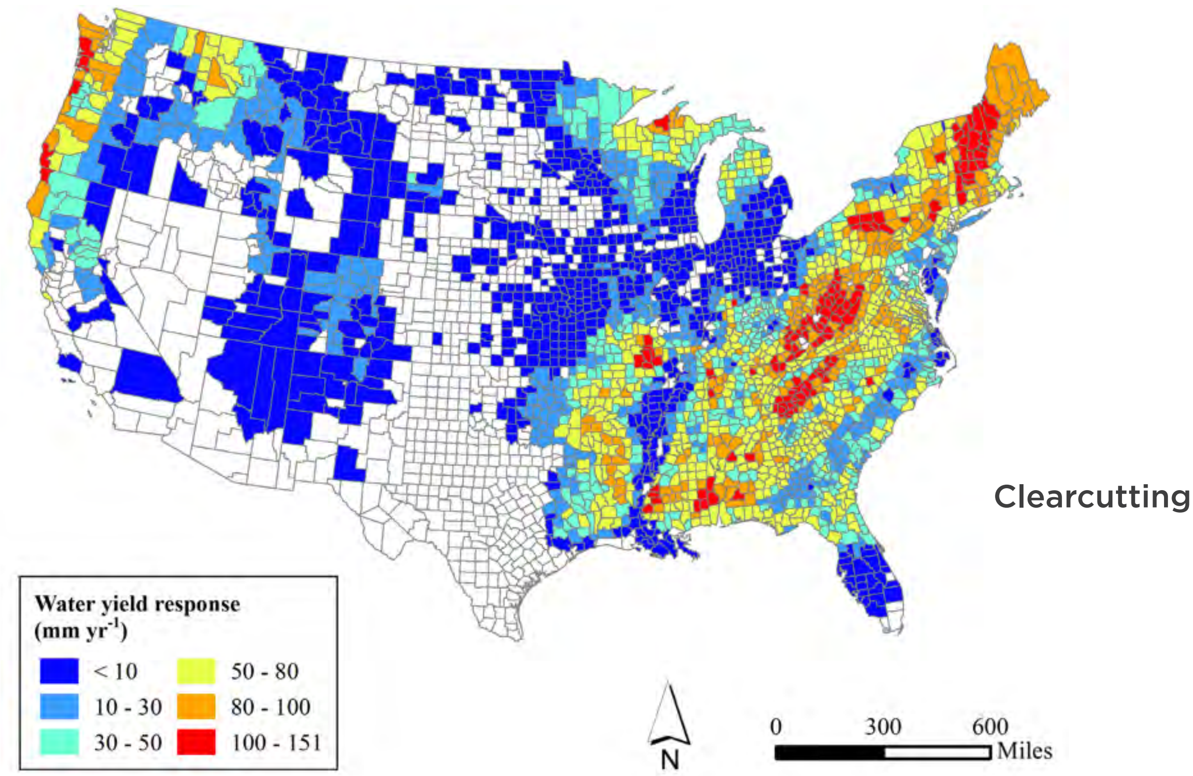


Figure 7.5 | WaSSI modeled maximum response of mean annual water yield to A, clearcutting and B, thinning by county across the United States

A.



B.

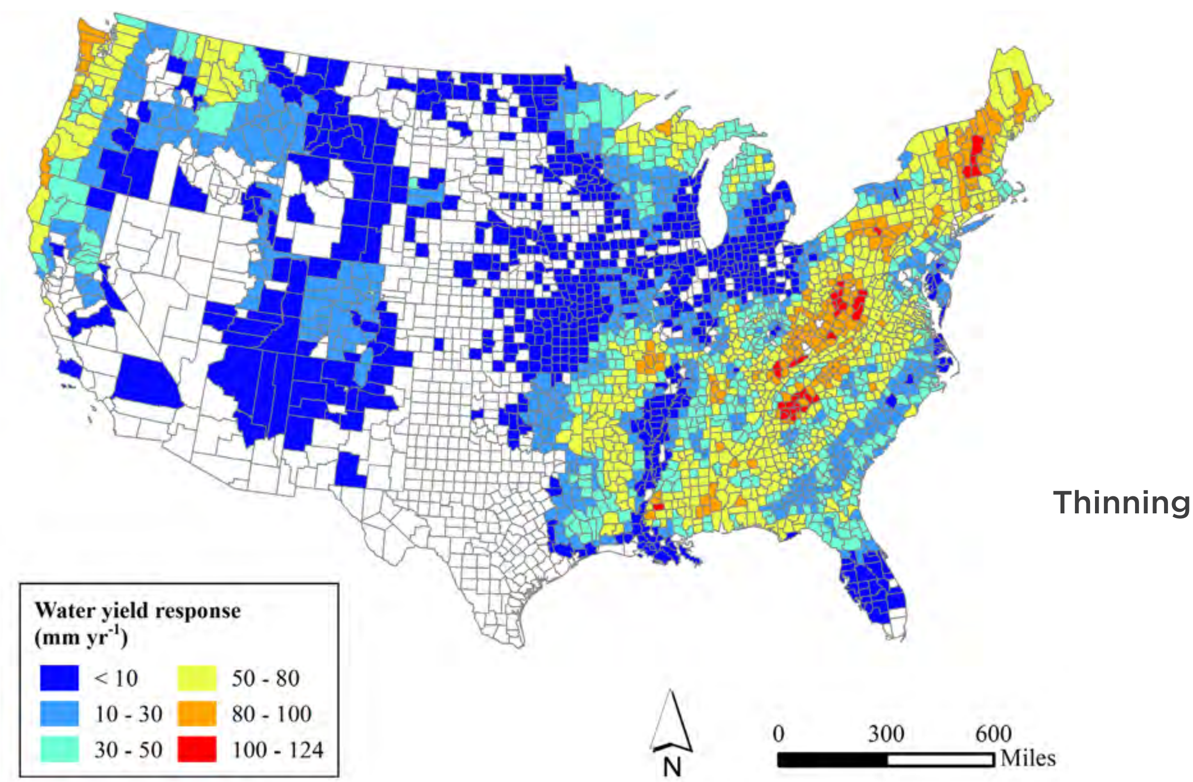
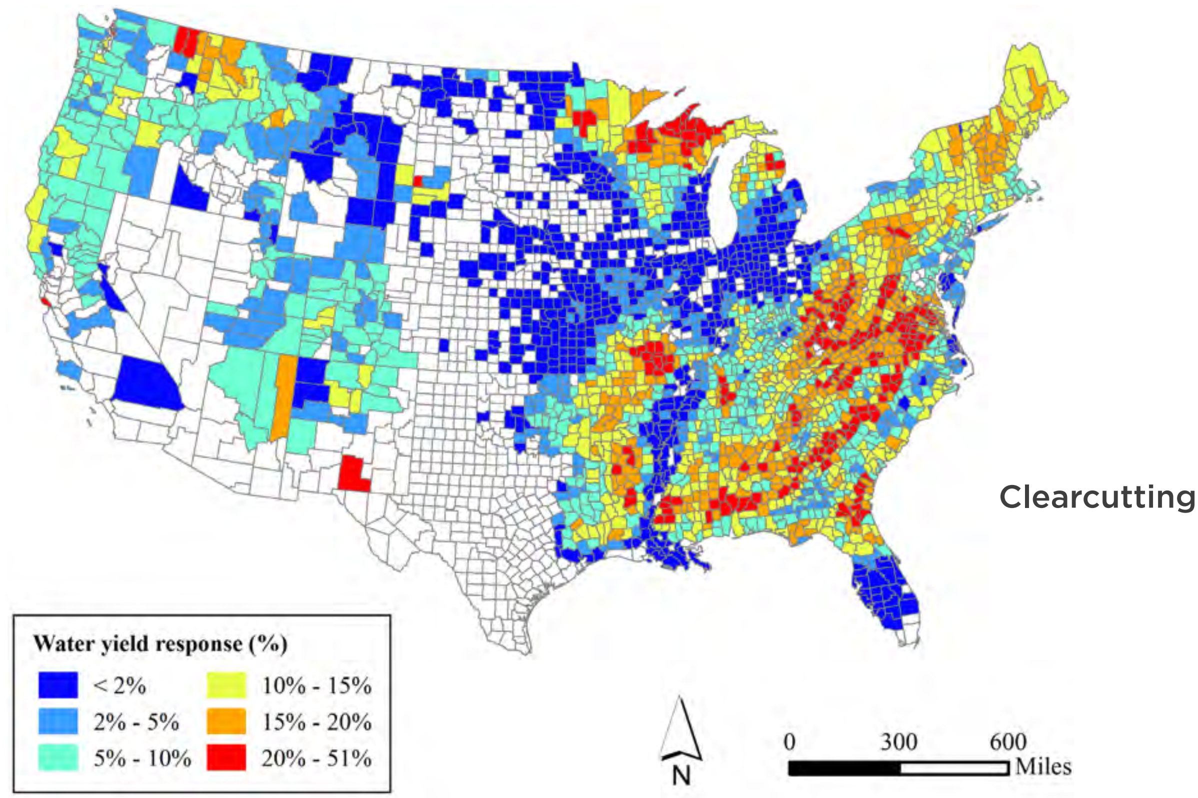
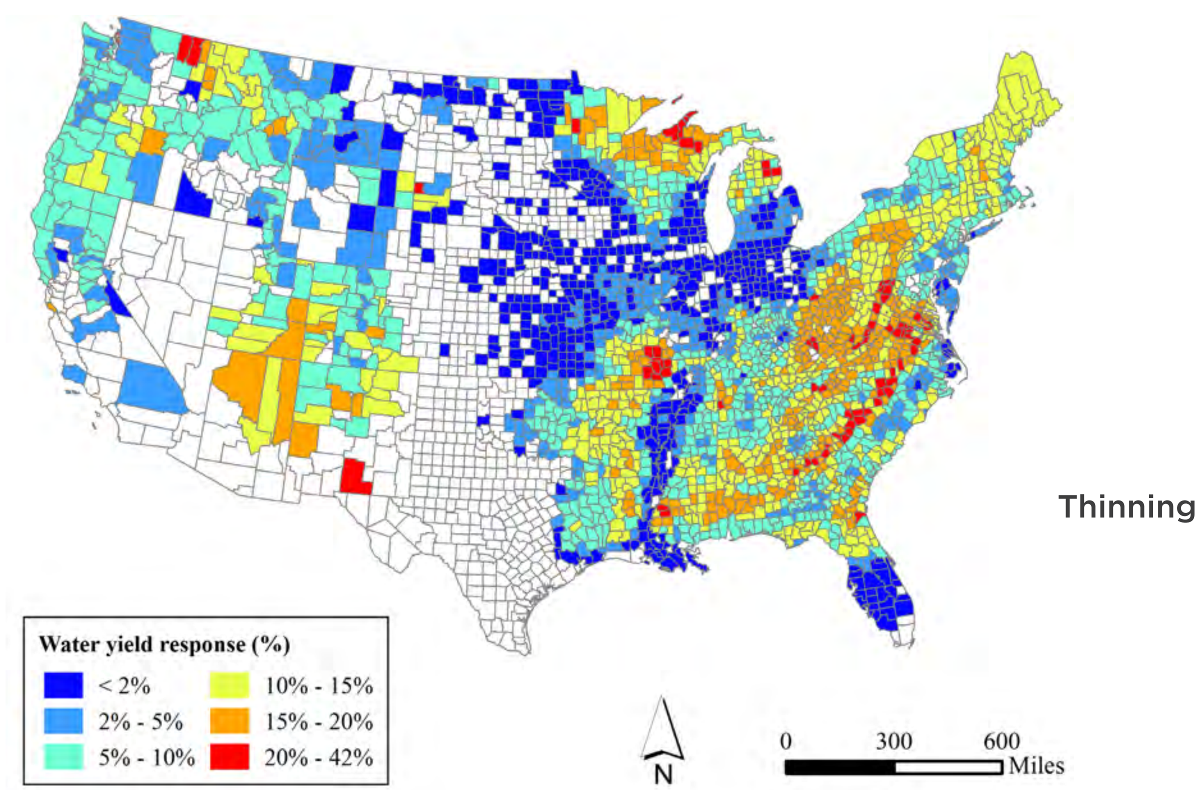


Figure 7.6 | WaSSI modeled maximum relative response of mean annual water yield to A, clearcutting and B, thinning by county across the United States

A.



B.



To normalize the hydrological response to forest removal, the water yield response can also be expressed as relative change by the following formula. The long-term mean water yield is the reference condition:

(water yield under harvesting – long-term mean water yield)/long-term mean water yield

Relative changes in water yield compared to the reference condition (fig. 7.6) show different spatial patterns from those for the absolute water yield response. For example, areas that have low absolute water yield response in the arid Midwest or the Lower Coastal Plains in the humid Southeast show a relatively large change in water yield, while the regions with high absolute water yield, such as the wet Pacific Northwest (<10%) and the Northeast (<20%), have low relative response. The Piedmont region in the Southeast also shows high relative hydrological response to forest harvesting compared to the reference condition, as high as 50% greater water yield.

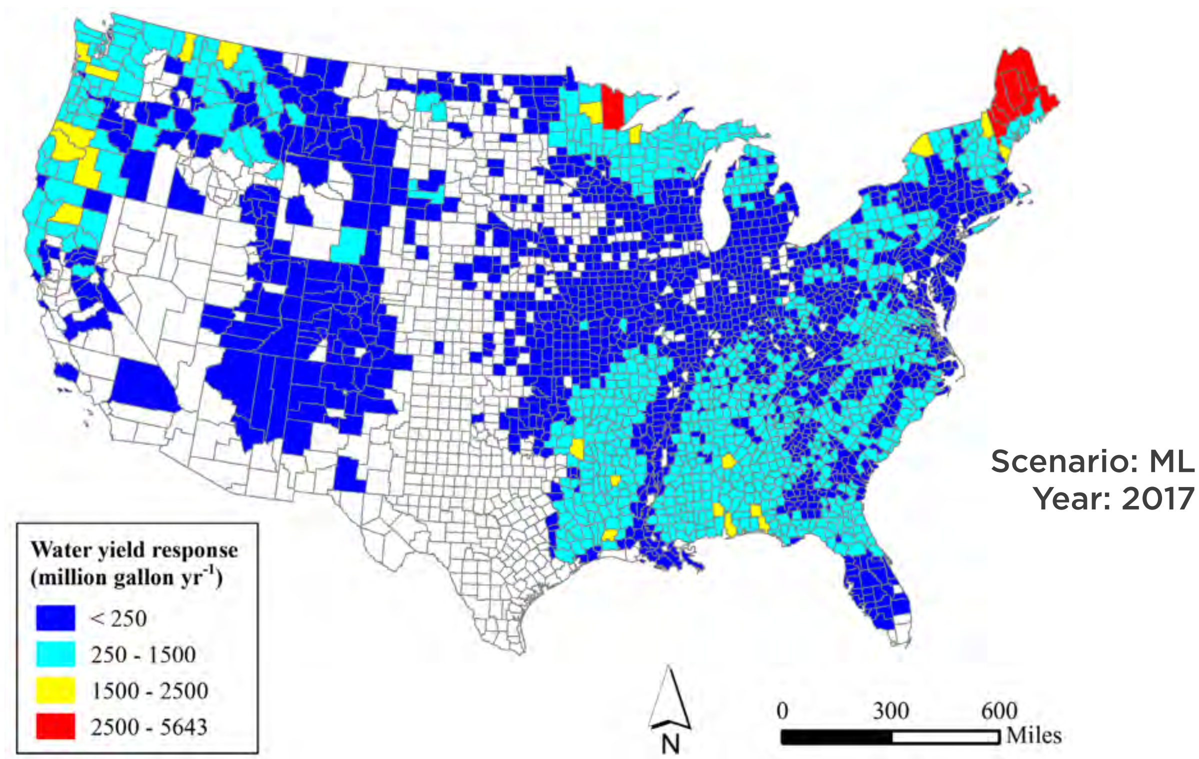
7.3.2 Impacts of Forest Removal on Water Yield by County under Three Scenarios

7.3.2.1 Baseline Case ML 2017

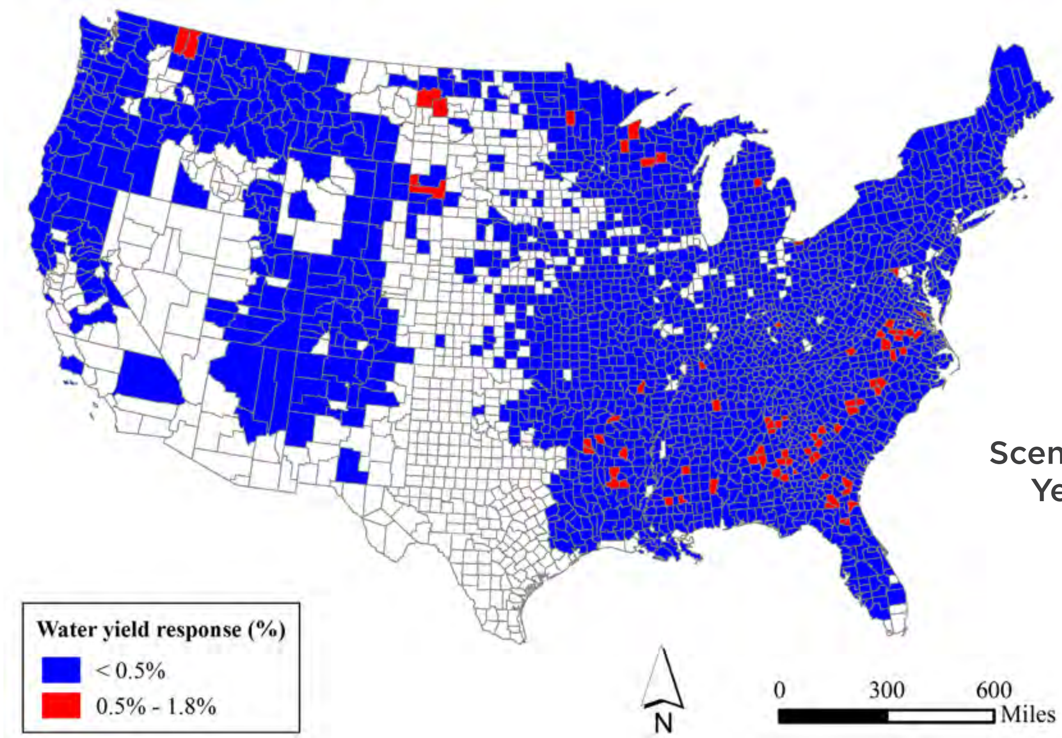
The projected water yield response to harvesting at the county level in the BT16 scenarios was presented as absolute changes in million gallons (fig. 7.7A) and relative changes in percentages (fig. 7.7B). Counties with highest responses (>2,500 million gallons) were found in the high water yield regions in Maine, Minnesota, and Oregon. The relative responses at the county level were rather small (<1.8%) when compared to total water yield of the reference. As discussed earlier, the projected water yield response in the scenarios is controlled by the amount of forest removal, the local hydrological regimes, and the maximum potential water yield response presented in figure 7.5. A majority of the counties had water yield responses of less than 1,500 million gallons per year, or 0.5% of annual water yield.

Figure 7.7 | WaSSI modeled projected response of mean annual water yield to reference under the ML 2017 harvesting scenario across the United States showing A, absolute response in million gallons per year and B, relative response by percentage

A.



B.



Scenario: ML
Year: 2017

This analysis identified 10 counties that show the highest percentage increases in water yield under the ML 2017 scenario (table 7.2). The maximum relative responses of these counties if the entire forest area in the county were harvested vary from 9% to 153%. These counties are located in Maine, Minnesota, Oregon, and Oklahoma in areas that are heavily forested with high runoff (>450 mm per year). St. Louis County in Minnesota is the exception, as runoff is lower (266 mm per year)r and there is extensive biomass removal (1%–2.6%). The largest absolute water yield response was found in Aroostook County in Maine. Nonetheless, the county's 5,643 million gallons per year increase in water yield was considered rather small, representing only 0.2% of the water yield.

The 10 counties that are projected to have the highest relative water yield response (0.8%–1.7%) to biomass harvesting in ML 2017 are listed in table 7.3. These counties are found in both dry (e.g., North Dakota) and wet areas (e.g., North Carolina). The hydrological response was considered to be rather small as a relative water yield, compared to the reference. The maximum relative responses of these counties if the entire forest area in the county were harvested are also presented.

Table 7.2 | The 10 Counties That Have the Highest Water Yield Response (Million Gallons per Year) to Forest Biomass Removals under the Baseline ML 2017 Scenario

County	State	Projected Water Yield Response		Precip-itation (mm/ year^{-1})	Runoff (mm/ year^{-1})	Harvest Area		Maximum Potential Water Yield Response					
		(million gallon/ year)	(%)			(km^2)	(%)	(mm/year^{-1})		(billion gallons /year)		(%)	
								Clear-cutting	Thin-ning	Clear-cutting	Thin-ning	Clear-cutting	Thin-ning
Aroostook	Maine	5,643	0.2	1,044	592	229	1.4	87	73	403	339	15	12
Somerset	Maine	3,798	0.2	1,143	672	152	1.5	90	75	254	212	13	11
Piscataquis	Maine	3,786	0.2	1,130	657	142	1.4	93	78	277	231	14	12
Oxford	Maine	3,529	0.3	1,229	729	124	2.4	111	92	162	135	15	13
Penobscot	Maine	3,256	0.2	1,130	634	120	1.4	97	81	236	196	15	13
Washington	Maine	3,051	0.2	1,227	729	119	1.8	93	77	171	142	13	11
Franklin	Maine	2,891	0.3	1,229	763	108	2.5	104	88	125	105	14	11
St. Louis	Minnesota	2,890	0.2	705	266	252	1.7	39	34	178	155	15	13
Douglas	Oregon	2,376	0.1	1,361	700	116	1.0	72	60	250	209	10	9
McCurtain	Oklahoma	2,069	0.3	1,299	482	76	2.6	65	55	85	72	13	11

Table 7.3 | The 10 Counties That Have the Highest Relative Water Yield Response (%) to Forest Biomass Removals under the Baseline ML 2017 Scenario

County	State	Projected Water Yield Response		Precipitation (mm/year)	Runoff (mm/year)	Harvest area		Maximum Potential Water Yield Response					
		(million gallons/year)	(%)			(km^2)	(%)	(mm/year)		(billion gallons/year)		(%)	
								Clear-cutting	Thinning	Clear-cutting	Thinning	Clear-cutting	Thinning
Dunn	North Dakota	1,078	1.7	424	44	5	68.2	1	1	1	2	1	3
Middlesex	Virginia	379	1.3	1,170	339	20	8.7	52	43	4	4	15	13
Fairfield	South Carolina	1,379	1.2	1,084	245	61	3.5	93	72	45	35	38	29
Lancaster	South Carolina	864	1.0	1,082	246	43	3.2	85	71	30	25	35	29
Warren	North Carolina	964	1.0	1,125	320	44	4.3	78	62	24	19	24	19
Erie	Ohio	649	0.9	922	413	28	62.4	6	8	1	1	1	2
Brantley	Georgia	888	0.9	1,291	312	45	4.1	76	58	24	18	24	19
Lawrence	South Dakota	831	0.9	620	178	57	3.7	41	40	23	22	23	22
Echols	Georgia	687	0.8	1,288	280	39	3.7	72	54	21	16	26	19
Marshall	Kentucky	746	0.8	1,254	440	25	9.0	38	36	8	8	9	8

7.3.2.2 Baseline Case ML 2040

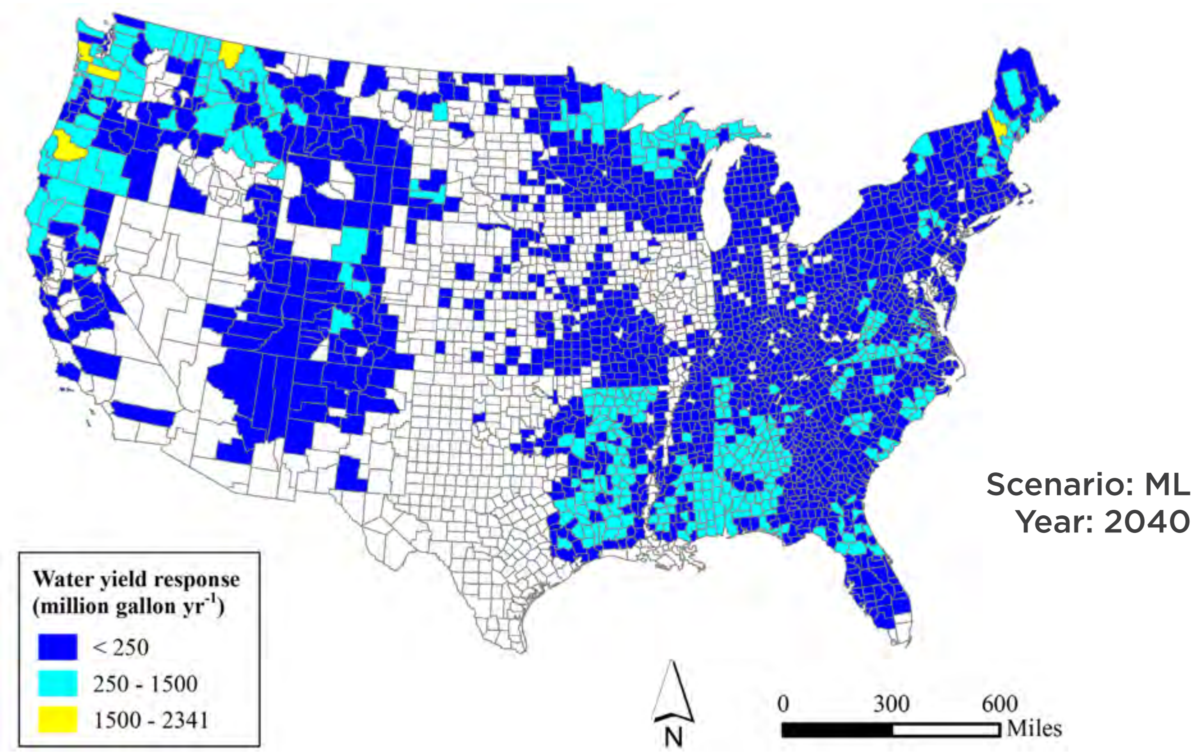
Compared to the water yield response under the baseline ML 2017 scenario, the water yield response under the ML 2040 scenario was found to be even smaller in both absolute and relative terms. A majority of the counties have annual water yield increases of less than 250 million gallons or 0.5% of background water yield (fig. 7.8). The decreased water yield response is due to the reduced forest harvesting area in 2040 as compared to 2017 (figures 7.2A and 7.2B).

7.3.2.3 High Yield Case HH 2040

Similar to the ML 2040 scenario, a majority of the county-level water yield responses under the HH 2040 scenario are less than 250 million gallons per year or 0.5% of background water yield (fig. 7.9). This scenario represents the lowest impacts on water yield among the three scenarios.

Figure 7.8 | WaSSI modeled projected response of county-level mean annual water yield to under the ML 2040 harvesting scenario across the United States, showing A, absolute response in million gallons per year and B, relative response in percentage change from reference conditions

A.



B.

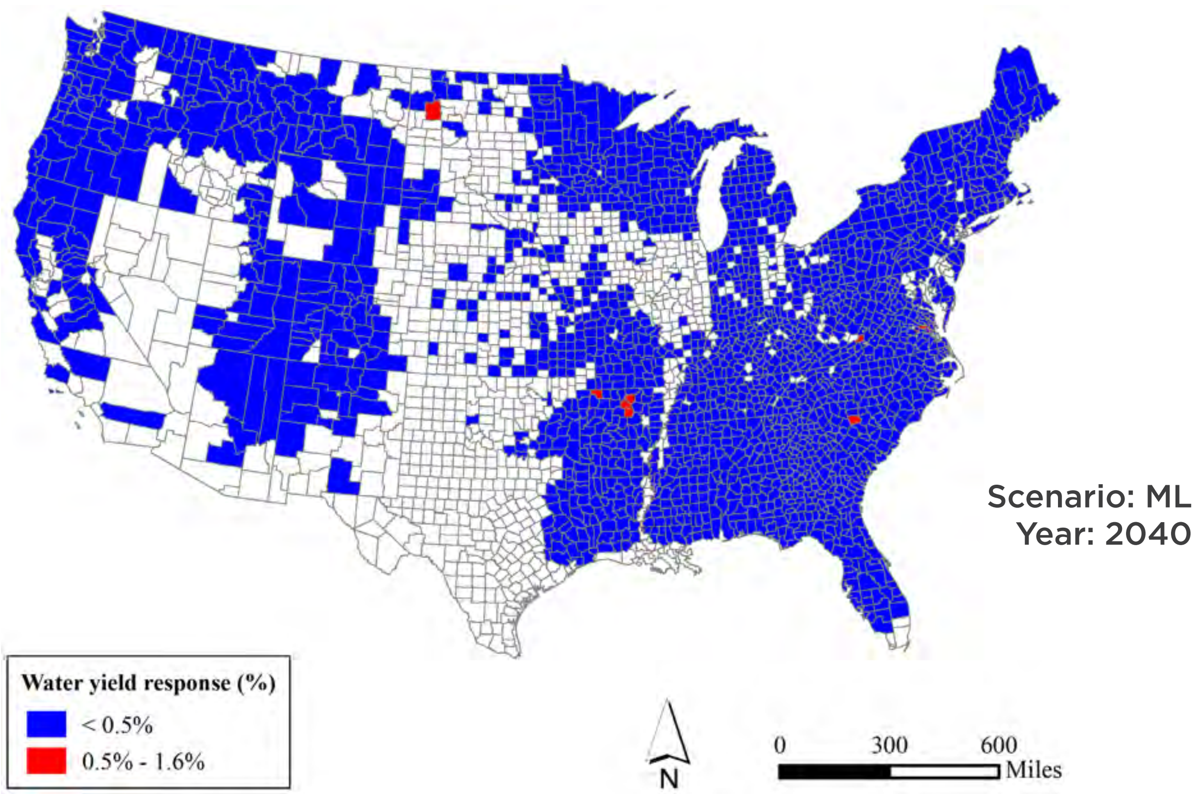
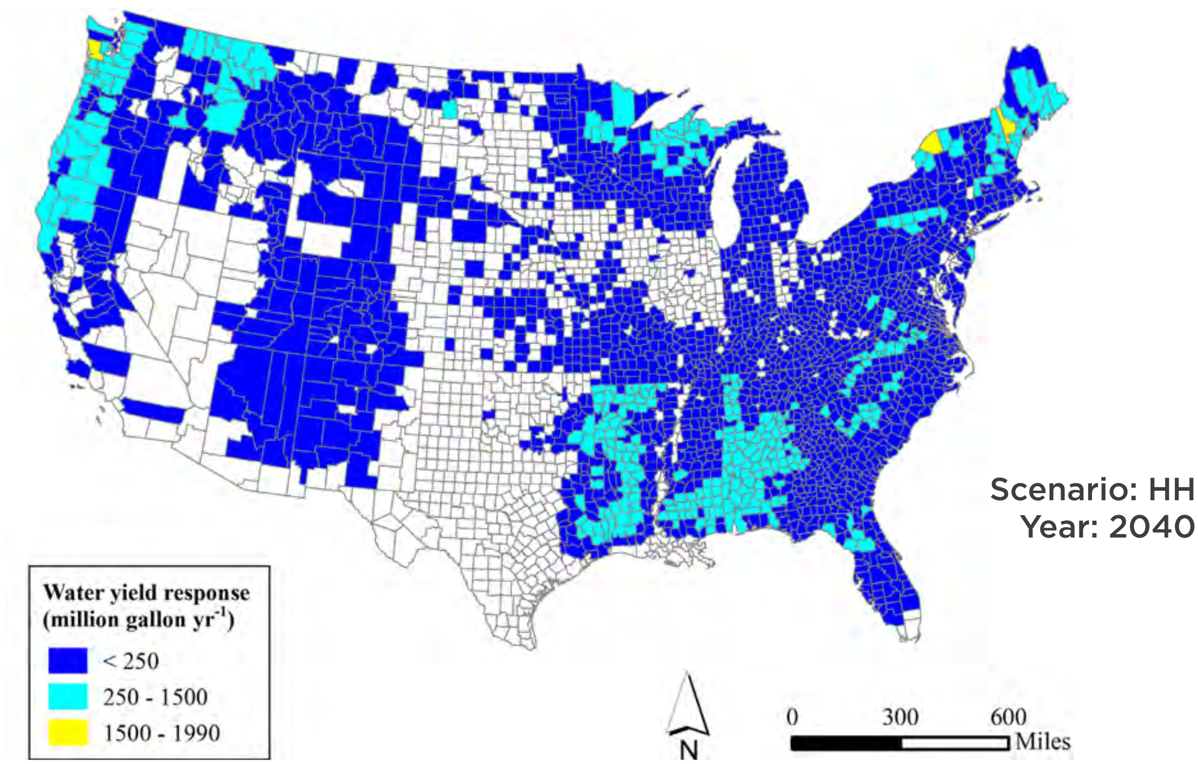
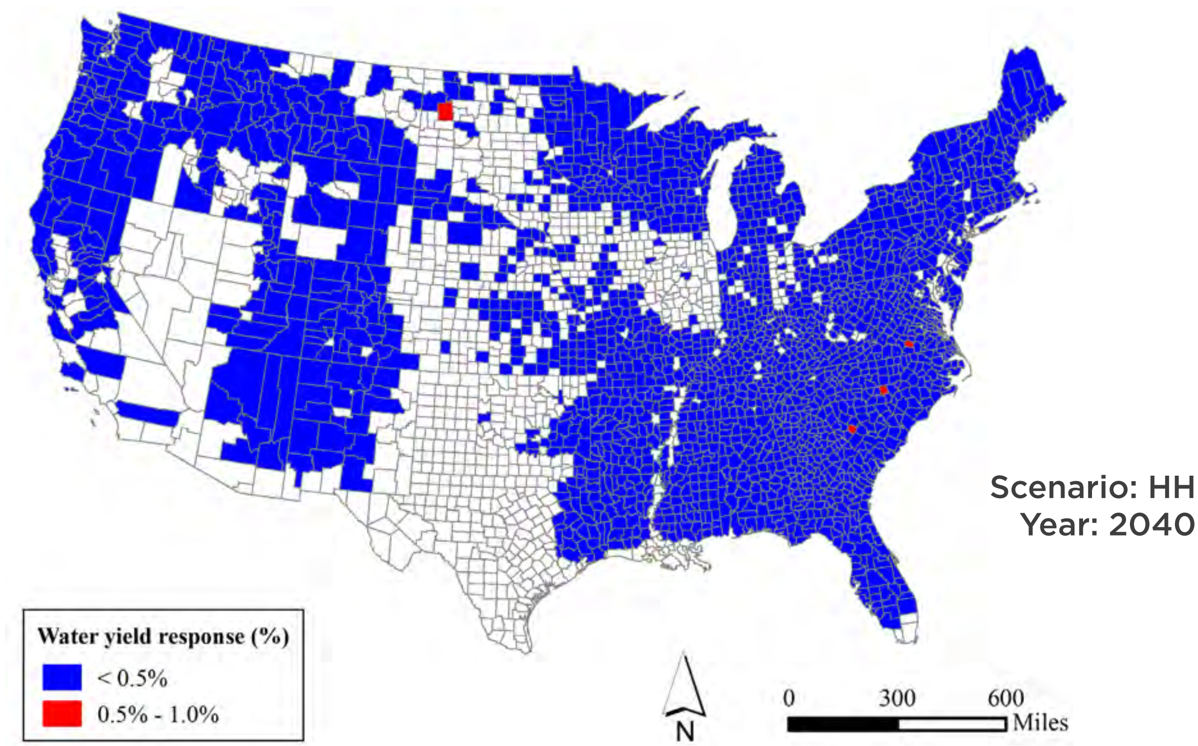


Figure 7.9 | WaSSI modeled projected response of county-level mean annual water yield under the HH 2040 harvesting scenario across the United States, showing A, absolute response in million gallons per year and B, relative response as a percentage change from reference conditions

A.



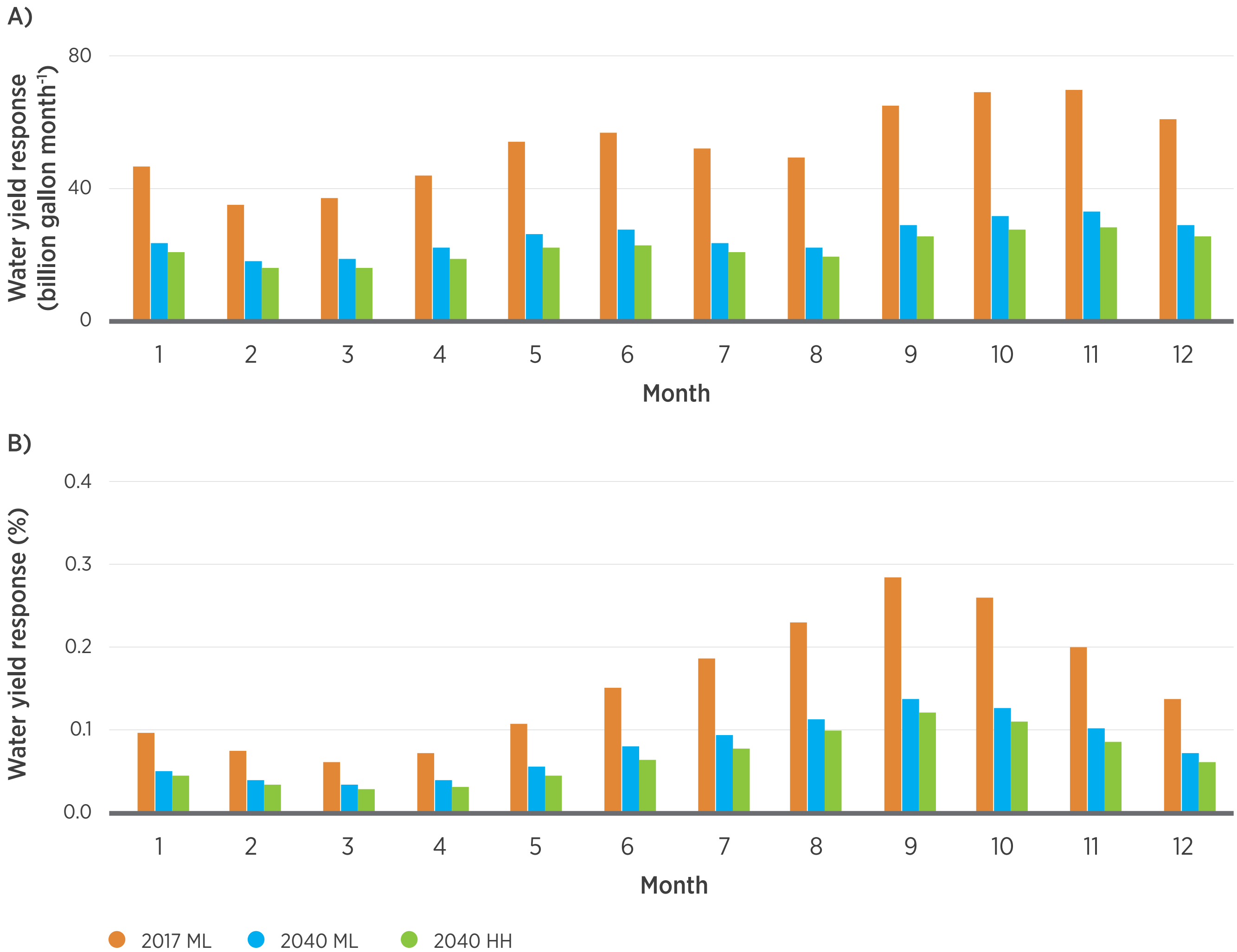
B.



7.3.3 Seasonal Response to Biomass Removal

Effects of different harvesting scenarios on water yield vary by scenario as well as by season (fig. 7.10). Figure 7.10A shows that biomass removal in 2017 has a much higher impact (>two times) on water yield than it does in 2040 at both harvesting levels. In general, the absolute water yield responses vary little seasonally, showing a uniform pattern (fig. 7.10A), while the relative changes peak during the fall season, when streamflow is the lowest in most of the U.S. watersheds (fig. 7.10B).

Figure 7.10 | WaSSI modeled response of seasonal water yield to three harvesting scenarios across the United States, showing A, total absolute response in billion gallons per year and B, relative response as a percentage

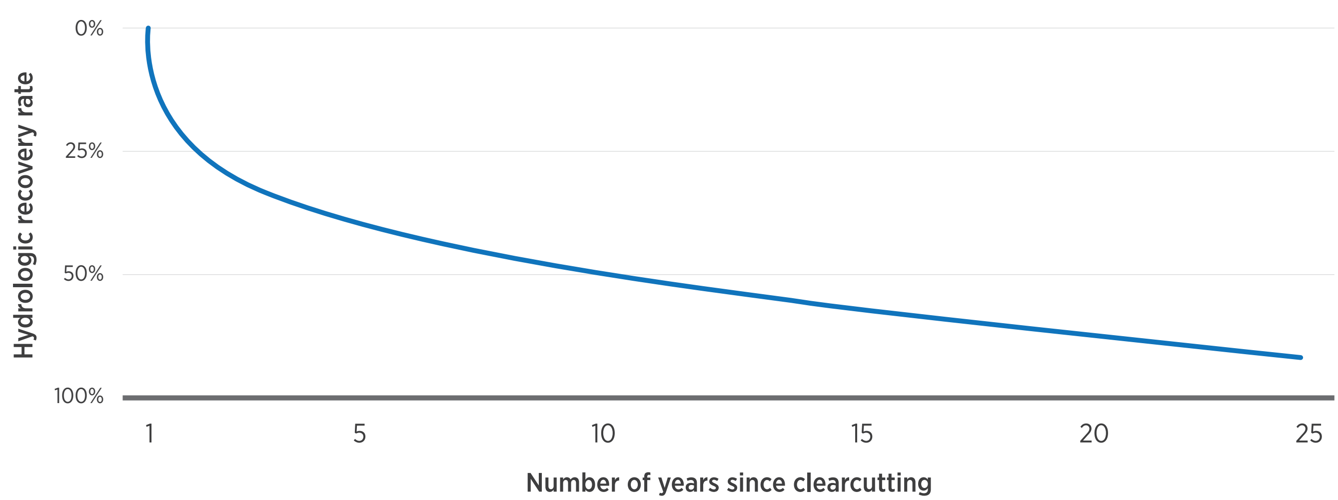


7.4 Discussion

This study applied a watershed water balance model, WaSSI, to estimate seasonal and mean annual hydrological responses to three scenarios of biomass removals. Water yield changes are expressed at the county level, since biomass harvesting data are reported at that spatial scale. Removal of forests by clearcutting or severe thinning (70% reduction in LAI) has the potential to increase water quantity up to 50% in some regions. However, because the cutting areas are relatively small (<5%) when compared to the total forestlands at the county scale, this study projects that the hydrologic responses would be rather minor in the three biomass removal scenarios. The simulation results are consistent with the empirical notion that removing less than 10% of forest cover in a watershed does not have measurable impacts on streamflow.

Harvesting impacts presented in this study represent the immediate annual responses of water yield to forest clearcutting or thinning, or the maximum water supply change at the county scale. Since trees are likely replanted or would regenerate naturally, water yield impacts in subsequent time periods would gradually decrease while total forest ET rates increase (Ford et al. 2011). Depending on local climatic and vegetation characteristics, the hydrology of disturbed watersheds may recover within a few years to decades in the United States. For example, a watershed dominated by deciduous hardwoods in the southern Appalachians can recover to pre-disturbance levels 5–10 years after clearcutting. Similarly, clearcutting loblolly pine plantations can increase drainage up to 50%, but the increase of water may diminish after 10 years of replanting {Sun, 2004 #1661}. However, it may take over 50 years for forests in areas with low growth rates, such as the Rocky Mountains, to recover their hydrology. Fig. 7.11 presents a hydrologic recovery curve developed from experimental data at the Coweeta Hydrologic Laboratory in North Carolina to illustrate that water yield response to forest harvesting decreases over time. In this case, more than 85% of the initial increase in water yield (about 350 mm per year) diminishes by year 25 after the watershed was clearcut and trees are regenerated (fig. 7.11).

Figure 7.11 | A hydrological recovery curve for a watershed dominated by deciduous hardwood forests in the southern Appalachians, showing that the initial water yield increase due to forest clearcutting diminishes over time as a result of tree regrowth and associated increase in ET (Sun et al. 2004)



7.4.1 Implications of Modeling Results

The baseline 2017 biomass harvesting scenario (ML 2017) represents the largest hydrological disturbance related to forest biomass–based energy development. However, this study suggests that the projected biomass removal levels are rather low and may not cause concerns or large benefits to water quantity and water resources at the county scale. It is important to note that although the hydrological effects are negligible at the county level, the impacts can be significant if the biomass harvesting activities are concentrated within a watershed in a county. In such a case, forest removals may increase stormflow volume, potentially causing water quality concerns. Forest best management practices such as implementing forest riparian buffers may be effective to mitigate negative harvesting effects on stream hydrology and water quality (Cristan et al. 2016). Geographically, forest biomass removals may have fewer environmental issues in areas with a flat topography and vegetation that recovers quickly.

7.4.2 Uncertainties and Limitations

This study took a top-down approach in modeling the likely impacts of forest biomass removal on water quantity at the county level rather than a bottom-up approach that examines hydrological processes in forests in a spatially explicit manner. Although the WaSSI model considers the effects of climate, soil, and forest structure (LAI) on water balances at the watershed scale within a county, the simulated water yield responses by WaSSI represent a mean condition. Errors may occur as a result of not knowing the exact location that biomass removal activities would occur. Localized forest harvesting may have much higher impacts on the hydrology in certain watersheds than at the county level. In addition, the water balance component of the WaSSI model was developed using ecohydrological data for multiple ecosystems and has been used to understand impacts of forest thinning, but results have not been thoroughly verified, specifically under forest disturbance conditions, because of the lack of experimental data.

This analysis used long-term (1991-2001) mean climate to simulate the hydrological effects of forest cover change and assumed that the climate in 2040 would remain the same as in 2017 (e.g., the historic conditions). Recent studies suggest that by 2040 the climate may be much warmer, and water yield is expected to decrease because of the rise of ET (Sun et al. 2015a; Duan et al. 2016). Thus, forest biomass harvesting in 2040 is expected to have more pronounced effects in terms of relative change in water yield in most regions across the United States.

7.5 Summary and Future Research

The amount and distribution of forest live biomass is closely related to water yield and water supply, one of the important ecological functions and services of forest ecosystems. Biomass harvesting has the potential to alter water quantity by altering ecohydrological processes (ecosystem ET in particular).

This analysis applied a monthly watershed hydrological model, WaSSI, to the 88,000 HUC 12 watersheds and quantified how three select BT16 forest-harvesting scenarios affect mean seasonal and annual water yield at the county level. The research shows that all scenarios would have minor impacts on water quantity at the county level because of the small areas of harvesting (<5%) in most counties. The small magnitude of hydrological response (<2%) to biomass removal may not have much significance, positive or negative, in terms of water supply at the county level. However, it is important to note that concentrated biomass-removal activities may cause substantial local impacts on watershed hydrology. Unfortunately, current projections of biomass harvesting do not provide the spatial information sufficient for watershed-scale assessment, and therefore, the study presented here only shows county-level water yield responses. Research is needed to model biomass removal at finer spatial scales, such as a watershed rather than a county.

This analysis assessed water yield impacts on an annual basis; however, hydrological and environmental impacts are cumulative. Future studies should examine the cumulative effects of forest biomass removal in specific watersheds where harvesting activities are expected to occur. This study only examined total water yield, without looking at other hydrologic parameters, such as base flow and peak flow rates. Future watershed-scale studies should focus on ecologically relevant indicators of streamflow. In addition, future studies should link water quantity and quality to allow for a comprehensive assessment of water resources at the watershed to county levels.

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08 | Water Consumption Footprint of Producing Agriculture and Forestry Feedstocks

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8.1 Introduction

The management of our nation's water resources faces increasingly pressing challenges that are exacerbated by an expanding population, growing energy demands, and a changing climate. To build a sustainable water future, crosscutting, innovative strategies are needed (White House 2016). A recent SECURE Water Act Section 9503(c) report identifies warmer temperatures, changes in precipitation, decreasing snowpack, and the timing and quality of streamflow runoff across major river basins as threats to water availability (DOI 2016). The U.S. Department of Energy (DOE) identified water use and water resources as critical components of environmental sustainability to be addressed in bioenergy development (DOE 2016a).

As with any biological system, the production of bioenergy feedstocks relies on water, as well as soil, climate, and other environmental variables. Water use in bioenergy production varies extensively by feedstock and region (Phong, Kumar, and Drewery 2011; Georgescu, Lobell, and Field 2011; Wu et al. 2009; Evans and Cohen 2009). Industrial development, however, can significantly affect the availability of water resources (Schuol et al. 2008; Faramarzi et al. 2009; Glavan, Pintar, and Volk 2012). From an economic perspective, the value of water varies from one location to another, depending on the richness of water resources in that vicinity (Frederick, VandenBerg, and Hanson 1995). Hoekstra and Hung (2005) analyzed water intensity across the supply chain and from production system to use communities. In addition, the different priorities for water use, both regionally and across time, result in economic and environmental trade-offs that must be identified and addressed (Williams and Al-Hmoud 2015). Variations in stressors (e.g., drought, competing water use) associated with water supply and consumption among regions could result in substantial impacts on energy production, and the ripple effect of these stressors can be felt across regions in multiple sectors (Fulton and Cooley 2015; Heberger and Donnelly 2015; Scown, Horvath, and McKone 2011). Historically, irrigation has been a major factor in the water footprint of conventional bioenergy and agricultural products because of the demand from annual crops in certain regions (White and Yen 2015; Chiu and Wu 2012; Scanlon et al. 2012; Gerbens-Leenes, Hoekstra, and van der Meer 2009).

Irrigation accounts for about 80% of water demand globally; if it is not appropriately managed, irrigation could have significant effects on the global water system (Rost et al. 2008). A recent report showed that the rate of groundwater depletion has increased markedly since about 1950, with maximum rates of depletion occurring during the most recent period (2000–2008) (Konikow 2013). Improvements in technology and irrigation practices can impact water use substantially (Levidow et al. 2014; Cooley, Gleick, and Christian-Smith 2009).

Evapotranspiration is the sum of evaporation from the land and water surface and plant transpiration to the atmosphere. Research indicates transpiration is the larger component of evapotranspiration (ET) (Jasechko et al. 2013). Transpiration accounts for the movement of water within a plant and the subsequent loss of water as vapor through stomata in its leaves. Evapotranspiration is an important part of the water cycle.

Despite the facts that all biomass requires water and that the water demand is to be met by either rainfall or irrigation, some biomass, such as perennial grasses, can grow without irrigation or with significantly less irrigation than other crops in some regions. The long root system of perennial grasses is able to retrieve moisture from deep soil, which can also benefit water quality (chapter 4). However, biomass feedstock yields depend heavily on regional soil and climate conditions, so no single type of crop is an appropriate feedstock for the entire United States. A regional feedstock portfolio that provides high yield while demanding less irrigation would be ideal.

The *2016 Billion-Ton Report* (*BT16*) presents scenarios of a gradual transition from the current biomass feedstock-production system to a future feedstock mix. It focuses on the production of non-food, high-yield cellulosic energy crops (DOE 2016b). The current U.S. energy portfolio could be further diversified by increasing the share of bioenergy; this would improve energy security as mandated by Congress in the Energy Policy Act of 2005, which was expanded under the Energy Independence and Security Act of 2007 (Pub. L. 110-140 2007).

The objectives of this chapter are (1) to develop an estimate of water consumption for major potential *BT16* production scenarios and (2) to conduct geospatial analysis to examine the interplay between feedstock mix and water consumption, as well as geospatial patterns of water consumption footprints for different feedstock mixes. A further aim is to support planning for future regional water resources at federal and local levels. Water footprint analysis considers consumptive water use for biomass production, representing water resource demand and geospatial trends for future scenarios. Water footprint analysis highlights the impact a future scenario would have on water demand at the national scale and, in this case, provides county-level details—a key issue in natural resource availability. Water consumption is particularly relevant when analyzing regional water scarcity and the impact of human activities on water availability.

This assessment focuses on the water consumption aspect of water use. Whereas the term "water use" sometimes refers to water withdrawal, here we designate water use to refer to water consumption. Thus, the terms "water use," "consumptive water use," and "plant water requirement" that appear in this chapter all refer to water consumption by feedstocks in their growing stage. In addition, this study calculates the rainwater demand of all feedstocks and irrigation water demand of conventional crops, not the actual irrigation water volume withdrawn. By definition, water consumption in feedstock production represents the quantity of water that is (1) removed from a defined system via ET and (2) not immediately returned to the original water source.

This work builds upon previous studies (Wu, Zhang, and Chiu 2014; Chiu and Wu 2012, 2013; Wu and Chiu 2014) on the geospatially explicit water footprint of bioenergy feedstock production in the United States, as well as related model development (Wu et al. 2015; ANL 2013). The chapter examines the water resource requirements of select *BT16* scenarios by conducting a geospatial analysis and estimating the water consumption footprint at three scales: county, state, and national (at a regional resolution). Changes and the distribution of water consumption are analyzed. These results can improve our understanding of the implications that transitioning to cellulosic biomass production would have on regional water use and highlight regional characteristics under the scenarios, thereby aiding the planning and development of new bioenergy and other biomass projects.

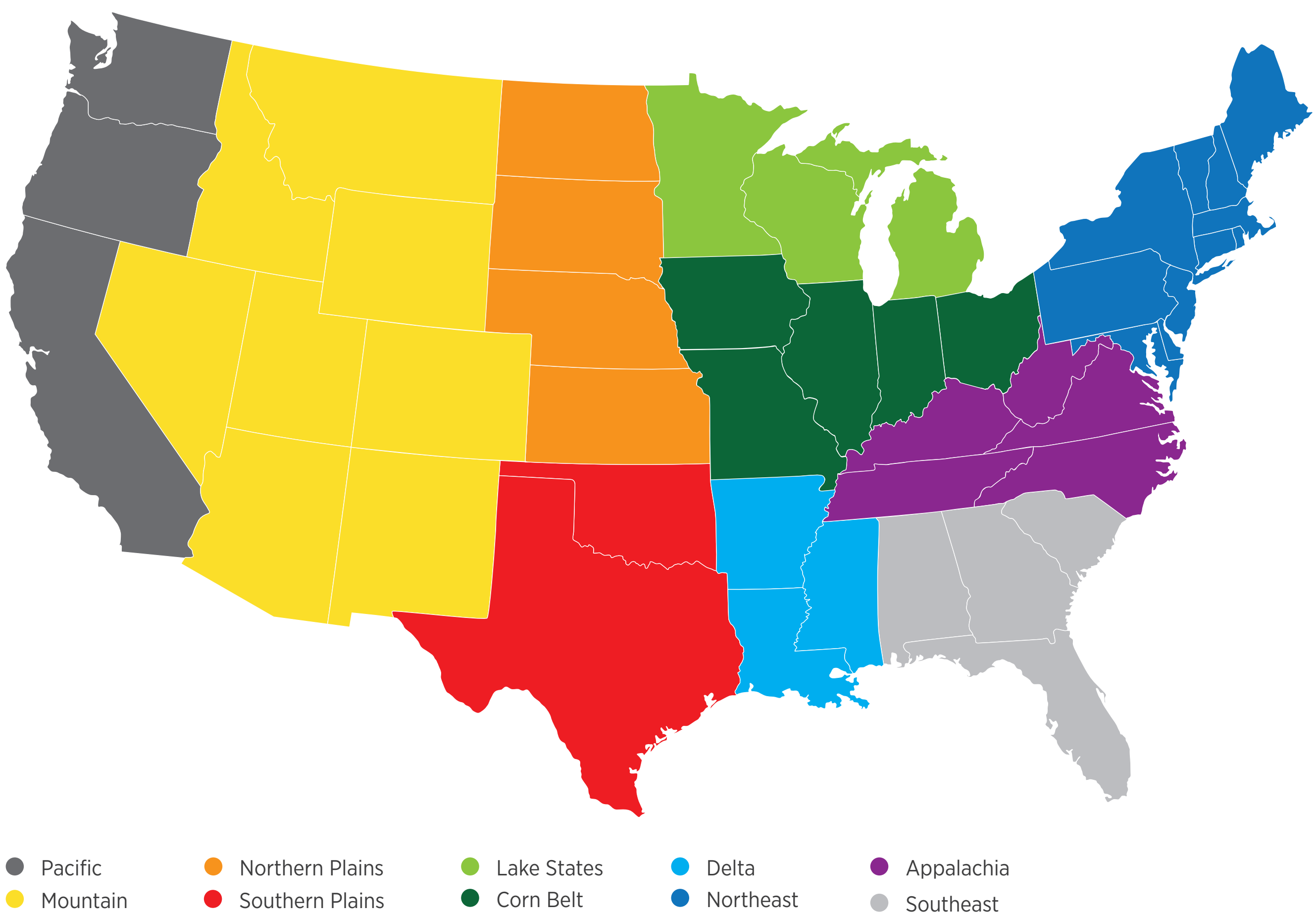
8.2 Methods

8.2.1 Scope of Assessment

A water footprint is developed for the selected *BT16* feedstock production scenarios at county-level resolution for the conterminous United States. The study analyzes select biomass feedstocks, including the following: corn grain (the portion used for ethanol), corn stover, soybean (the portion used for biodiesel), wheat straw, switchgrass, *Miscanthus* × *giganteus*, short-rotation woody crop (SRWCs) (willow, hybrid-poplar, and southern pine), and resources from softwood and hardwood forest stands. Other energy crops and municipal solid waste (MSW) are not included, either because they are in the early stages of development or because complete county-level data are lacking. (A qualitative analysis of water consumption in *BT16* microalgae scenarios is included in chapter 12.) The analysis does not include food crops or plants that serve non-bioenergy purposes. The term "biomass" designates feedstock produced for bioenergy or other purposes, which is all of the feedstock analyzed in this chapter.

Crops receive water from either precipitation or irrigation. In this study, irrigation of conventional crops (e.g., corn and soybeans) is attributed to corn grain and soybeans. Energy crops (e.g., perennial grasses, SRWCs) are assumed to be rain-fed. Water withdrawn and applied for irrigation can be used by crops, contributes to runoff to streams, or percolates into the soil. The water footprint analysis accounts for consumptive water use by crops. In this chapter, we define rainwater stored in the soil or intercepted by the plant and subsequently used in plant growth as "rainfall" and consumptive irrigation water requirements as "irrigation." This analysis does not account for irrigation efficiency due to irrigation technology differences or biorefinery water use. The water footprint analysis is conducted at the county, state, and regional levels. Figure 8.1 depicts the agriculture resource regions for the United States analyzed in this chapter. This chapter differs from chapter 7 in that this chapter addresses the water footprint in producing feedstock from annual crops, perennial grasses and SRWCs, and residues and whole trees from forests, whereas chapter 7 examines the impacts that removing feedstocks from the forest would have on water yield in the forestland. Water quality is described in chapters 5 and 6 of this report.

Figure 8.1 | Biomass feedstock production regions for this analysis



8.2.2 Scenarios

This chapter analyzes the water consumption that may be associated with realizing the potential biomass availability scenarios from *BT16* volume 1, all assuming a roadside price of up to $60 per dry ton (see executive summary, fig ES.1). Six agricultural and forest scenarios were selected for this study: BC1 2017, ML 2017, BC1 2040, ML 2040, HH 2040, and HH3 2040 (see chapter 2). Each scenario covers a different feedstock mix and production year at the feedstock price of $60 per dry short ton,[1] representing current and future biomass potential. Scenario BC1 2017 represents feedstock harvested from current annual crops for which the yield increases at an annual rate of 1%; scenario ML 2017 represents feedstocks collected from forest stands in the form of residues and whole-tree biomass in 2017; scenario BC1 2040 illustrates crop-yield increases at the same rate as that of scenario BC1 2017 with the addition of energy crops (perennial grasses and SRWCs); scenario ML2040 represents a scenario where a slightly decreased quantity of forest resources is available as feedstock; scenario HH2040 is a future scenario where feedstock potentially available from forest resources is further decreased; and scenario HH3 2040 illustrates a simulation in which the agriculture crop yield increases at a 3% annual rate by 2040 (see *BT16* volume 1). Feedstocks included in each scenario are presented in table 8.1, which shows the pairs of agricultural and forestry scenarios in a particular year that were evaluated together for water consumption. Forestry feedstock production under scenarios ML 2017, ML 2040, and HH 2040 are estimated separately from agriculture scenarios (BC1 2017, BC1 2040, and HH3 2040). In the *BT16* volume 1 scenarios, perennial crops (switchgrass, miscanthus, and SRWCs) are not available in 2017; they are available in 2040. The estimated water footprint for the feedstock production scenarios in this chapter reflects that model assumption. Descriptions of the forestry scenarios can be found in chapter 2, 6, and 7 and in more detail in *BT16* volume 1.

Table 8.1 | *BT16* Feedstock Scenarios

Scenario	Feedstock Types				
BC1&ML 2017	Corn stover, wheat straw	Corn grain, soybean			Forest residues and whole-tree biomass (hardwood, softwood)
BC1&ML 2040	Corn stover, wheat straw	Corn grain, soybean	Switchgrass, *Miscanthus × giganteus*	SRWC: willow, hybrid poplar, pine	Forest residues and whole-tree biomass (hardwood, softwood)
HH3&HH 2040	Corn stover, wheat straw	Corn grain, soybean	Switchgrass, *Miscanthus × giganteus*	SRWC: willow, hybrid poplar, pine	Forest residues and whole-tree biomass (hardwood, softwood)

[1] Tons are reported as dry short tons throughout this report, unless specified otherwise.

The forest resource is broken down into residue, saw log, pulp, and whole-tree biomass through clearcut and thinning operations (table 8.2). Only residue and whole-tree biomass are used as feedstock in the *BT16* assessment. The distribution of each feedstock type and harvested feedstock volume is described in *BT16* volume 1.

8.2.3 Description of Water Footprint Accounting for Crops, Grasses, and Forest Resources

Water-footprint accounting has been recognized as a useful method for assessing regional water-resource availability and use for water governance, policy analysis, and planning (White and Yen 2015; Ringersma, Satjes, and Dent 2003; Falkenmark and Rockstrom 2004 and 2006), and it was incorporated into the International Organization for Standardization's (ISO's) standard 14046 for the water footprint (ISO 2014). The concept of water footprint accounting was first introduced by Chapagain and Hoekstra (2004) and Hoekstra and Hung (2005) under the United Nations' Food and Agriculture program. The application of the water footprint in various regions and countries was well documented in peer-reviewed literature (Ayres 2014; Mekonnen and Hoekstra 2011; Liu, Zehnder, and Yang 2009; Staples et al. 2013; Wu, Chiu, and Demissie 2012; Hoekstra et al. 2011; Siebert and Döll 2010; Hoekstra and Chapagain 2007). The central part of the water footprint for bioenergy is feedstock water use. Mekonnen and Hoekstra (2011) used the CROPWAT model (FAO 2013) to simulate consumptive water use for 126 crops based on the Penman-Monteith method. Crop water use was estimated at 0.5° grid scale globally by using the G-Epic model (Liu et al. 2007). Similar approaches were adopted in the Soil and Water Assessment Tool (SWAT)[2] (Williams 1990) and the CENTURY model for plant-soil nutrient cycling.[3]

Various researchers (Gerbens-Leenes, Hoekstra, and van der Meer 2009; Scown, Horvath, and McKone 2011; Chiu and Wu 2012, 2013; Staples et al. 2013) analyzed water footprints for different types

Table 8.2 | Forest Resources Feedstock Categories in Scenarios ML 2017, ML 2040, and HH 2040

Forest Type	Stand Category	Operation	Feedstock Type	
Hardwood	Lowland	Clearcut	Whole tree	Residue
		Thinning	Whole tree	Residue
	Upland	Clearcut	Whole tree	Residue
		Thinning	Whole tree	Residue
Softwood	Natural	Clearcut	Whole tree	Residue
		Thinning	Whole tree	Residue
	Planted	Clearcut	Whole tree	Residue
		Thinning	Whole tree	Residue
Mixedwood		Clearcut	Whole tree	Residue
		Thinning	Whole tree	Residue

[2] See http://swat.tamu.edu.

[3] See http://www.cgd.ucar.edu/vemap/abstracts/CENTURY.html.

of biomass feedstocks (e.g., corn, sugarcane, soybean, wheat, perennial grasses, SRWCs, and forest resources) in the United States. The calculated crop water-use values were verified with measurements gathered by field instrumentation and remote sensing, as well as on the basis of values derived from satellite imagery data (Wu, Chiu, and Demissie 2012). Results indicate that the water footprint methodology closely resembles peak monthly water use by corn in the crop-growing season. A county-level water footprint resource, called the Water Analysis Tool for Energy Resources (WATER) (http://water.es.anl.gov), was recently developed to assess water sustainability of fuels in the United States (Wu et al. 2015).

In this study, we adopt a water footprint approach to assess consumptive water use for various feedstock production scenarios from agriculture and forestry by using the WATER model. The methodologies employed in this chapter are consistent with methods used in the Water Supply Stress Index Model (WaSSI) (chapter 7), SWAT (chapter 5), and CENTURY (chapter 4). Descriptions of water consumption in the growth stage of crops, perennial grasses, and forest resources are presented in appendix 8-A. Consumptive water use is quantified for the production of feedstocks (corn and soybeans, grasses, SRWCs, and forest resources) by estimating ET. Methods for estimating ET that were used in this analysis are described in appendix 8-A. Water footprint is presented as water intensity, which is annual volume of water consumption per volume of feedstock produced (in dry short tons), or total annual volume of water consumption.

8.2.4 Data Sources

The water footprint is estimated by using historical climate data, including temperature, precipitation, solar radiation, and wind speed, which are available as national average values between 1970 and 2000 from the National Oceanic and Atmospheric Administration (NOAA). These data points, collected from more than 3,000 weather stations throughout the United States, were screened for data quality and geographic coverage and processed to generate a historical climate norm (Chiu and Wu 2012; Wu, Chiu, and Demissie 2012). That data set was used for this study. *BT16* scenario land management and feedstock production data are generated by the POLYSYS model (*BT16* volume 1). Acreages of each feedstock type and production yield, as well as county-level biomass feedstock mix for each agriculture scenario, were provided by POLYSYS. Types of feedstock, harvest operation, total production, and acreages of forest residues and whole-tree biomass growing the feedstock were provided by the ForSEAM model (see *BT16* volume 1).

Water footprint modeling parameters are adopted from the WATER model and literature. WATER provides monthly crop water use parameter K_c, accounting for the entire growing season, for each crop. The leaf area index (LAI) of different types of forest stands is collected from McCarthy et al. (2007); Oishi, Oren, and Stoy (2010); Albaugh et al. (1998); Antonarakis et al. (2010); and Sampson et al. (2003). The LAI of perennial grass is from the SWAT model. The proportion of hardwood and softwood in mixed stands is based on historical data from the USDA Forest Inventory and Analysis (FIA) Program (see http://www.fia.fs.fed.us). Additional climate and geography data were collected as needed from NOAA, USDA, and U.S. Geological Survey (USGS) databases.

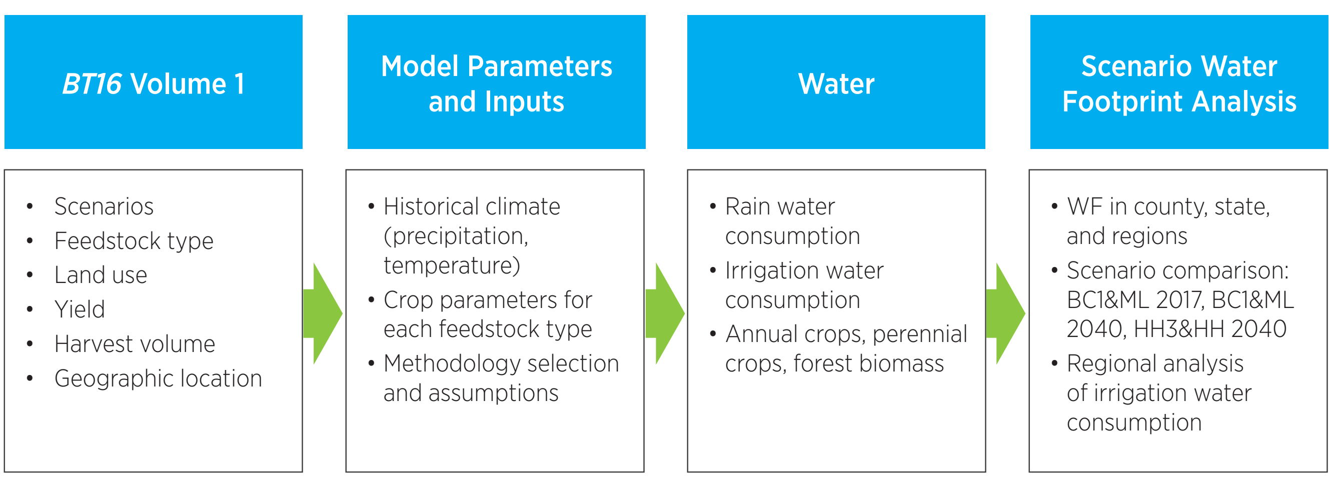
8.2.5 Description of Water Footprint Implementation

This study models water footprint for feedstock production scenarios by using eight steps, as illustrated in figure 8.2:

1. County-level feedstock production and harvest volume, acreage, and fertilizer management options are assembled into a water footprint database for each scenario, feedstock, and product class.
2. The *BT16* raw data for each scenario are sorted by feedstock component and processed into the input format.
3. Climate parameters supporting reference and feedstock ET calculations in various regions are determined; crop growth parameters and modeling methods for each feedstock type (annual crops, grasses, trees) are selected.
4. Using monthly time steps for each feedstock, growing-season plant water demand is computed according to equations 8.1–8.19 (see appendix 8-A) by using WATER.
5. County-level consumptive water use is estimated based on input feedstock data for each feedstock for each scenario.
6. Weighted average water footprint at the county level is obtained by aggregating results for individual feedstocks to determine the county-level water footprint of the feedstock-mix.
7. The state and regional water footprints for each scenario are processed from the county-level values.
8. The data are examined and regional analysis is applied to dissect the interplay between production yield, feedstock type, and water footprint in different regions. Results are presented on annual basis.

Consumptive water use by biomass is allocated on the basis of the fractions of the crop that are harvested for potential biomass production. Table 8.3 shows the fraction of corn grain for bioenergy production at the national level, ranging from 36.2% (scenario BC1 2017) to 28.4% (scenario HH3 2040), as estimated in *BT16* volume 1. The fraction of wheat straw that was collected for feedstock is negligible at the national level (*BT16* volume 1). The consumptive water use estimate is based on harvest acreage. Agriculture residues (corn stover and wheat straw) are harvested at different rates from county to county in *BT16* scenarios (see volume 1).

Figure 8.2 | Water footprint modeling for biomass production scenarios



The attribution method is an important determinant of water consumption. Irrigation water consumption for conventional crops could be attributed to grain or grain and residue, depending on allocation methods. With the purpose-based method, the irrigation water consumption during the crop growing season is allocated to grain. In the mass-based method, the irrigation water is allocated between grain and residue. Both methods are available in WATER. To be consistent with carbon and GHG accounting in Chapter 4, in which fuel and chemical inputs and resultant emissions during the crop growing season are attributed to grain (additional chemical inputs after harvest attributed to residue), the purpose-based attribution method was elected.

BT16 volume 1 estimated land areas and production volume for the forest feedstocks. Forest resources include several feedstock types that are harvested from the same piece of land. For example, saw log, residue, and pulp are different components of the whole tree. The feedstock types also differ depending on the forest operations, either clearcutting or thinning (see chapter 7 for details). Clear-cutting operations generate residue, saw log, and pulp, while thinning operations generate residue and whole tree. Therefore, the model allocates land area to each feedstock (residue and whole-tree) based on weight proportion of biomass harvested for bioenergy and operations and historical residue yield derived from forest inventory analysis (FIA) database (http://www.fia.fs.fed.us/). The feedstock allocation scheme applies to all three types of forest—softwood, hardwood, and mixedwood—to generate a county-level land allocation map for each forestry scenario.

Mixed stands are composed of softwood and hardwood trees. The water footprint of mixed stand types is calculated based on the proportion of hardwood and softwood in the total feedstock. *BT16* volume 1 provides county-level ratios of softwood to hardwood for the mixedwood harvest, which are calculated from the historical forestry dataset in FIA. This dataset is used to derive water footprints for mixedwood in all scenarios. The consumptive water use for forest resources is established by totaling the water footprints of mixedwood, softwood, and hardwood.

Irrigation water is not applied to forest resources and SRWC feedstocks because they are assumed to receive their required water from rainfall. Similar assumptions are applied to perennial grass feedstocks. See chapter 2 for a discussion of the irrigation assumptions embedded in the biomass yields in *BT16* volume 1.

8.3 Results and Discussion

8.3.1 Water Footprint of the Biomass Production Scenarios

8.3.1.1 BC1&ML 2017 Scenario

The BC1&ML 2017 scenario combines estimates of potential feedstock production from both annual conventional agriculture crops (scenario BC1 2017) and forest residues and whole-tree biomass (scenario ML 2017). In calculating potential forest residues and whole-tree biomass feedstock volumes, scenario ML

Table 8.3 | Corn Grain Harvest Scheme for Future Scenarios

	Scenario[4]		
Parameter	BC1 2017	BC1 2040	HH3 2040
Average fraction of corn harvest for use in bioenergy production	36.2%	32.16%	28.4%

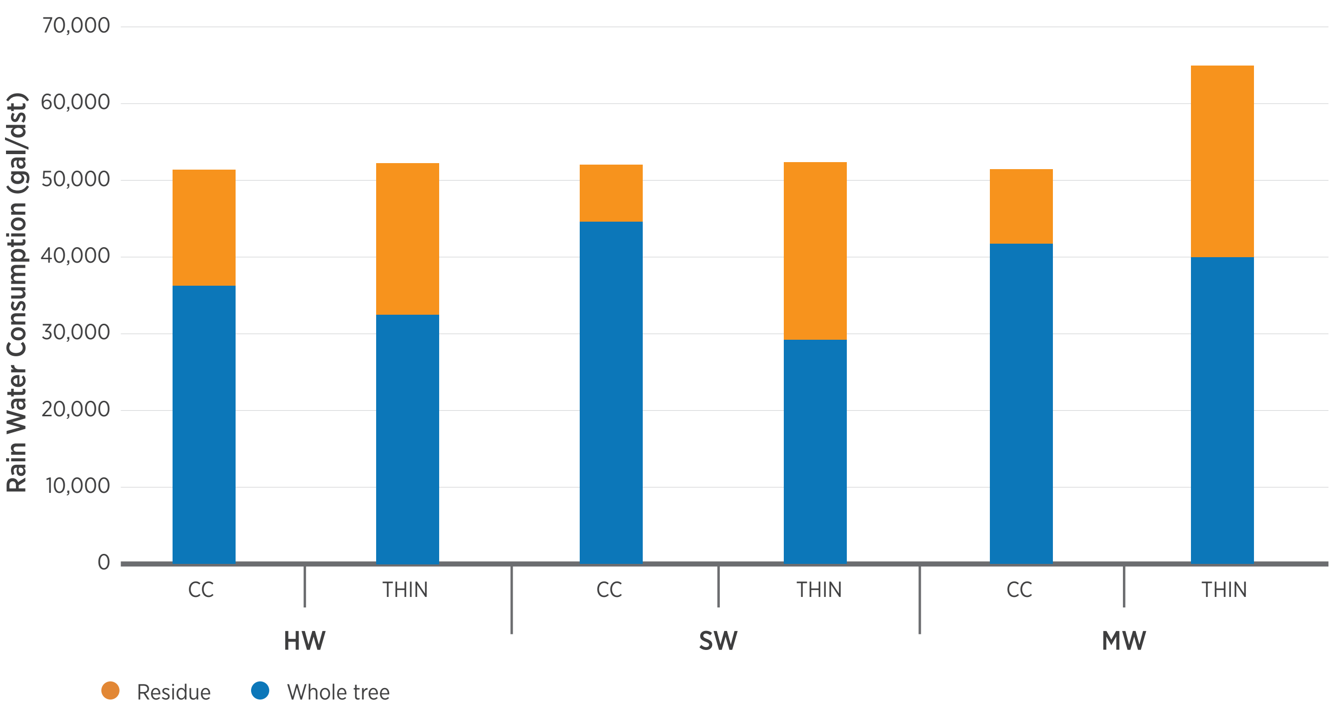
[4] The forestry scenarios are not included here because they have no corn grain.

2017 assumes existing forest land acreage. Figure 8.3 presents rainfall consumption by forest feedstock for scenario ML2017. The water footprint in scenario ML 2017 includes estimates for hardwood, softwood, and mixed stands under different harvest operations. Chapter 2 describes in detail the different types of forest harvesting operations indicated here, such as clearcut and thinning. A total of 88 million tons of biomass is harvested from forestlands in ML 2017 (*BT16* volume 1).

Scenario BC1 2017 represents current modeled biomass feedstock production from annual agriculture crops and residues. A total of 235 million tons of corn grain, soybean, corn stover, and wheat straw could be harvested nationally. For the areas that grow agriculture feedstock for biomass, production varies significantly, from 2 tons to 1.8 million tons for each county. The majority of the production is generated in the upper Midwest region of the country. (See figure 8.1 for a map of regions.) When agriculture crops and forest biomass are combined, the BC1&ML 2017 scenario shows four major production regions: agriculture feedstock–dominant in the Midwest, and forest biomass feedstock–dominant in the Southeast, Pacific, and Northeast (figure 8.4). Together, these regions could generate a total of 323 million tons of feedstocks from annual crops and forest biomass.

Rain water consumption for biomass production is spatially heterogeneous (figure 8.5) as a result of the aggregated distribution of regional feedstock types under BC1&ML 2017. Of the rain water used for biomass production in the scenario, a majority contributed to forest biomass in the Southeast (90%) and Delta (67%) regions. In the Corn Belt and Northern Plains, 80% of the rainwater consumed by biomass would be used to grow annual crop-based feedstock. As illustrated in figure 8.5, total rainwater use in each state depends on the acreages used for biomass production. In the BC1&ML 2017 scenario, states in Southern Plains, Delta, Southeast and Appalachia regions

Figure 8.3 | Rainwater requirements under scenario ML 2017



Abbreviations: CC = clearcut; THIN = thinning; HW = hardwood; SW = softwood; MW = mixedwood.

Figure 8.4 | Biomass feedstock production under scenario BC1&ML 2017

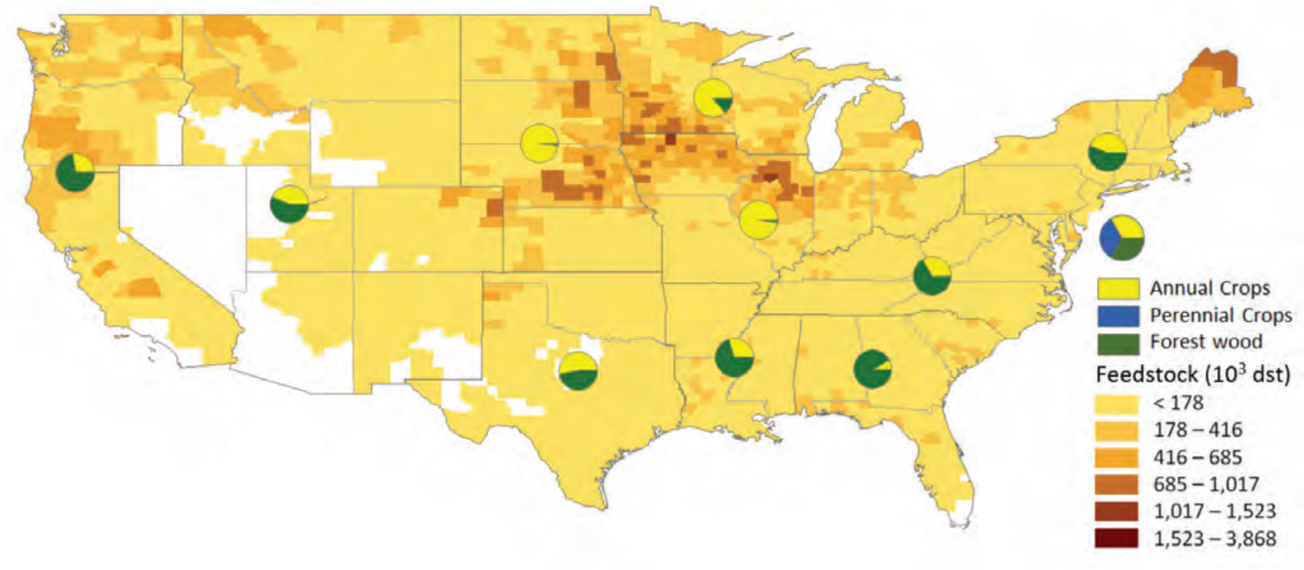


Figure 8.5 | Biomass feedstock production rainwater requirements under scenario BC1&ML 2017. Gal/ac is gallon per acre of biomass. Bgal is billions of gallons.

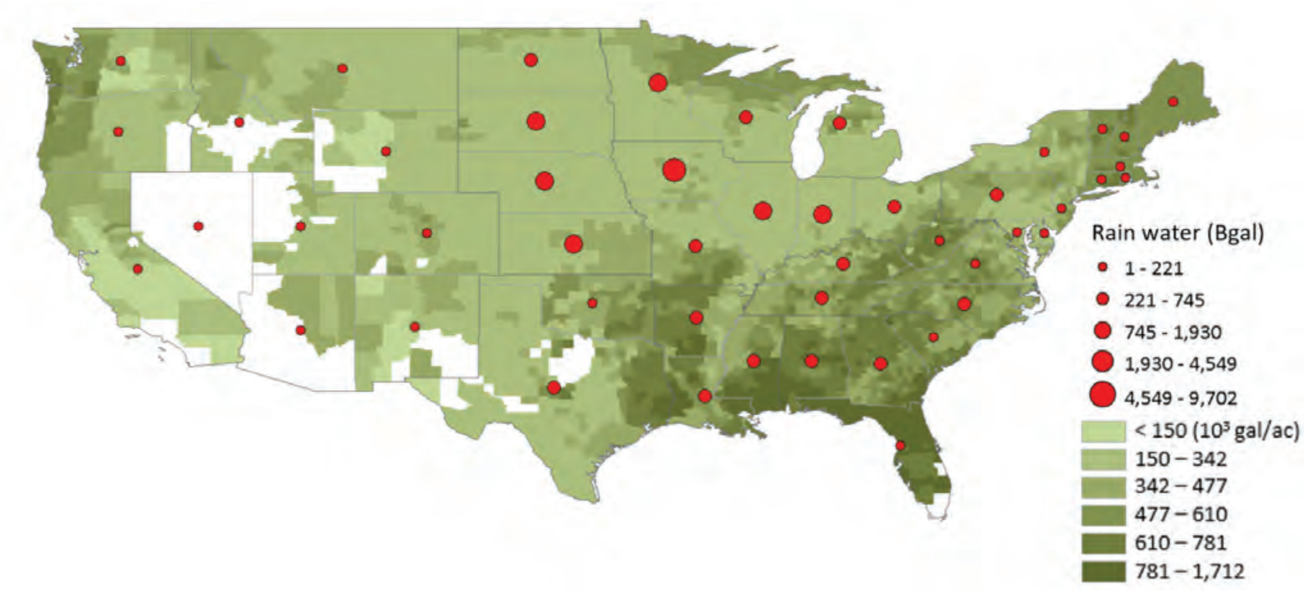
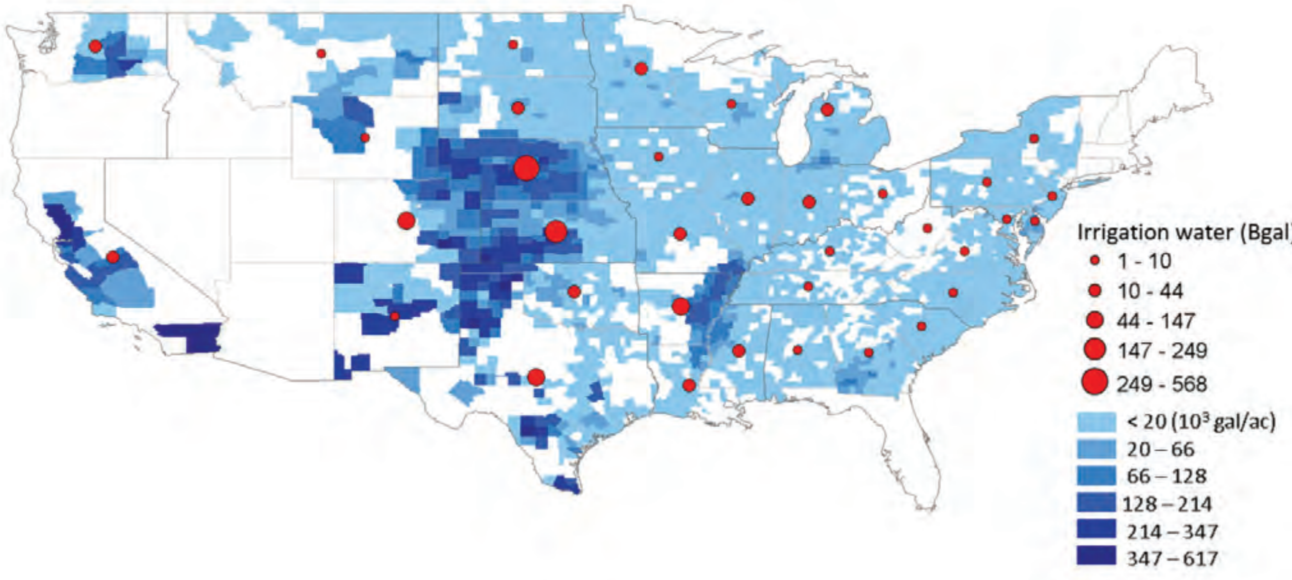


Figure 8.6 | Biomass feedstock production irrigation requirements under scenario BC1&ML 2017. Irrigated biomass consists entirely of corn grain and soybean. Gal/ac is gallon per acre of biomass. Bgal is billions of gallons.



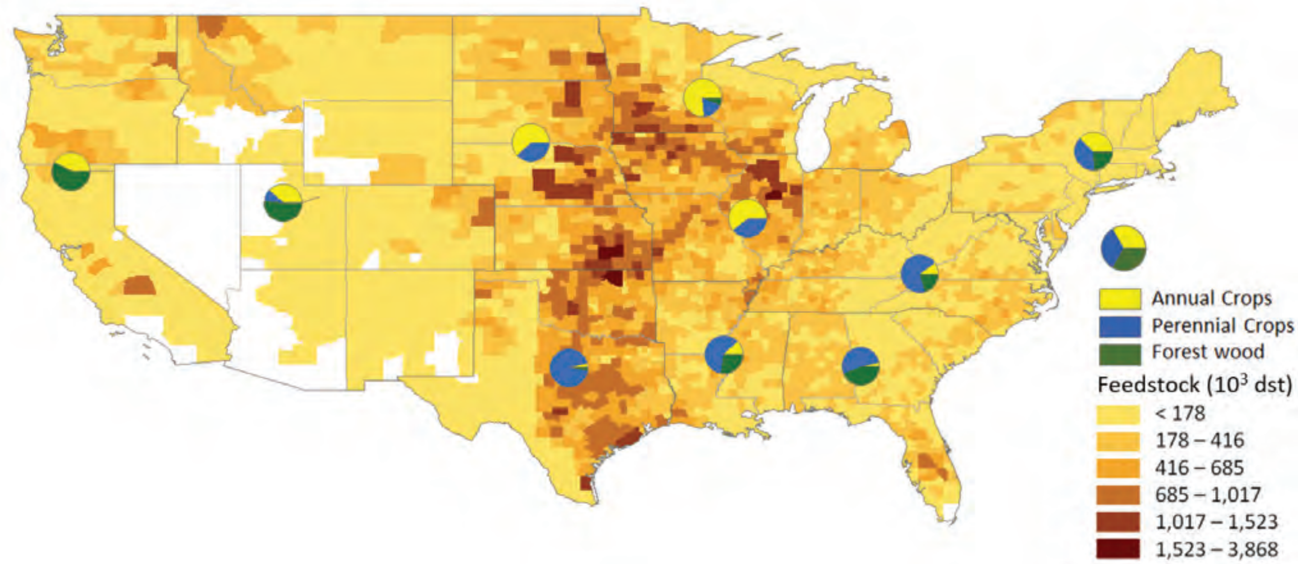
would require low to modest total rainwater because land area for biomass feedstock is relatively small. In the Corn Belt, a total of 6 trillion gallons of rainwater could be consumed per year by biomass production from agricultural and forest resources. The Northern Plains could consume 3.6 trillion gallons. Biomass production in the BC1&ML 2017 scenario would consume 17 trillion gallons of water from rainfall. In both figures 8.5 and 8.6, the shaded areas represent county-level rainwater use per acre of biomass (gal/acre), and circles represent state total volume (billion gallons).

Similarly, state total irrigation water consumption varies depending on feedstock type and acreage. For the regions that require irrigation to grow the feedstock (i.e., corn grain and soybean), approximately 1.4 trillion gallons of water would be consumed in this production scenario. Quantities varying from 20,000 gallons to 617,000 gallons of irrigation water would be required to grow an acre of annual crop feedstock in each county across the United States (figure 8.6). A significant portion of this water would be concentrated in the High Plains. Several other states—for example, New Mexico, California, and Washington—have similar irrigation demand per acre. Because of relatively small acreages attributed to feedstock production, the total volume of irrigation water in these states is low. Irrigation consumption in the states of the Corn Belt, where the bulk of current annual feedstock is produced from much larger land areas, is also low as a result of minimal irrigation requirements to grow each acre of crops (figure 8.6) in the region.

8.3.1.2 BC1&ML 2040

The BC1&ML 2040 scenario estimates a larger increase in potential feedstock production due to the growth of perennial grasses (switchgrass, miscanthus) and SRWC (willow, hybrid poplar, southern pine), in addition to annual agricultural crop feedstock (crop residue) and a slight decrease in the mass of forest residues and whole-tree biomass feedstocks. Scenario BC1&ML 2040 estimates a potential production of about 800 million tons of biomass per year, nationally (See chapter 1). This total is approximately 40% more than the estimated biomass production volume under the BC1 2017 scenario. The production area increased in the Southern Plains, Northern Plains, and Midwest regions compared to BC1 2017.

Figure 8.7 | Biomass feedstock production under scenario BC1&ML 2040



In BC1 2040 other locations began to produce perennial feedstock, such as switchgrass and miscanthus, as well. Perennial grasses and SRWCs can retrieve rainwater percolated in deep soil through their long root systems. As a result, they consume primarily rainwater. In land that was previously idle or used to grow annual crops, introducing a perennial cropping system translates to increased rain water consumption (figure 8.8). Total rainwater used for production of potential biomass under BC1&ML 2040 would be 43 trillion gallons in the United States. Of the quantity of rainwater consumed for feedstock, 31% would be used by annual crops, 67% by perennial crops, and 3% for forest biomass. Regional distribution of the rainwater consumption in BC1&ML 2040 is similar to that in BC1&ML 2017. Although both rainwater

Figure 8.8 | Biomass feedstock production based on rainwater under scenario BC1&ML 2040. Gal/ac is gallon per acre of biomass. Bgal is billions of gallons.

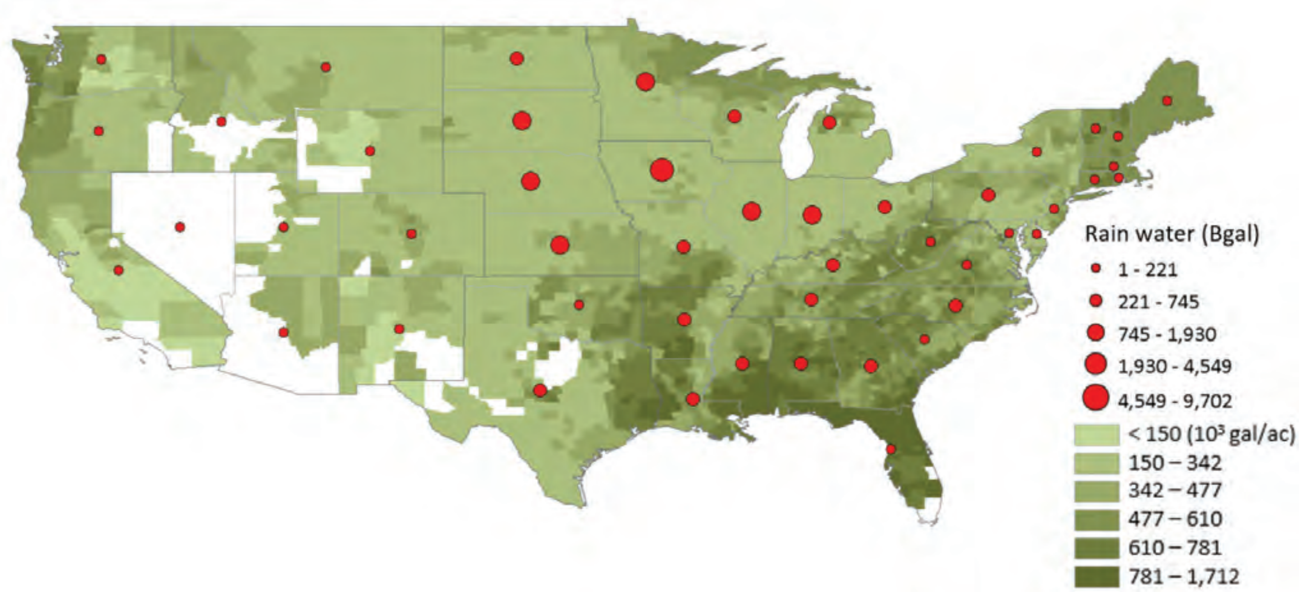
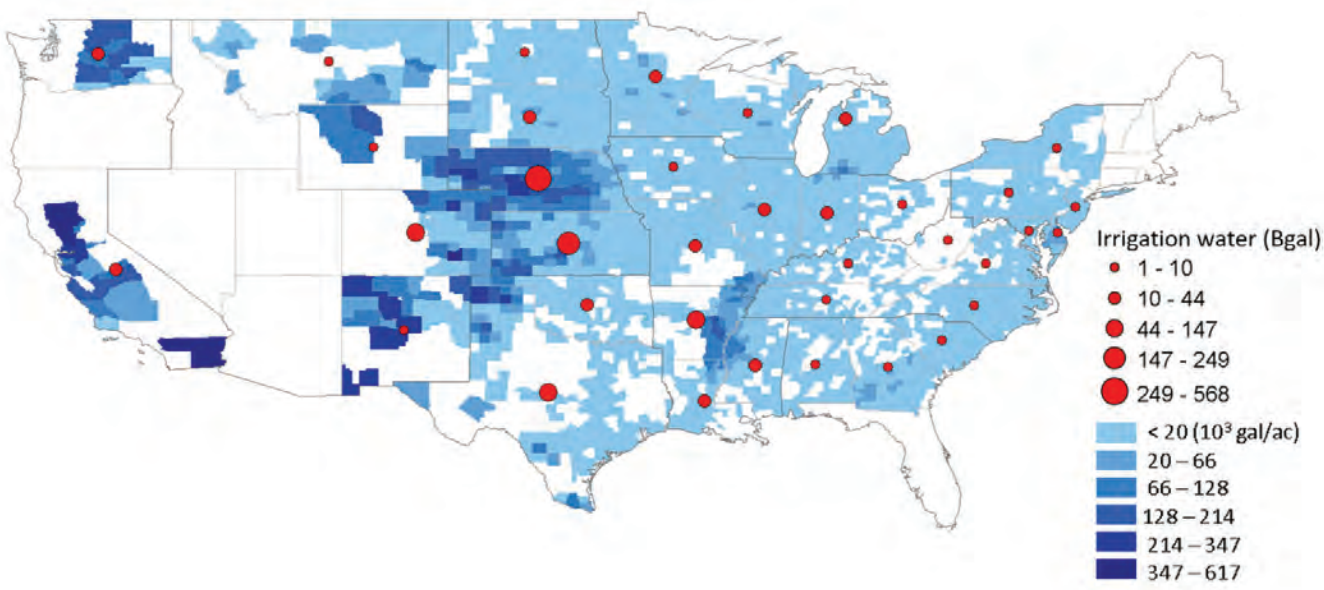


Figure 8.9 | Irrigation requirements for biomass feedstock production under scenario BC1&ML 2040. Gal/ac is gallon per acre of biomass. Bgal is billions of gallons.



and irrigation water are consumed in the production of biomass, rainwater is generally preferred because of its low cost, both economic and environmental, especially in water-rich regions.

Under scenario BC1&ML 2040, irrigation intensity (gal/acre) for biomass production either remained at the same level or decreased, compared to scenario BC1&ML 2017, especially in a few states in the High Plains region (figure 8.9). This decrease results from a reduction of annual crop acreages and an increase of perennial energy crops. For example, corn and soybean acreages were reduced by 75,000 acres in Nebraska and Kansas compared to BC1&ML 2017. In the same period the energy crop acreage increased by 6.3 million acres. The energy crops are not irrigated; therefore, irrigation water consumption decreased while feedstock production increased between the two scenario periods. Nationally, the scenario would consume 1,186 billion gallons of irrigation water, which is a 14% reduction from consumption in the BC1&ML 2017 scenario.

8.3.1.3 Scenario HH3&HH 2040

Under the high-yield scenario, HH3&HH 2040, which combines the high-yield feedstock production scenario from agriculture (scenario HH3 2040) and the high housing-high wood energy scenario from forestry (scenario HH 2040), estimates of perennial feedstock production could increase significantly and become dominant in the 1.3-billion-ton total-feedstock availability (see *BT16* volume 1). The majority of the potential perennial feedstock is produced in the Midwest and Southern regions (figure 8.10). Also under the scenario, more land that historically grows highly irrigated crops would move to the production of less- or non-irrigated perennial energy crops.

For the same reason as indicated in the discussion of the BC1&ML 2040 scenario, slightly more rainwater would be consumed by feedstock growth (figure 8.11), whereas irrigation demands would further decrease, compared to BC1&ML 2017 (figure 8.12). It is estimated that under the scenario, 48 trillion gallons of rainwater would be consumed for feedstock production. Figure 8.11 shows increased rainwater-use intensity in the Northern Plains, South-

Figure 8.10 | Biomass feedstock production under scenario HH3&HH 2040

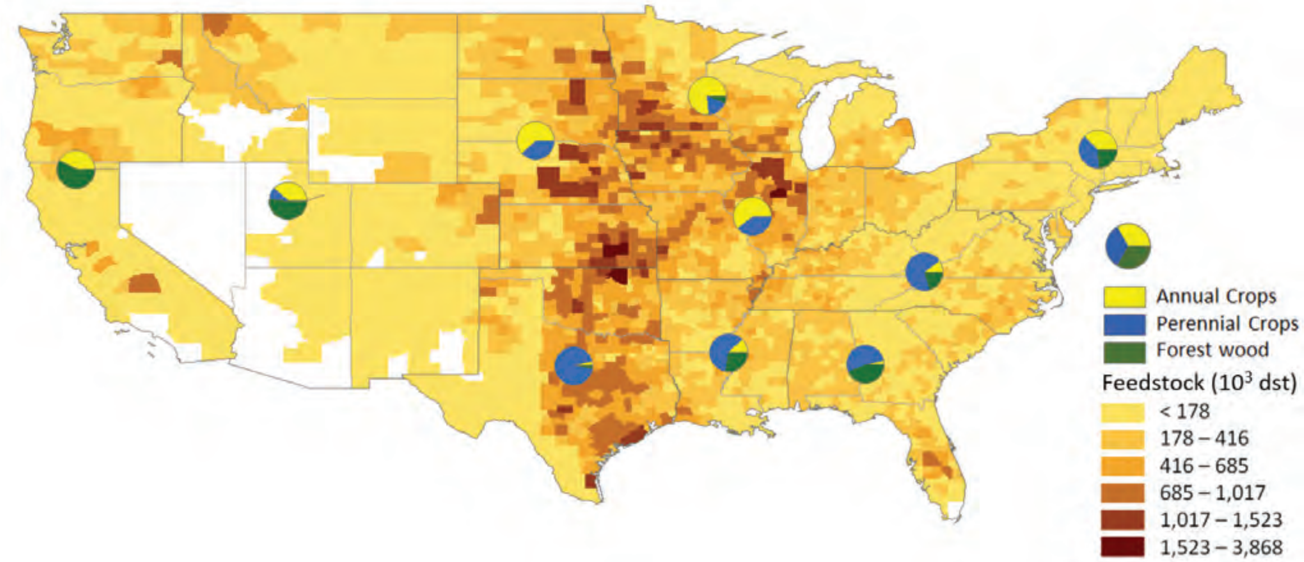


Figure 8.11 | Rainwater requirements for biomass feedstock production under scenario HH3&HH 2040. Gal/ac is gallon per acre of biomass. Bgal is billions of gallons.

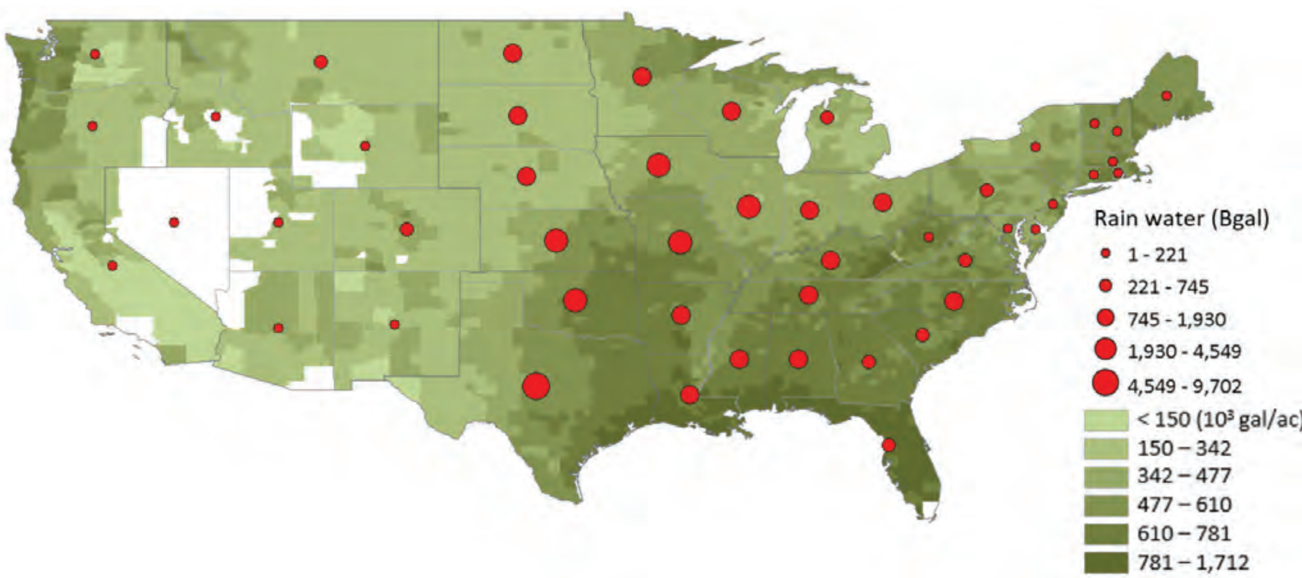
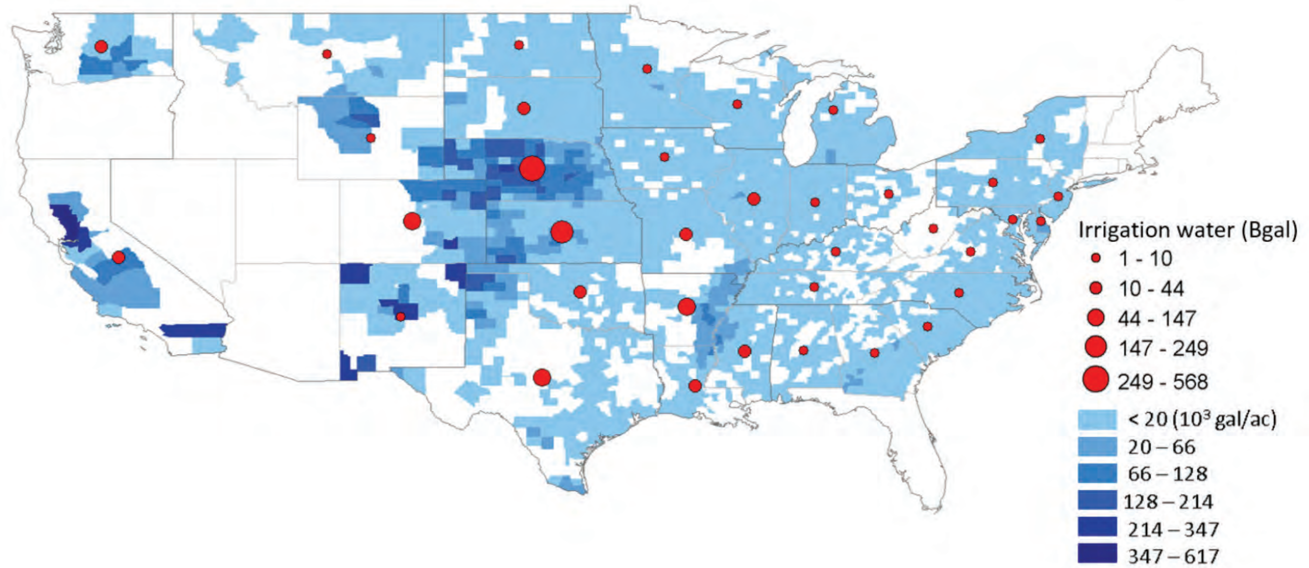


Figure 8.12 | Irrigation requirements for biomass feedstock production under scenario HH3&HH 2040. Gal/ac is gallon per acre of biomass. Bgal is billions of gallons.



ern Plains, Appalachia, Northwestern, and Cornbelt regions. Total irrigation water consumption would decrease to 1.0 trillion gallons from the 1.4 trillion gallons in the BC1&ML 2017 scenario.

8.3.2 Impact on Groundwater Irrigation

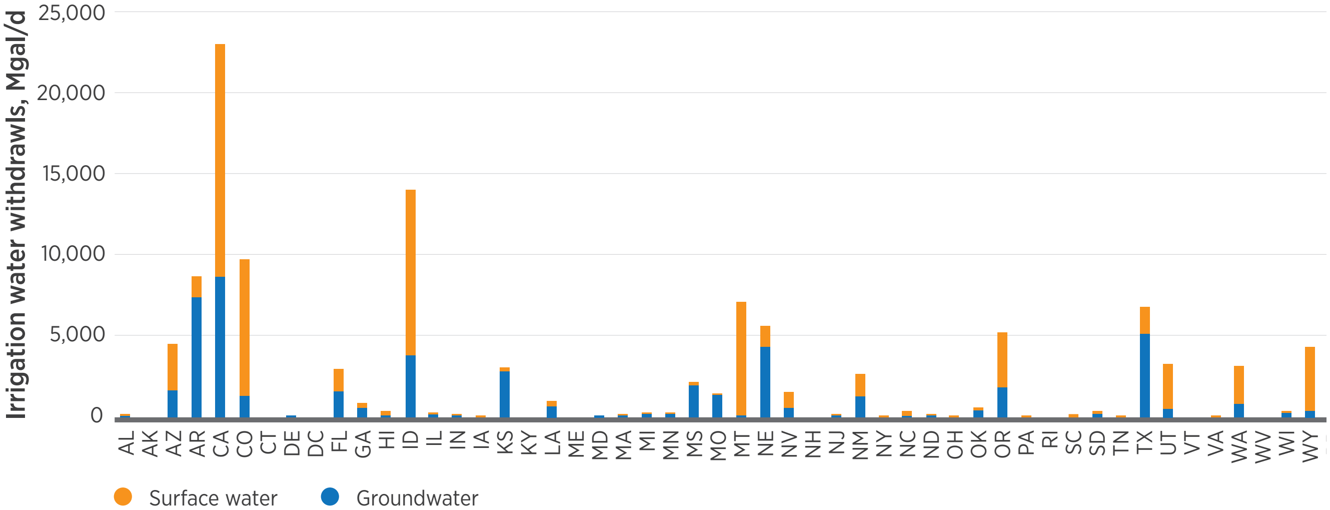
The U.S. agriculture sector withdrew 41.98 trillion gallons of fresh water for irrigation in 2010 (Maupin et al. 2014), which accounts for 38% of freshwater withdrawal for all uses. About 43% of the total irrigation water comes from groundwater sources (figure 8.13). In 2010, 18 trillion gallons of groundwater were withdrawn for irrigation. Geographically, 83% of U.S. irrigation withdrawal took place in the 17 conterminous western states. Surface water was the primary source of water in the western states, with the exception of Kansas, Oklahoma, Nebraska, Texas, and South Dakota, where groundwater was the main option.

The Ogallala Aquifer provides 20% of irrigation water to agriculture crops and cattle produced in the United States (Maupin et al. 2014). The counties in the High Plains withdrew a total of 5.8 trillion gallons from the Ogallala Aquifer for agricultural crop irrigation in 2010. (The western states that use groundwater for irrigation are mostly in the High Plains region.) The Ogallala Aquifer is facing depletion—the rate of water withdrawal far exceeds water replenishment. The area-weighted, average water-level changes in the aquifer were an overall decline of 15.4 feet from predevelopment to 2013, and a decline of 2.1 feet from 2011 to 2013. Total water in storage in the aquifer in 2013 declined 36.0 million acre-feet from 2011 to 2013. (McGuire 2014).

In 2011, the USDA Natural Resources Conservation Service (NRCS) launched the Ogallala Aquifer Initiative (OAI) to reduce aquifer water use, improve water quality, and enhance the economic viability of croplands and rangelands in Colorado, Kansas, Oklahoma, Nebraska, New Mexico, Texas, South Dakota, and Wyoming.[5] OAI aims to reduce water withdrawal

[5] See http://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/programs/initiatives/?cid=stelprdb1048809.

Figure 8.13 | Agriculture irrigation water withdrawals in the United States (Maupin et al. 2014)



and extend the life of the aquifer by implementing multiple conservation measures. One of the strategies is converting operations to dryland farming, which is defined as the non-irrigated cultivation of crops. This strategy is consistent with one of the sustainability principles in *BT16*: the production of non-irrigated biomass.

We analyze the impact of the *BT16* scenarios on groundwater use. The *BT16* feedstock production scenarios incorporate land management changes from irrigated land to non-irrigated land for biomass production. The feedstock portfolio changes from mostly starch-based material (scenario BC1&ML 2017) to mostly cellulosic-based material (scenario HH3&HH2040), and rain-fed acreages are increased in the latter scenario. Overall, irrigation water requirements could decrease significantly for each ton of feedstock grown in the United States if it replaces irrigated crops. As a result, groundwater consumption for irrigation in this region would decrease because irrigation water accounts for about 30% of ground water withdrawal from the Ogallala aquifer (McGuire et al. 2000). We compared groundwater irrigation for feedstock production in the High Plains between scenarios BC1&ML 2017 and HH3&HH 2040. On the basis of relative volume of the surface- and groundwater for irrigation in each state, we estimated the irrigation consumption for each county in the Ogallala Aquifer under both biomass production scenarios. As indicated in figure 8.14, irrigation water use for feedstock production would decrease in the 2040 scenario, compared to the 2017 scenario. The annual requirement for groundwater-based irrigation could be reduced from 720 billion gallons (Bgal) (scenario BC1 2017) to 540 Bgal (scenario HH3 2040), which is a savings of 179 Bgal in the High Plains. This quantity translates to 3.9% of the 5.8 trillion-gallon irrigation withdrawal from the Ogallala Aquifer in 2010 (assuming 80% of the water withdrawal is consumed). The reduction in groundwater irrigation in Ogallala counties is primarily due to reduced corn acreage for feedstock. A transition from irrigated-feedstock to non-irrigated feedstock could contribute to groundwater resource conservation. This would be consistent with federal and regional efforts to slow the depletion of the Ogallala Aquifer.

Figure 8.15 presents county-level irrigation con-

Figure 8.14 | Annual county-level groundwater consumptive use for feedstock irrigation under scenarios BC1&ML 2017 and HH3&HH 2040 in the High Plains Region

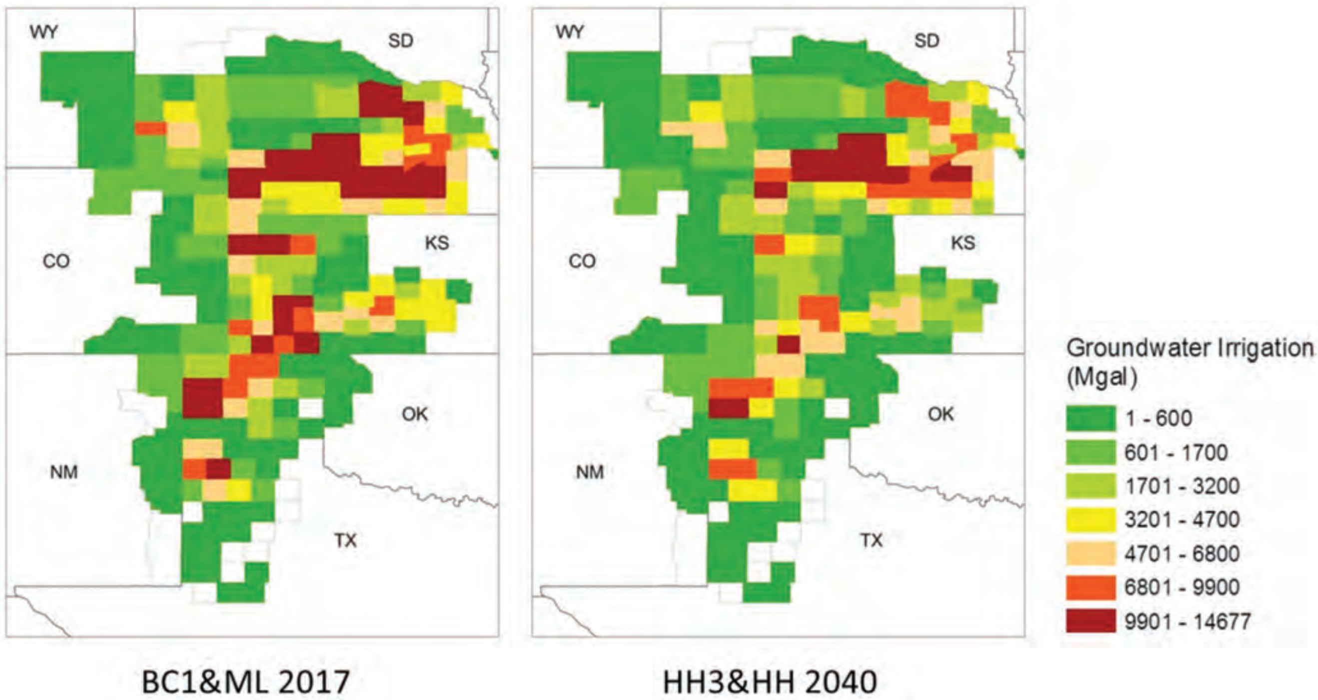


Figure 8.15 | County-level irrigation consumption, state surface water and groundwater fraction for biomass feedstock production under scenario HH3&HH 2040, and the reductions of irrigation from scenario BC1&ML 2017 to scenario HH3&HH 2040 in the conterminous United States

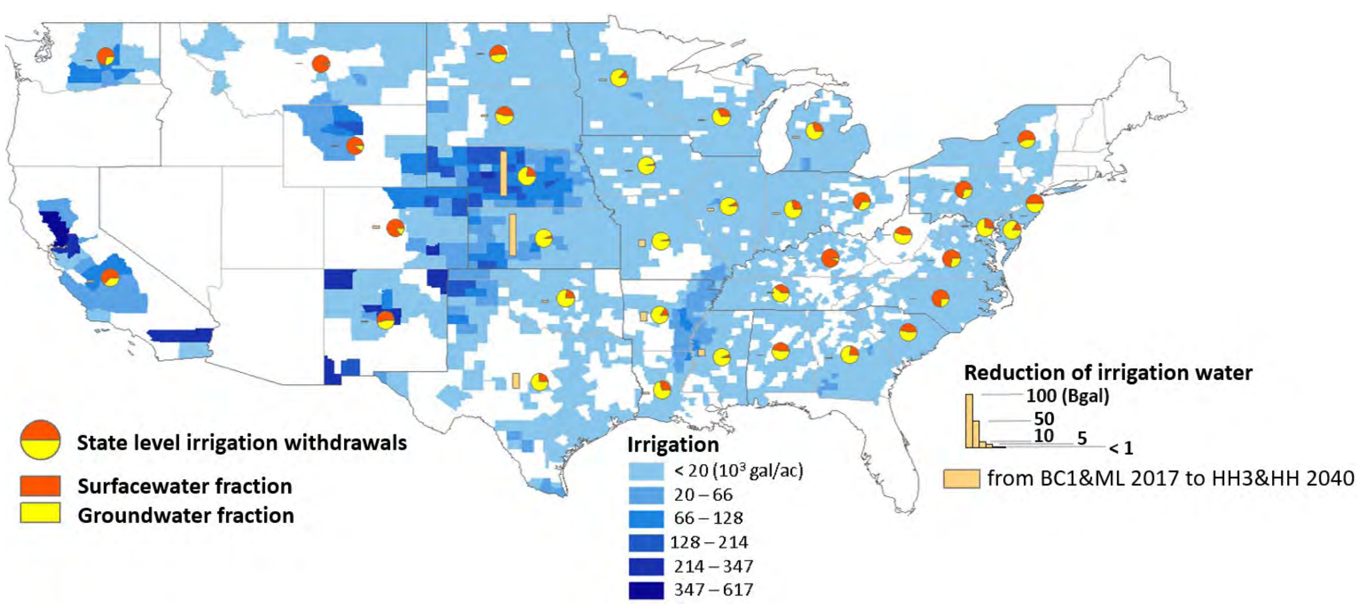


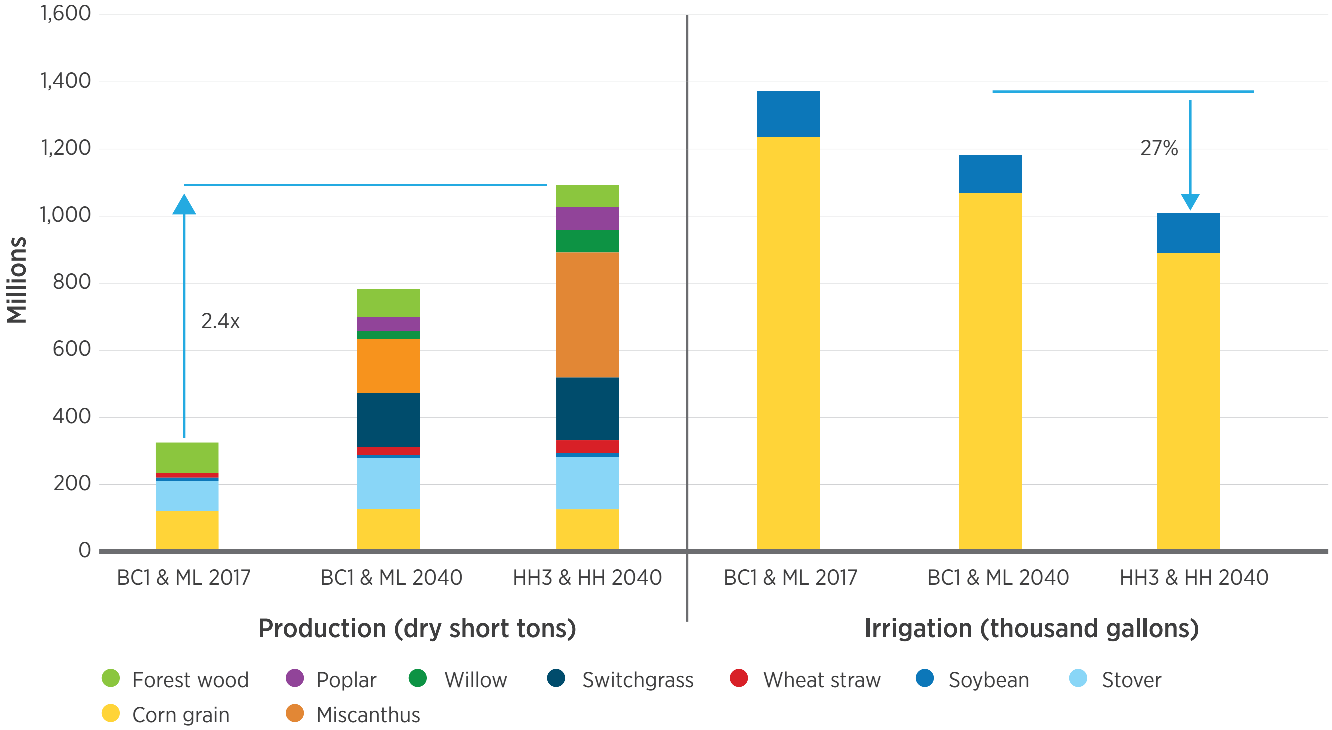
Table 8.4 | Groundwater Irrigation Consumption in Arkansas, Kansas, Nebraska, and Texas under Different Future Scenarios (billion gallons)

States	BC1 2017	HH3 2040	Change
Arkansas	65.0	47.2	17.9
Kansas	223.1	144.9	78.2
Nebraska	413.9	330.0	83.9
Texas	105.4	78.3	27.1
Sum	807.5	600.5	207.0

sumption, potential irrigation decreases from scenario BC1&ML 2017 to scenario HH3&HH 2040, and the surface- and groundwater fractions for each state. The key states that would benefit most from the potential biomass production scenarios are Nebraska and Kansas, followed by Texas and Arkansas. Compared with scenario BC1&ML 2017, annual groundwater irrigation would be 84 billion gallons less in Nebraska and 78 billion gallons less in Kansas in scenario HH3&HH 2040. The groundwater savings in Texas and Arkansas would be 27 billion gallons and 18 billion gallons per year, respectively, for the 2017 to 2040 period. Together, the four states could decrease 207 billion gallons of groundwater for irrigation (table 8.4).

Nationally, a total of 276 billion gallons that would otherwise be used for groundwater-based irrigation could be saved by transitioning from scenario BC1&ML 2017 to scenario HH3&HH 2040. Of the national total volume of irrigation water, 75% is

Figure 8.16 | Irrigation water footprint in the transition to cellulosic dominant feedstock mix for biomass production



attributed to the four states (Nebraska, Kansas, Texas, and Arkansas). Figure 8.16 shows a comparison of feedstock production volume and irrigation volume when scenarios progressed from BC1&ML 2017 to HH3&HH 2040. We found that from scenario BC1&ML 2017 to scenario HH3&HH 2040, biomass production could increase by a factor of 2.4, while irrigation water consumption from both surface and groundwater could decrease by 27% in the contiguous United States (figure 8.16).

8.4 Uncertainties and Future Work

Uncertainties are associated with these estimates, as in all analyses, because of incomplete data sources and assumptions made in developing the future scenarios. This study calculates the irrigation water demand of biomass crops and forest biomass, not the actual irrigation water volume withdrawn. In practice, irrigation operations often have water use variation—causing over- or under-irrigation. In particular, over-irrigation can affect regional water budgets and constrain resource use. The USDA NRCS has developed several tools to address this problem (USDA NRCS 2012), and many states have programs in place to provide guidance for irrigation management (USDA NRCS 2016).

The irrigation efficiency is a key factor related to water use. The U.S. Geological Survey reported that the number of acres irrigated by using water-efficient sprinklers and micro-irrigation systems continues to increase and accounted for 58% of all irrigated lands in 2010. The adoption of these new irrigation systems is believed to have contributed to an overall decrease in irrigation in 2010 (Maupin et al. 2014). Regionally, the level of adoption of advanced irrigation technology varies. In addition, the local availability of water resources also limits the ability of irrigation operations to fully meet the demand for crop water. Irrigation withdrawal monitoring data and technology adoption data for 2015, which were not available at the time of this work, would be an excellent source to enhance the analysis.

A number of methods have been proposed to estimate crop ET in the past few decades. The method adopted for this study (see appendix 8-A) is the American Society of Civil Engineers standard, which has been the dominant method used in the United States, with some variations. Mass-based allocation methods for attributing consumption of irrigation water to different parts of the plant (grain and residue) are available for additional analyses. Results generated from other methods may vary slightly compared to those of this study because of differences in approach and parameters. Therefore, it is highly desirable to conduct a full uncertainty analysis in the future.

Finally, this study represents an estimate of water consumption under specific future scenarios and their attendant assumptions based on available data, knowledge, and models. Further regional analysis to examine the water availability would be helpful, especially in water-stressed areas. A full water footprint analysis, accounting for water required in the biofuel-production life cycle, would provide a more complete picture of water consumption.

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Appendix 8-A

8A.1 Crops

Consumptive water use is quantified for the production of feedstocks (crops, grasses, short-rotation woody crops [SRWCs], and forest wood) by estimating evapotranspiration (ET). A large number of empirical methods have been developed over the last 50 years to estimate ET from different climate variables (Jensen and Allen 2000). The Penman-Monteith method (Allen et al. 1998) was standardized by the American Society of Civil Engineers' (ASCE's) Environmental and Water Resources Institute (EWRI) (ASCE-EWRI 2005), as illustrated in great detail by Howell and Evett (2004) and Allen et al. (2005a). In this study, the water footprint of agricultural crops adopts the so-called two-step Penman-Monteith method in which the crop ET is estimated by the Penman-Monteith reference ET method and crop coefficient (Jensen 1968; Allen et al. 2005b; Evett et al. 2000). The method has been widely used for nearly half a century and is relatively robust (Jensen 2010; Allen and Robison 2007). The Penman-Monteith method determines the reference ET of a crop using the following equation.

Equation 8.1:

$$ET_0 = \frac{[0.408\Delta(R_n - G)] + \gamma \frac{900}{T + 273} u_2(e_s^o - e_a)}{\Delta + \gamma(1 + 0.34u_2)}$$

Where:

ET_0 = reference ET rate (mm d^{-1}),

Δ = slope of the saturated vapor pressure curve $\delta e^o/\delta T$,

e^o = saturated vapor pressure (kPa),

T = daily mean air temperature (°C),

R_n = net radiation flux (MJ m^{-2} d^{-1}),

G = sensible heat flux into the soil (MJ $m^{-2}d^{-1}$),

γ = psychrometric constant (kPa $°C^{-1}$),

e_s^o = mean saturated vapor pressure (kpa),

e_a = mean daily ambient vapor pressure (kPa), and

u_2 = wind speed (m s^{-1}) at 2 m above the ground.

The crop-specific ET value is calculated from the reference ET and crop coefficients (Kc) at monthly intervals at each location and summed to annual crop ET. The water sources that support plant growth can be rainfall stored in the root zone, rainfall in the canopy layer, and/or irrigation. The quantity of rainfall available for the crops is described by the effective precipitation variable. Effective precipitation, which accounts for rainfall available for crop consumptive use, is obtained by applying the definition and method proposed by the U.S. Department of Agriculture's (USDA's) Natural Resources Conservation Service (NRCS) (Kent 1972; USDA NRCS 1997). Thus, the crop ET provided by rainfall is calculated each month by using equation 8.2, and these values are summed to find the annual value.

Equation 8.2:

$$Crop\ ET\ provided\ by\ rainfall = ETc, IF\ ETc < \ Eff\ prcp, else\ Eff\ prcp$$

Where:

ETc = calculated crop ET (mm/month) and

$Eff\ prcp$ = effective rainfall (mm/month).

The consumptive irrigation water requirement is estimated from the precipitation deficit, which represents the quantity of water beyond effective rainfall needed to sustain the growth (Allen and Robison 2007). The precipitation deficit is obtained by the differential of crop ET and effective rainfall at each monthly step, as shown in equation 8.3.

Equation 8.3:

$$Precipitation\ deficit = 0, IF\ ETc\ < Eff\ prcp, else\ ETc - Effprcp$$

$$Consumptive\ irrigation\ water\ requirement = Precipitation\ deficit$$

The monthly crop-consumptive, irrigation-water requirement is obtained from the calculated monthly precipitation deficit together with crop area. These monthly values are summed to find the annual irrigation demand.

8A.2 Perennial Grasses

To estimate actual evapotranspiration (AET) from perennial grassland, using the Penman-Monteith reference ET, ET losses from three major components are considered: (1) rain captured and evaporated from the grass canopy (Ecan), (2) vegetation transpiration (TP), and (3) evaporation from soil (Es). This study defines the sum of these three components as the AET of grasslands. Key parameters are adopted from the SWAT. The AET and its three components are computed in monthly steps by incorporating 30-year monthly input data for average climate (temperature, precipitation, solar radiation, humidity, and wind speed).

Equation 8.4:

$$ET_0 = \max\left(0.01, \frac{\left[\dfrac{\Delta \times Rn + \gamma \times (1710 - 6.85AvgT) \times (es - ea)/r_a}{\Delta + \gamma \times \left(1 + \frac{r_c}{r_a}\right)}\right]}{\lambda}\right) \times SunD$$

The input parameters in equation 8.4 are defined as follows:

Δ = slope of saturated vapor pressure,

Rn = net solar radiation (MJ m^{-2}/day^{-1}),

γ = psychrometric constant (kPa $°C^{-1}$),

$es - ea$ = difference in vapor pressure (kPa),

r_c = canopy resistance (s m^{-1}),

r_a = aerodynamic resistance (s m^{-1}),

λ = latent heat of vaporization (MJ kg^{-1}), and

SunD = number of sunny days = day count in a given month – RainD.

Rain captured and evaporated from grass canopy (E_{can}):

Equation 8.5:

$$E_{can} = if\ AvgT < 0{,}0, else$$

$$if\ ET_0 < 0.0004 \times LAI \times 1000 \times RainD, ET_0, else$$

$$0.0004 \times LAI \times 1000 \times RainD$$

Where:

AvgT = average monthly temperature (°C), using monthly maximum and minimum temperature as inputs;

ET_0 (mm $month^{-1}$) = reference ET (mm $month^{-1}$); and

LAI = leaf area index, estimated from vegetation height (*Hc*, in cm) in a given month.

RainD = average raining days in a given month.

Equation 8.6:

$$LAI = 1.5 \times LN(Hc) - 1.4$$

Vegetation transpiration (TP):

Equation 8.7:

$$TP = if\ LAI \leq 3, (ET_0 - E_{can}) \times LAI/3, else\ ET_0 - E_{can}$$

The calculation of E_{can} is completed by linking the estimates of plant *LAI*, ET_0, and E_{can} on a monthly basis.

Evaporation from soil (E_s):

Equation 8.8:

$$E_s = \min\left[E'_{s\,adj}, 0.8 \times (W_s - P_w)\right]$$

Where:

E_s = quantity of water evaporated from soil (in mm),

$E'_{s\,adj}$ = adjusted evaporated demand (in mm),

W_s = water content in the soil layer (in mm), and

P_w = wilting point (in mm).

The value of W_s fluctuates over time because of variability of ET, and Pw is defined by the local soil type. Together with the water content and wilting point, the evaporated demand (E'_s) can be adjusted ($E'_{s\,adj}$):

Equation 8.9:

$$E'_{s\,adj} = E'_s \times \exp\left(\frac{2.5(W_s - F_c)}{F_c - P_w}\right) \qquad if \quad W_s < F_c\,, else\; E'_{s\,adj} = E'_s$$

Equation 8.10:

$$E'_s = E''_s \times \left[\frac{100}{100 + \exp\,(2.374 - 0.00713 \times 100)} - \frac{10 \times 0.95}{10 + \exp\,(2.374 - 0.00713 \times 10)}\right]$$

Equation 8.11:

$$E''_s = \min\left[(ET_0 - E_{can}) \times \exp\,(-5 \times 10^{-5} \times M_{grass}), \frac{(ET_0 - E_{can}) \times \exp\,(-5 \times 10^{-5} \times M_{grass}) \times (ET_0 - E_{can})}{(ET_0 - E_{can}) \times \exp\,(-5 \times 10^{-5} \times M_{grass}) + TP}\right]$$

$$E''_s = 0,\;\; if\; AvgT < 0\; or\; ET_0 - E_{can} = 0$$

Where:

Fc = field capacity (in mm) and

M_{grass} = the plant coverage (kg ha^{-1}) on the soil.

The water content in the soil compartment at start of month (t) is then computed as equation 8.12.

Equation 8.12:

$$W_{s\,t+1} = W_{s\,t} + Prcp_t - AET_t$$

Equation 8.13:

$$Pw \le W_{s\,t+1} \le Fc$$

$$Ws_0 = Pw + 0.15(Fc - Pw)$$

The monthly AET values for grasses are summed to an annual value.

8A.3 Wood from Forests

Estimates of ET for wood from forests are based on the same principles as the estimates for perennial grasses, SRWCs, and crops. Again, the reference ET is determined by using the Penman-Monteith equation (Allen et al. 1998). ET equations for hardwood and softwood are adopted from previous studies (Sun et al. 2011; Tang et al. 2006; Irmak and Whitty 2003; Oishi et al. 2008; Ford et al. 2011). The hardwood ET calculation uses the accumulation method, which considers evaporation from the soil and the tree canopy, as well as transpiration from the canopy. The total ET is expressed as the sum of water lost from each component. Sun et al. (2011) proposed a method that estimates forest ET on a monthly basis by using tree leaf area index (LAI), precipitation (P), and the Penman-Monteith reference ET as inputs. Using this method, a study compared results with field data and showed improved ET estimates for softwood (Chiu and Wu 2013). ET calculations for SRWCs are based upon their categorization as either hardwood (poplar and willow) or softwood (pine).

Hardwood Evapotranspiration, AETḣw

- ***Soil Evaporation***

The equation for soil evaporation is as follows

Equation 8.14:

$$E_{sd} = 0.0123 \times \Delta\varepsilon^{1.3003} \times 48 \times \frac{M_D \times D_L}{24}$$

Where:

E_{sd} = daily soil evaporation in mm/month,

M_D = number of days of a given month.

D_L = daytime length in a given day of a year (h), and

$\Delta\varepsilon$ = vapor pressure deficit (kPa).

$$D_L = \left[24 \times \frac{ACOS\left(1-\left\{1-\tan\left(LAT_{county} \times \frac{PI}{180}\right)*\tan\left[\left(23.439\times\frac{PI}{180}\right) \times \cos\left(PI \times \frac{MD_{mid}}{182.625}\right)\right]\right\}\right)}{PI}\right]$$

Equation 8.15:

$$\Delta\varepsilon = 0.6108 \times \left(EXP\left\{17.27 \times \frac{2.24 + 0.49 \times (Tmax + Tmin)}{[2.24 + 0.49 \times (Tmax + Tmin)] + 237.3}\right\} - EXP\left(17.27 \times \frac{Tmin}{Tmin + 237.3}\right)\right)$$

Tmax and *Tmin* are the maximum and minimum monthly temperature in °C.

- ***Canopy Transpiration***

The tree canopy transpiration (mm $month^{-1}$), E_{tc}, is determined by equation 8.16.

Equation 8.16:

$$E_{tc} = 2.17 \times [1 - EXP(-2.27 \times \Delta\varepsilon)] \times M_D \times \frac{D_L}{24}$$

- ***Evaporation of the Intercepted rain***

Evaporation of the intercepted rain is the part of water loss that is equal to the portion of precipitation intercepted by the tree canopy. The equation can be described as follows:

Equation 8.17:

$$E_{ic\ ann} = (0.083\ P_{ann} + 0.036\ n) \times 25.4$$

Where:

$E_{ic\ ann}$ (mm yr^{-1}) is the annual precipitation (P_{ann} in. yr^{-1}) intercepted by tree canopy, and *n* is the number of rain events in the growing season.

To downscale the annual value to the monthly basis ($E_{ic\ m}$), $E_{ic\ ann}$ is weighted by monthly tree leaf area index (LAI).

Softwood ET, AET_{sw}

Equation 8.18:

$$AET_{sw} = 11.94 + 4.76 \times LAI + ET_0 \times (0.032 \times LAI + 0.0026 \times P + 0.15)$$

Where P (mm/month) is the monthly precipitation.

Equation 8.19:

$$P = 0.001013 \times \frac{P_{elv}}{0.622 \times (2.501 - 0.002361 \times AvgT)}$$

Where P_{elv} is the air pressure in kPa determined by a county's average elevation.

Equation 8.20:

$$P_{elv} = 101.3 - 0.01152 \times Elv + 0.544 \times 10^{-6} \times Elv^2$$

Where *Elv* is the county's average elevation.

8A.4 References

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9 Implications of Air Pollutant Emissions from Producing Agricultural and Forestry Feedstocks

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9.1 Introduction

Minimizing human health impacts is one tenet of sustainability. Human health problems associated with air pollution are not confined to urban areas. In fact, agricultural production is one of the largest contributors to the emissions of particulate matter and ozone precursors, which are regulated by the U.S. Environmental Protection Agency (EPA) due to their significant health (e.g., respiratory) and environmental (e.g., visibility, vegetation damages) impacts (EPA 2016c). Not surprisingly, across the biofuel supply chain, biomass production is one of the largest contributors to the emission of particulate matter and ozone precursors (Nopmongcol et al. 2011; Hill et al. 2009; Cook et al. 2011). In the context of air pollution, the magnitude combined with the spatial and temporal distribution are key to assessing the human health risks associated with a given emission source. Because biomass production and supply systems vary spatially, temporally, and by the types of biomass used, the potential expansion of biomass supply systems to meet large-scale bioenergy demands could lead to substantial changes in air pollutant concentrations across the United States (DOE 2016; Hill et al. 2009; Cook et al. 2011; Tessum, Marshall, and Hill 2012; Andersen 2013; Yu et al. 2013; Tessum, Hill, and Marshall 2014; Zhang et al. 2016).

Air emissions from biomass production have been modeled previously (e.g., Andersen 2013; Nopmongcol et al. 2011; Hill et al. 2009; Tessum, Marshall, and Hill 2012; Tessum, Hill, and Marshall 2014; Tsao et al. 2011; Huo, Wu, and Wang 2009; Cook et al. 2011). However, modeling in the literature is limited with regard to assessing potential large-scale deployment of biomass supply systems envisioned for the near-term and long-term future (Andersen 2013). Only in the last few years have small-scale studies of emissions from potential future biomass-collection and -transportation systems been performed (e.g., Yu et al. 2013). Most studies evaluate current or past feedstock-supply systems (Nopmongcol et al. 2011; Andersen 2013; Hill et al. 2009; Tessum, Marshall, and Hill 2012; Tessum, Hill, and Marshall 2014); the exceptions being Tsao et al. (2011) and Huo, Wu, and Wang (2009)—both of which considered scenarios that are representative of future feedstock-supply systems. In addition to not representing anticipated future conditions for biomass production, many studies are limited in terms of the feedstocks evaluated, emissions assessed, and spatial resolution modeled.

Across the biomass supply chain, multiple operations emit air pollutants; however, the type and source of emissions varies by feedstock. Characteristics of emission sources, their locations, and their time signatures are essential pieces of information for air-quality and human health impact modeling. This analysis develops an emissions inventory of emission sources associated with biomass production and supply, which can serve as a foundation for a subsequent air pollutant concentration and human health impact analysis. Our analysis also allows for the identification of key factors that contribute to emissions, which can inform the development of mitigation options. However, our analysis does not evaluate potential change in ambient air quality that may result from the emissions associated with increased biomass feedstock production and supply.

The objectives of this analysis of air pollutant emissions implications from potential biomass production and logistics from the three *BT16* scenarios (see chapter 2) are to

- Quantify air pollutant emissions associated with select scenarios of potential biomass production, harvest, transportation, and preprocessing that align with the select scenarios described in chapter 2 of this volume and compare emissions among feedstocks
- Estimate the spatial distribution of modeled air emissions and assess how these changes could potentially impact local air quality
- Identify opportunities to minimize potential adverse impacts, given that the design of the entire supply chain for primarily cellulosic biomass is still in its infancy.

Assessing the change to air pollutant emissions attributed to future potential biomass production requires the estimation of emissions for both a future scenario and a reference scenario, as well as the difference between the two. The scenarios analyzed in this study are consistent with those in the rest of *BT16* volume 2. However, in the context of this study, the reference scenario would need to include emissions from local agricultural and forestry sources, as well as other important sources of emissions, such as transportation. *BT16* lacks the detailed characterization of such a reference scenario; therefore, we report estimates of mass emissions for the scenarios evaluated and compare our results to EPA's most recent National Emissions Inventory (NEI) of U.S. air pollutant emissions (EPA 2016d).

The methods employed to achieve the stated objectives expand on those developed in Zhang et al. (2016) for estimating inventories of air pollutant emissions from potential biomass production. Key enhancements to the Zhang et al. methods are that we developed a method to estimate air emissions associated with biomass feedstock transportation and preprocessing. We also included new feedstock types and adopted crop budgets at higher spatial resolution than were available in the previous databases from Zhang et al. We updated assumptions regarding biomass production and harvest to ensure consistency with those in *BT16* volume 1. *BT16* focuses on the supply chain stages of producing biomass and supplying a subset of that biomass to the reactor throat of a biorefinery; biomass conversion to energy (e.g., biofuels) and biofuel combustion in vehicles are not a part of this analysis. This chapter does not take land management change results from *BT16* volume 1 and chapter 3 and estimate net changes in emissions.

Our inventory approach allows for an assessment of potential biomass feedstock production and logistics scenarios as compared to a set of baseline conditions. In particular, this chapter focuses exclusively on estimating air emissions from biomass supply systems to

- Understand how emissions differ among various biomass feedstocks and by location (i.e., counties in the contiguous United States), and how these emissions may evolve over time under different scenarios
- Identify the major emission contributors along the biomass supply chain in order to inform emission-mitigation strategies
- Compare the magnitude of feedstock-related emissions to county-level emissions (derived from EPA's NEI) to identify geographic areas at higher risk for potential negative air quality impacts, for instance, for those counties currently not in compliance with National Ambient Air Quality Standards (NAAQS) as of 2015.

9.2 Methods

County-level air pollutant emissions are estimated from anthropogenic sources for each of the three *BT16* scenarios described in table 9.1 and section 9.2.1. These scenarios are for 2017 (agricultural base case yield growth [BC1] and the moderate housing–low wood energy [ML] forestry scenarios combined: BC1&ML 2017) and 2040 (BC1&ML 2040, high-yield growth [HH3] and the high housing–high wood energy [HH] scenarios combined: HH3&HH 2040). The air pollutants analyzed are carbon monoxide (CO), particulate matter ($PM_{2.5}$, PM_{10}), oxides of nitrogen (NO_X),[1] oxides of sulfur (SO_X),[2] volatile organic compounds (VOCs),[3] and ammonia (NH_3). Air pollutants emitted from fuel used by equipment (e.g., agricultural machinery, transport vehicles); fertilizer and pesticide (collectively referred to as "chemicals") applications; soil and plant matter disturbance by mechanical force (e.g., wheels); and feedstock-drying processes (if applicable) are quantified.

Our analysis is focused on modeling direct, local air pollutant emissions. Indirect upstream emissions associated with fuel and chemical production are not included in this analysis, but they are discussed in section 9.3.4.1 in reference to the estimated emissions inventory. In addition, biogenic pollutants such as VOCs from biomass vegetation and cutting of biomass during harvest are not included, with one exception—VOC emissions from feedstock preprocessing and drying are included as they are biogenic emissions induced through an anthropogenic industrial process. Furthermore, we do not assess avoided emissions due to displacing production and extraction of fossil fuel (the part of the fossil fuel supply chain equivalent to biomass production). These limitations are discussed further in sections 9.3.4 and 9.4.2.

9.2.1 Scope of the Analysis

Our analysis is focused on developing air pollutant emissions inventories for three potential biomass production and harvest (hereafter referred to as "production") scenarios and three potential biomass feedstock transportation and preprocessing (hereafter referred to as "supply logistics") scenarios that align with the select scenarios evaluated in other chapters of this volume; complete scenario descriptions can be found in chapter 2. These scenarios are based on biomass production and supply logistics from *BT16* volume 1, and they include BC1&ML 2017 (near-term supply logistics to deliver bales or wood chips to the biorefinery), BC1&ML 2040 (long-term supply logistics to transform raw biomass to a pelletized commodity), and HH3&HH 2040 (long-term supply logistics). Each biomass production scenario corresponds to a supply logistics scenario, but energy crop production in the potential biomass production scenario for 2017 is expected to be zero because *BT16* volume 1 had reported that no crops were established in 2017, and the supply of conventional crops (e.g., corn grain) to biorefineries was not modeled. 1 had reported that no crops were established in 2017 and the supply of conventional crops (e.g., corn grain) to biorefineries was not modeled.

Model inputs to estimate air emissions for these scenarios include three sets of data: (1) regional equipment use and chemical application budgets for biomass production; (2) county-level biomass production data; and (3) supply logistics data for the subset of produced biomass supplied to biorefineries (including equipment, biomass transportation distance, and quantity of biomass). In a given county, potential biomass produced (e.g., all wheat straw and corn grain) may not be used for biofuel production in the *BT16* scenarios used in this chapter. The data sets are derived from *BT16* volume 1 or are in agreement

1 This includes nitric oxide and nitrogen dioxide.

2 This primarily includes sulfur dioxide, but it also includes other oxides of sulfur, such as sulfur monoxide and sulfur trioxide.

3 The list of VOCs accounted for from EPA methods and data sources are documented by EPA (2015a).

Figure 9.1 | Potential Biomass Production and Supply Logistics Scenarios from *BT16* Volume 1 Evaluated in This Chapter

Feedstock Type	Segment of the Supply Chain	BC1&ML[a]		HH3&HH[b]
		2017	2040	2040
Corn Grain	Biomass Production	Up to $60/ dry ton (dt)	Up to $60/dt	Up to $60/dt
	Biomass Supply Logistics, Near-Term	NM[c]	NM	NM
	Biomass Supply Logistics, Long-Term	NM	NM	NM
Cellulosic Agricultural Residues, Energy Crops, Whole-Tree Biomass and Logging Residues	Biomass Production	Up to $60/dt	Up to $60/dt	Up to $60/dt
	Biomass Supply Logistics, Near-Term	Up to $100/ dtd	NM	NM
	Biomass Supply Logistics, Long-Term	NM	Up to $100/ dt[d]	Up to $100/dt[d]

[a] BC1&ML scenarios assume 1% yield growth per year.
[b] HH3&HH scenario assume 3% yield growth per year.
[c] Not modeled (NM) as a part of *BT16*.
[d] Includes the cost to produce and supply the biomass.

with assumptions and inputs used to generate results in volume 1 (refer to section 9.2.2). Emissions for each scenario are estimated for all counties within the contiguous United States.

Table 9.2 presents the potential availability of biomass at a mean market clearing price of $60 per dry ton (dt) for years 2017 and 2040. We estimate emissions that would occur for biomass from the agriculture and forestry sectors listed in table 9.2. In this chapter we evaluate all cellulosic feedstocks potentially produced in 2017 and about 90% of cellulosic feedstocks potentially produced in 2040. In 2040 we do not evaluate the following: biomass sorghum, energy cane, eucalyptus, pine, poplar, or willow. We consider corn grain (*Zea mays L.*) because it is currently the most commonly used conventional crop for biofuel production in the United States; it is used as a point of comparison for all other biomass feedstocks assessed in this study. For the purposes of this analysis, we aggregate some feedstocks into a single category based on equipment similarities and low production volume as indicated in table 9.2. For example, corn stover and sorghum stubble are aggregated into the "stover" category, whereas corn grain, switchgrass, and miscanthus are all kept as separate categories.

The dimensionality in equipment and chemical application budgets for whole-tree forestry biomass (hereafter referred to as "whole-tree biomass") and logging residues vary by wood type, location, stand type, etc. (DOE 2016). Whole-tree biomass and logging residues are tracked separately because residue

[4] For example, this includes fertilizer and pesticide application rates, equipment types, equipment operation type (e.g., harvest), equipment hours of operation per unit of biomass or acre, and equipment horsepower.

budgets only include chipping and loading of the biomass at the roadside. To simplify our results within a county across budget dimensionality, we aggregated the emissions for all stand types and wood types, such as hardwoods, softwoods, and mixedwoods, into whole-tree biomass and logging residues categories for each county.

Biomass production scenarios represent total potential production at a mean market clearing price of $60 per dt regardless of use (i.e., includes biomass for all markets). Biomass supply responds to economic signals from several markets, and thus, biomass for biofuel is but one potential market for the biomass grown. Biomass supply logistics scenarios represent the potential supply of a subset of the biomass produced at a cost of up to $60 per dt that meets an average cost of up to $100 per dt delivered to biorefineries for biofuel production.

Potential agricultural residues and energy crop biomass production would increase from 2017 to 2040. However, due to the *BT16* assumption that no additional land will be used for forestry and that there will be no expansion of planted forest into "natural" forestland, logging residues biomass production would decrease from 2017 to 2040.

Table 9.2 | Potential Biomass-Production Levels (in dt) Evaluated in This Chapter

Biomass Feedstock Description	Biomass Feedstock Categories in This Chapter	BC1&ML[a] (million dt yr^{-1})		HH3&HH[b] (million dt yr^{-1})
		2017	2040	2040
Conventional Agricultural Crop				
Corn grain	Corn grain	390	450	510
	Subtotal	390	450	510
Agricultural Residues[c]				
Corn stover	Stover	89	150	160
Sorghum stubble	Stover	0.71	1.1	1.5
Wheat straw	Straw	13	21	37
Barley straw	Straw	0.41	0.57	0.48
Oats straw	Straw	0.0049	0.0081	0.0066
	Subtotal	100	180	200
Energy Crops[d]				
Miscanthus	Miscanthus	0	160	370
Switchgrass	Switchgrass	0	160	190
	Subtotal	0	320	560

Biomass Feedstock Description	Biomass Feedstock Categories in This Chapter	BC1&ML[a] (million dt yr^{-1})		HH3&HH[b] (million dt yr^{-1})
Forestry Biomass				
Hardwood Trees	Whole-Tree Biomass	39	25	18
Softwood Trees	Whole-Tree Biomass	28	33	20
Mixedwood Trees	Whole-Tree Biomass	2.8	2.4	2.4
Hardwood Residues	Logging Residues	6.9	8.0	7.9
Softwood Residues	Logging Residues	6.8	10	9.6
Mixedwood Residues	Logging Residues	4.2	2.7	2.4
Subtotal		88	81	61
Grand Total		**590**	**1,000**	**1,300**

[a] BC1&ML scenarios assume 1% yield growth per year.

[b] HH3&HH scenarios assume 3% yield growth per year.

[c] Agriculture residues include current feedstocks with production quantities available as bioenergy feedstocks.

[d] Dedicated energy crops are feedstocks that are not currently in production but are expected to be available as bioenergy feedstocks in the future.

9.2.2 Description of Feedstock Production Emissions to Air Model (FPEAM)

The National Renewable Energy Laboratory's (NREL's) FPEAM (fig. 9.1) is developed in Python v2.7.11 (Python Software Foundation 2016) and joins data and models from many disparate sources, discussed below, to estimate anthropogenic air emissions from the sources and supply chain stages described in the previous section. FPEAM uses input and output data from the Policy Analysis System (POLYSYS) model, Forest Sustainable and Economic Analysis Model (ForSEAM), and the Supply Characterization Model (SCM) to estimate air pollutant emissions of CO, $PM_{2.5}$, PM_{10}, NO_X, SO_X, VOCs, and NH_3. FPEAM uses regional equipment and chemical application data that are inputs to these models, biomass production estimates that are outputs from POLYSYS and ForSEAM, and biomass supply to the biorefinery estimates that are outputs from SCM. Input and output data from POLYSYS, ForSEAM, and SCM are generated externally and provided as model inputs to FPEAM simulations. Section 9.2.2.1 provides an overview of the scope of included emissions and emission sources. Section 9.2.2.2 describes FPEAM emission estimation methods, with details included in appendix 9-A section 9A.1, and section 9.2.2.3 summarizes the FPEAM outputs.

FPEAM's core methods for estimating emissions inventories are based on Zhang et al. (2016). However, FPEAM was expanded and improved for this chapter's analyses by including additional biomass feedstocks (e.g., miscanthus, whole-tree biomass) and emissions from the biomass supply logistics system. In this chapter, we reproduce documentation of many of the methods in Zhang et al. (2016) to ensure they are clear, as there have been many small changes to FPEAM to both update datasets and better align our analysis with the *BT16* study.

9.2.2.1 Emissions Inventory Scope

EPA regulates both the ambient concentration of pollutants with negative health impacts or other deleterious effects (so called "criteria" pollutants and hazardous air pollutants) and the mass emissions of precursor pollutants that either could lead to the formation of criteria pollutants or could have direct negative health effects (EPA 2013). Table 9.3 presents criteria pollutant and precursor chemicals for which emissions are estimated in this study. Emission sources considered are as follows:

- EPA regulates both the ambient concentration of pollutants with negative health impacts or other deleterious effects (so called "criteria" pollutants and hazardous air pollutants) and the mass emissions of precursor pollutants that either could lead to the formation of criteria pollutants or could have direct negative health effects (EPA 2013). Table 9.3 presents criteria pollutant and precursor chemicals for which emissions are estimated in this study. Emission sources considered are as follows:
- Fuel use by on-farm machinery operations (e.g., soil preparation, planting, chemical application, irrigation [corn grain only], harvesting, and on-farm transport of biomass)
- Fuel use from off-farm transportation; fuel use from biomass preprocessing; chemical application
- Chemical application of fertilizers and pesticides
- Fugitive dust emissions (PM_{10} and $PM_{2.5}$) from soil-disturbing activities (e.g., land preparation, fertilizer application, harvesting, and transportation)
- Drying of feedstock.

Table 9.3 | NAAQS Criteria Air Pollutants and Paired Air Pollutants or Precursors

NAAQS Criteria Pollutant	Ozone	$PM_{2.5}$ and PM_{10}	SO_2	NO_2	CO	Lead (Pb)*
Air Pollutant or Precursor	NO_X, VOC	NO_X, VOC, SO_2, directly emitted $PM_{2.5}$ or PM_{10}, NH_3	SO_X	NO_X	CO	Pb

*Lead is not evaluated in this study. **Acronyms:** SO_2 – sulfur dioxide; NO_2 – nitrogen dioxide.

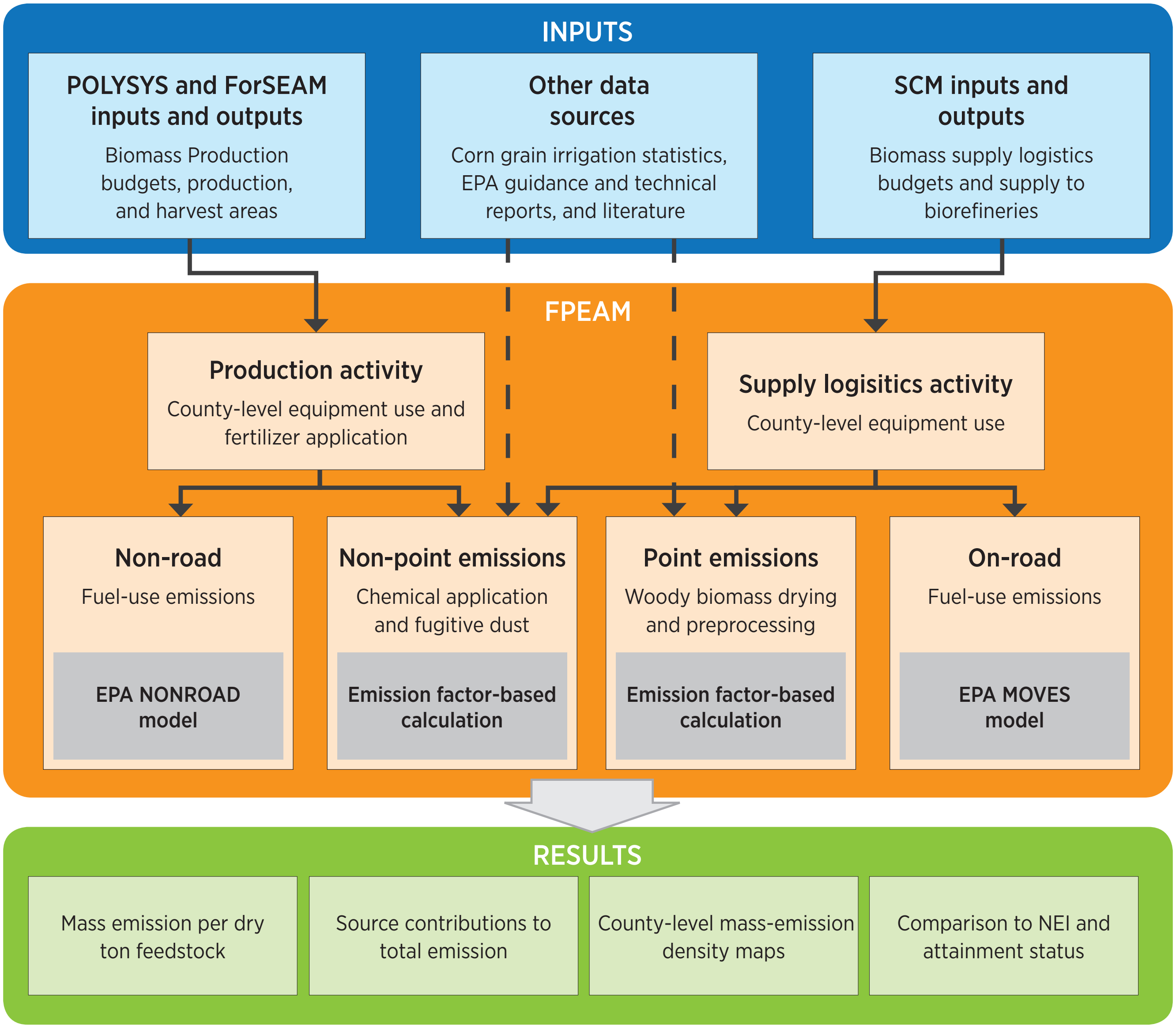
9.2.2.2 Emissions Modeling

Depending on the emission source, FPEAM estimates annual county-level emissions through one of two approaches:

- Linking the annual activity data (e.g., equipment usage, type of equipment) to EPA's MOtor Vehicle Emission Simulator (MOVES) model to generate estimates based on fuel use characteristics in the equipment/vehicle
- Applying emission factors (EFs) to applicable non-combustion sources (e.g., chemical application or fugitive dust from soil and plant matter disturbance).

Figure 9.1 summarizes the interlinkages between the primary FPEAM inputs and air pollutant estimation methods to generate model outputs (i.e., county-level air emissions). Table 9.4 builds on this by summarizing the sources and scope of the core elements of FPEAM's methods for estimating emissions. See below for a brief description of table 9.4. See appendix 9-A section 9A.1.1 for more details on estimating annual activity and see appendix 9-A section 9A.1.2 for greater details on EFs and total emissions estimation.

Figure 9.1 | FPEAM Model (orange shade) summary of the linkages between primary inputs (blue shade), emission estimation models and methods (gray boxes) used in or with FPEAM (orange boxes), and analysis results (green shades)



Annual activity of equipment (e.g., hours of operation per year) and chemical application that would be associated with each county under each scenario are estimated based on *BT16* volume 1. These data are based on the biomass production and supply logistics budgets used as inputs to POLYSYS, ForSEAM, and SCM. They are also based on POLYSYS and ForSEAM estimates of potential harvested area and biomass production, and SCM estimates of potential biomass supply (DOE 2016). Our method also considers the use of irrigation equipment for corn grain-based irrigation based on data from the U.S. Department of Agriculture (USDA) (USDA 2009).

In alignment with *BT16* budget data, product-purpose-based allocation is assumed for allocating emissions among multi-product production systems, such as those generating residues as byproducts (Johnson et al. 2004; Wang, Huo, and Arora 2011). Equipment operations associated with biomass production are entirely attributed to grain or wood rather than residues; in agriculture, harvest activities are allocated between the crop and agricultural residue; and additional chemical and nutrient applications (to compensate for nutrient loss) are attributed to stover, straw, or logging residues.

Switchgrass and miscanthus are perennial crops with 10- and 15-year production cycles, respectively; each with differing equipment budgets for each rotation year. To compare them to annual crops, we annualize emissions from equipment use and chemical application over all rotation years for these crops by assuming 10% of total switchgrass and 6.67% of miscanthus acres are in production in each rotation year in each county. Year-to-year emissions may be more variable depending on where the crops are in the rotation cycle.

For air pollutant emissions that would be generated by mobile and non-mobile equipment, emissions are estimated in FPEAM by using EPA's MOVES Model version 2014a (EPA 2016a). For non-road equipment, the MOVES Model relies on the submodel NONROAD 2008a (EPA 2016b; hereafter referred to as NONROAD) to compute county-level air pollutant emissions for machinery like combines, tractors, and chippers. In addition, the main MOVES Model uses county-level EFs to compute county-level air pollutant emissions from on-road machinery such as trucks. While MOVES estimates CO, NO_X, SO_X, PM_{10}, $PM_{2.5}$, NH_3, and VOCs emissions directly, NONROAD only calculates CO, NO_X, SO_X, PM_{10}, and total hydrocarbon (THC) emissions. As a result, for NONROAD equipment, we estimate the emissions of NH_3, $PM_{2.5}$, and VOCs using EPA EFs based on fuel consumption, THCs, and PM_{10}, respectively (see appendix 9-A section 9A.2.1 for further details).

Transportation distance for potential biomass supplied to biorefineries is determined using the SCM (DOE 2016). While on-road transportation emissions are being estimated at a county level, we do not have the necessary pathing (i.e., course routing) data for specific biomass streams. As a result, all on-road transportation emissions are allocated to the county producing the biomass.

NH_3 and NO_X (in the form of NO) emissions from the application of nitrogen (N) fertilizers are estimated based on EFs specific to each fertilizer and pollutant (EPA 2015d; Hall and Matson 1996; Veldkamp and Keller 1997; Goebes, Strader, and Davidson 2003). For the pollutants examined, no EFs for the application of potassium and phosphorus fertilizers were found, so this analysis excludes emissions that would be generated by these fertilizers.

Fugitive dust is $PM_{2.5}$ and PM_{10} that is emitted from the mechanical disturbance of granular material (typically soil and plant matter) exposed to the air and from mechanical systems preprocessing operations (chipper, hogs, tubs, etc.) (USDA 2011; EPA 2006). This kind of dust is called "fugitive" because it is not created in a confined flow stream. Typical sources of fugitive dust include unpaved roads, agricultural tilling operations, aggregate storage piles, and heavy construction operations. Dust is typically generated by two basic physical phenomena: (1) pulverization and abrasion of surface materials by applying mechanical force with implements (wheels, blades, etc.); and (2) entrainment of dust particles by the action of turbulent air currents, such as the wind erosion of an exposed surface. No methods for estimating fugitive dust from forestry activities were found, so we assume fugitive dust emissions are zero. There is evidence that this gap may not have a significant impact on our results because research has shown that vegetation in forested areas can potentially remove 80%–100% of particulate emissions (Pace 2005). Fugitive dust from preprocessing equipment was assumed to be zero because of the dust collection systems included in both near-term and long-term supply logistics designs (INL 2013; INL 2014).

Drying woody biomass is the main approach for lowering the moisture content of the biomass in both near-term and long-term supply logistics designs (INL 2013; INL 2014). During the drying process, biogenic VOC emissions would be expected to be emitted to the air (EPA 2002), and they are accounted for in our emissions inventory. Due to the limits of the available data on herbaceous feedstocks (e.g., EPA 1996), we assume there are no VOC emissions from herbaceous feedstock drying. We do not include other biogenic related air pollutant emissions, for instance, from the growth of herbaceous or woody feedstocks.

Logging residues are sometimes piled and burned. The use of this practice varies based on a number of factors, including ownership, location, type, regeneration, and forest productivity. Because we did not have access to spatial data on specific logging residue management practices, this analysis does not estimate any credits from the offsetting of burning logging residues.

Although we do not include upstream emissions in the study, we do discuss potentially large sources of upstream emissions and present example estimates that could be expected, such as upstream emissions from biomass preprocessing equipment that consumes electricity. These results are presented and discussed in sections 9.3.4.1 and 9.3.4.2, respectively. Emissions from electricity use would not be local, and even the general location of their release would be difficult to pinpoint. In section 9.3.4.3, we discuss a sensitivity estimate of emissions assuming 99%, rather than 100% dust collection and compare it to other sources of PM emissions. In section 9.4.2, we discuss other important shortcomings of our approach and methods, such as the limitations in evaluating fugitive dust emissions and biogenic emissions from forestry and open burning of whole-tree biomass.

Table 9.4 | FPEAM Model Summary and Documentation of Methods

Purpose	FPEAM Modeling Method	Emission Species	Spatial Resolution	Estimation Methods/Data Sources	Details in Appendix Section
Annual Equipment Usage and Chemical Application	Equipment and Chemical Application Budgets[a]	CO, NO_x, SO_x, $PM_{2.5}$, PM_{10}, VOCs, NH_3	**Agriculture** 13 regional budgets **Forestry** 5 regional budgets **Supply Logistics** National **Corn Grain Irrigation** State	POLYSYS, ForSEAM, and SCM modeling inputs (DOE 2016) **Corn Grain Irrigation** USDA (2007)	9A.1.1
	Harvest Area and Biomass Production	CO, NO_x, SO_x, $PM_{2.5}$, PM_{10}, VOCs, NH_3	County	POLYSYS, ForSEAM, and SCM modeling output (DOE 2016)	9A.1.1
Emission Factors (EFs) For Estimating Annual Emissions	Off-Road Fuel Use	CO, NO_x, SO_x, $PM_{2.5}$, PM_{10}, VOCs, NH_3	State EFs	NONROAD (EPA 2016b)	9A.1.2.1
	On-Road Fuel Use	CO, NO_x, SO_x, $PM_{2.5}$, PM_{10}, VOCs, NH_3	State EFs	MOVES (EPA 2016a)	9A.1.2.2
	Preprocessing Fuel Use	CO, NO_x, SO_x, $PM_{2.5}$, PM_{10}, VOCs, NH_3	State EFs	NONROAD (EPA 2016b)	9A.1.2.3
	Chemical Application	NO_x, VOCs	National EFs	EPA (2015c) ANL 2015 USDA (2010) Davidson et al. 2004 Huntley (2015)	9A.1.2.4
	Fugitive Dust	$PM_{2.5}$ and PM_{10}	EFs based on a combination of state and national data	**Agriculture Harvest and Non-Harvest** CARB (2003), Gaffney and Yu (2003) **Forestry** No methodology or data could be found **Transportation** EPA (2006) **Preprocessing** None due to dust-collection equipment (INL 2013; INL 2014)	9A.1.2.5
	Drying and Preprocessing	VOCs	National EFs	Herbaceous: Assumed to be zero Woody: EPA (2002)	9A.1.2.6

[a] Budgets include additional dimensions not described here (e.g., budgets by tillage type, rotation year for energy crops, and forestry land type).

9.2.2.3 Emission Metrics

Three metrics were used in this study to provide insights about the differences in emissions from potential feedstock production, sources of emissions, and the comparison to historic emissions:

- Air pollutant emissions per unit of biomass produced or supplied, which are used to compare corn grain and cellulosic feedstocks (section 9.2.2.3.1)
- Percent contribution of emissions by activity type to identify the activities that contribute most to the emissions of each pollutant (section 9.2.2.3.2)
- Ratios of emissions from *BT16* scenarios and current national emissions inventories, specifically the 2011 NEI and NAAQS 2015 attainment status (section 9.2.2.3.3).

9.2.2.3.1 Emission By Feedstock

The metric of air pollutant emissions per unit of biomass produced or supplied is calculated as a ratio. For biomass produced in *BT16* scenarios, the numerator is the sum of county-level mass emissions associated with the production of a given feedstock, and the denominator is calculated based on the county-level feedstock produced. For biomass supplied to biorefineries in *BT16* scenarios, the numerator is the sum of county-level mass emissions associated with the supply of a given feedstock, and the denominator is the mass of a given feedstock supplied in a given county.

9.2.2.3.2 Contribution of Emissions by Activity Category

For each feedstock, we estimate and compare the relative contribution of each of five activity categories (described below) to the total aggregated emissions from biomass production. Relative contribution is determined at a county level and displayed as national distributions of county-level emissions for each feedstock and pollutant. This metric provides insight into which activities are major contributors to certain air pollutant emissions, which can help focus future research on mitigation strategies, as well as the variability of contribution which can suggest mitigation strategies. Below, we detail how emissions are aggregated for each of the five categories:

- **Non-Harvest Emissions:**[5]
  - Fuel use-related emissions from machinery operations associated with chemical application and field preparation (e.g., cultivating, discing, plowing, and irrigation)
  - Fugitive dust emissions from non-harvest equipment usage.
- **Chemical Application Emissions:**[6]
  - NH_3 and NO_X from nitrogen fertilizer application
  - VOC emissions from pesticide application.
- **Harvest Emissions:**
  - Fuel use and fugitive dust emissions from machinery operations (e.g., mower, rake, baler) associated with feedstock harvesting
  - Fuel use and fugitive dust emissions from equipment used to transport feedstock to a temporary on-farm storage facility
  - Fuel use emissions from loading biomass for on-road transportation
  - Fuel use emissions from preprocessing equipment used at the site of harvest (e.g., wood chipper).
- **On-Road Transport Emissions:** Fuel use and fugitive dust emissions from transporting feedstocks to biorefineries by truck from the farm to the depot and/or biorefinery depending on the type of logistics system.
- **Preprocessing Emissions:** VOC emissions from preprocessing and drying at the facility.

For biomass produced, equation 9.1 calculates the contribution of each individual biomass production activity (non-harvest, chemical application, and har-

[5] No methods for estimating fugitive dust from forestry activities were found so we assume fugitive dust emissions are zero.

[6] Note that for fertilizer and chemical applications, the fuel use and fugitive emissions associated with applying the fertilizers/chemicals are accounted for in the non-harvest activity category.

vest) to the overall emissions from potential biomass production. The ratio is computed by pollutant (*p*) and by feedstock, for each county, which produces a given feedstock.

As stated in section 9.2.1, only a subset of biomass produced would be supplied to biorefineries in the scenarios examined as a part of *BT16*. Therefore, in a given county, potential biomass produced may not be used for biofuel production (DOE 2016). For biomass, which is produced and supplied to biorefineries, equation 9.2 calculates the contribution of each individual activity to overall emissions from all feedstock production and supply-related activities.

Equation 9.1:

$$\textit{Production Activity Contribution}_p = \frac{\sum \textit{ emissions by activity}}{\sum \textit{emissions across biomass production activities}}$$

Equation 9.2:

$$\textit{Production and Transporation Contribution}_p = \frac{\sum \textit{ emissions by activity}}{\sum \textit{emissions across all activities}}$$

9.2.2.3.3 Comparison to NEI and Attainment Status for NAAQS

Our air pollutant emissions inventory is compared to the county-level NEI for 2011 to illustrate the magnitude of emissions from *BT16* biomass production and supply logistics scenarios relative to inventoried emissions in a county. The NEI is a comprehensive and detailed estimate of air emissions of criteria pollutants, criteria precursors, and hazardous air pollutants from air emissions sources (EPA 2016d). Every 3 years, EPA publishes a NEI of air pollutant emissions for regulatory and air quality-modeling purposes (EPA 2016d). The NEI is based primarily upon data provided by state, local, and tribal air agencies for sources in their jurisdictions and supplemented by data developed by the EPA (EPA 2016d). The NEI for 2011 was the most recent at the time of the analysis for this report. Emissions in the NEI are provided at the county level and categorized broadly as point (PT) or nonpoint (NP) for stationary sources, and on-road (OR) or non-road (NR) for mobile sources (EPA 2016d):

- PT sources include larger sources that are located at a fixed, stationary location.
- NP sources include emissions estimates for sources that individually are too small in magnitude to report as point sources.
- OR sources include emissions from on-road vehicles that use gasoline, diesel, and other fuels.
- NR sources include off-road mobile sources that use gasoline, diesel, and other fuels.

Emissions from non-harvest and harvest activities belong to the NP and NR categories. Emissions from chemical-application emissions fall under the NP category. For biomass supplied to biorefineries, emissions from on-road transportation fall under the OR category, while emissions from preprocessing belong to the PT category.

We construct ratios (R) that represent comparisons of the total mass of relevant direct and/or precursor emissions of criteria air pollutants from the scenarios (see table 9.3) to 2011 emissions of the same pollutants (from the 2011 NEI) and term these "emission ratios." Estimated ratios (equations 9.3–9.8) from mass emissions are intended solely as comparisons to show how the magnitudes of criteria air pollutant (or precursors to criteria air pollutant) emissions from the *BT16* scenarios compare to the baseline emissions. The emission ratios do not account for the temporal profiles and chemical speciation for each emission source that are necessary to understand potential changes in air quality. Therefore, these ratios are not meant to predict changes in ambient air quality (e.g., ozone, $PM_{2.5}$ concentrations). However, because managing air quality must start with controlling emissions from the sources, these ratios could be useful in identifying areas of concern for local air quality management. See section 9.4.2 for further discussion of the limitations of our results to predict impacts on air quality.

Some criteria air pollutants are emitted directly by sources (e.g., CO); some are formed in the atmosphere (like ozone) through chemical reactions of pollutants directly emitted (called precursor pollutants); and some are generated both directly and indirectly (e.g., $PM_{2.5}$, PM_{10}, and SO_X). The emission ratios for precursors to ozone, $PM_{2.5}/PM_{10}$, as well as sulfur dioxide (SO_2), nitrogen dioxide (NO_2), and CO emissions are calculated using equations 9.3–9.8, respectively, and are reported in maps for all counties with cellulosic biomass feedstocks produced.

Emissions from non-harvest and harvest activities belong to the NP and NR categories. Emissions from chemical-application emissions fall under the NP category. For biomass supplied to biorefineries, emissions from on-road transportation fall under the OR category while emissions from preprocessing belong to the PT category.

The Clean Air Act requires EPA to set NAAQS for pollutants considered harmful to public health and the environment and identifies two types of these standards. Primary standards provide public health protection, including protecting the health of "sensitive" populations such as asthmatics, children, and the elderly. Secondary standards provide public welfare protection, including protection against decreased visibility and damage to animals, crops, vegetation, and buildings.

It can also be useful to place air pollutant emission estimates within the context of counties that are currently not in compliance with the NAAQS for criteria pollutants, as determined and published by EPA (EPA 2015d; EPA 2016c) and labeled as nonattainment areas (NAAs).[7] The concentrations of certain criteria pollutants are affected by emissions upwind, so we visually display all counties with emission ratios alongside those counties currently in nonattainment for applicable NAAQS. The locations of NAAs for 8-hr ozone, $PM_{2.5}$, SO_2, and PM_{10} NAAQS in 2016 are overlaid on the maps of the emission ratios in

[7] A nonattainment area is defined as any area that does not meet (or that contributes to ambient air quality in a nearby area that does not meet) the national primary or secondary ambient air quality standard for the pollutant (EPA 2016d; EPA 2016c).

Equation 9.3:

$$R_{\textit{Ozone Precursor Emissions}} = \frac{\Sigma\,(NO_x \textit{ and } VOC)_{\textit{all activities}}}{\Sigma\,(NO_x)_{NEI\ NR\ +NP+OR} + \Sigma\,(VOC)_{NEI\ NR\ +NP+OR+PT}}$$

Equation 9.4:

$$R_{\textit{PM2.5 Precursor Emissions}} = \frac{\Sigma(NO_x, SO_x, NH_3, PM_{2.5}, \textit{ and } VOC)_{\textit{all activities}}}{\Sigma(NO_x, SO_x, NH_3, \textit{ and } PM_{2.5})_{NEI\ NR\ +NP+OR} + \Sigma(VOC)_{NEI\ NR\ +NP+OR+PT}}$$

Equation 9.5:

$$R_{\textit{PM10 Precursor Emissions}} = \frac{\Sigma(NO_x, SO_x, NH_3, PM_{10}, \textit{ and } VOC)_{\textit{all activities}}}{\Sigma(NO_x, SO_x, NH_3, \textit{ and } PM_{10})_{NEI\ NR\ +NP+OR} + \Sigma(VOC)_{NEI\ NR\ +NP+OR+PT}}$$

Equation 9.6:

$$R_{SO2} = \frac{\Sigma(SO_x)_{\textit{all activities}}}{\Sigma(SO_x)_{NEI\ NR\ +NP+OR}}$$

Equation 9.7:

$$R_{SO2} = \frac{\Sigma(NO_x)_{\textit{all activities}}}{\Sigma(NO_x)_{NEI\ NR\ +NP+OR}}$$

Equation 9.8:

$$R_{SO2} = \frac{\Sigma(CO)_{\textit{all activities}}}{\Sigma(CO)_{NEI\ NR\ +NP+OR}}$$

section 9.3.3. Maps of SO_2 emission ratios are only included in appendix 9-A because SO_2 is not typically a mobile pollutant that will impact upwind counties. Emission ratios in NAAs are discussed in section 9.3.3. No counties were in nonattainment for NO_2 and CO NAAQS in 2016 (EPA 2016d); thus, we do not compare our results to the NAAQS for those pollutants.

9.3 Results

The estimated county-level air pollutant emissions for the scenarios by feedstock and activity category are documented in sections 9.3.1 and 9.3.2, respectively, focusing on the BC1&ML 2040 scenario. The results of emissions for each feedstock and activity category do not differ significantly among the BC1&ML 2017, BC1&ML 2040, and HH3&HH 2040 scenarios because equipment budgets and chemical application rates do not change across these scenarios; thus, the insights gained from analysis of the BC1&ML 2040 scenario show the same relative emissions for all feedstocks for all other *BT16* scenarios.

County-level emission ratios for BC1&ML 2017 and 2040 are discussed in section 9.3.3. In the HH3&HH 2040 scenario, the emission ratios of criteria air pollutant emissions from biomass production would be similar in magnitude and location to those for the BC1&ML 2040 scenario. The benefit of the HH3&HH 2040 scenario relative to the BC1&ML 2040 scenario is additional biomass production with relatively small increases in mass emissions. Since estimated emissions from biomass logistics are in part a function of the quantity of biomass supplied, biomass supply logistics in the HH3&HH 2040 scenario where more biomass is supplied to biorefineries could lead to large increases (>1.5x) in NO_2 and SO_2 emissions. However, most of these changes are in rural areas. See appendix 9-A section 9A.2.2 for visualization of results for the HH3&HH 2040 scenario in comparison to the BC1&ML 2040 scenario.

Section 9.3.4 documents supplemental discussion of criteria air pollutant emissions and includes comparisons of emissions from biomass crops to emissions from crude oil, discussion of upstream emissions, and potential changes to fugitive dust emissions from preprocessing equipment.

9.3.1 Comparison of Emissions per dt of Biomass by Feedstocks

9.3.1.1 Biomass Production

Figure 9.2 shows the variation in county-level air pollutant emissions in pounds (lb) per unit of potential biomass produced. Figure 9.2 illustrates emissions generated during biomass production from all counties and does not include emissions from the biomass supply logistics system.

Corn grain production generally requires greater inputs of fossil energy and agricultural chemicals than does the production of the cellulosic feedstocks evaluated in this chapter (EISA 2007; USDA 2013). As a result, it is not surprising that corn grain has the highest median air pollutant emissions for all pollutants examined, except for PM_{10} and $PM_{2.5}$ (fig. 9.2). For agriculture, this is largely attributable to residues not having emissions associated with field preparation (other than fertilizer compensation), and energy crops as perennials, for example, require only initial field preparation (not annual as for corn) and use lower quantities of fertilizers and pesticides. Corn also has wider ranges for all emissions compared to agricultural cellulosic feedstocks. This is primarily due to county-level variation in corn grain yield and irrigation requirements. However, the variability in regional corn grain chemical inputs, machinery operations, and tillage practices is also larger than for other feedstocks, based on *BT16* budget data.

PM_{10} and $PM_{2.5}$ emissions from straw residues are estimated to be larger than those of corn grain due to fugitive dust emissions. While corn grain produces a larger absolute amount of fugitive dust, the yield would be much lower for residues on a per-acre basis. Furthermore, the most applicable fugitive dust EFs we could find for wheat straw (see appendix 9-A, section 9A.2.5) are based on the activities associated with wheat production. Therefore, if we had evaluated a conventional straw-producing crop, such as wheat, using the same methodology that was used for estimating fugitive dust from wheat straw, then the fugitive dust emissions from wheat would be higher than that of wheat straw because wheat straw does not require field establishment and preparation.

Figure 9.2 also shows that criteria air pollutant emissions would be higher for agricultural residues than for energy crops for all emission species except VOCs. The fossil fuel inputs and chemical application rates for energy crops are generally higher than for the agricultural residues, but the harvest yields for the energy crops are much higher, so emissions normalized by unit of biomass produced would be lower (DOE 2016). Variations in emissions for the agricultural cellulosic feedstocks are mostly attributable to differences in estimated county-level yields and chemical application. Agricultural residues are estimated to have lower VOC emissions than energy crops due to the lack of a need for pesticide application associated with residues (DOE 2016).

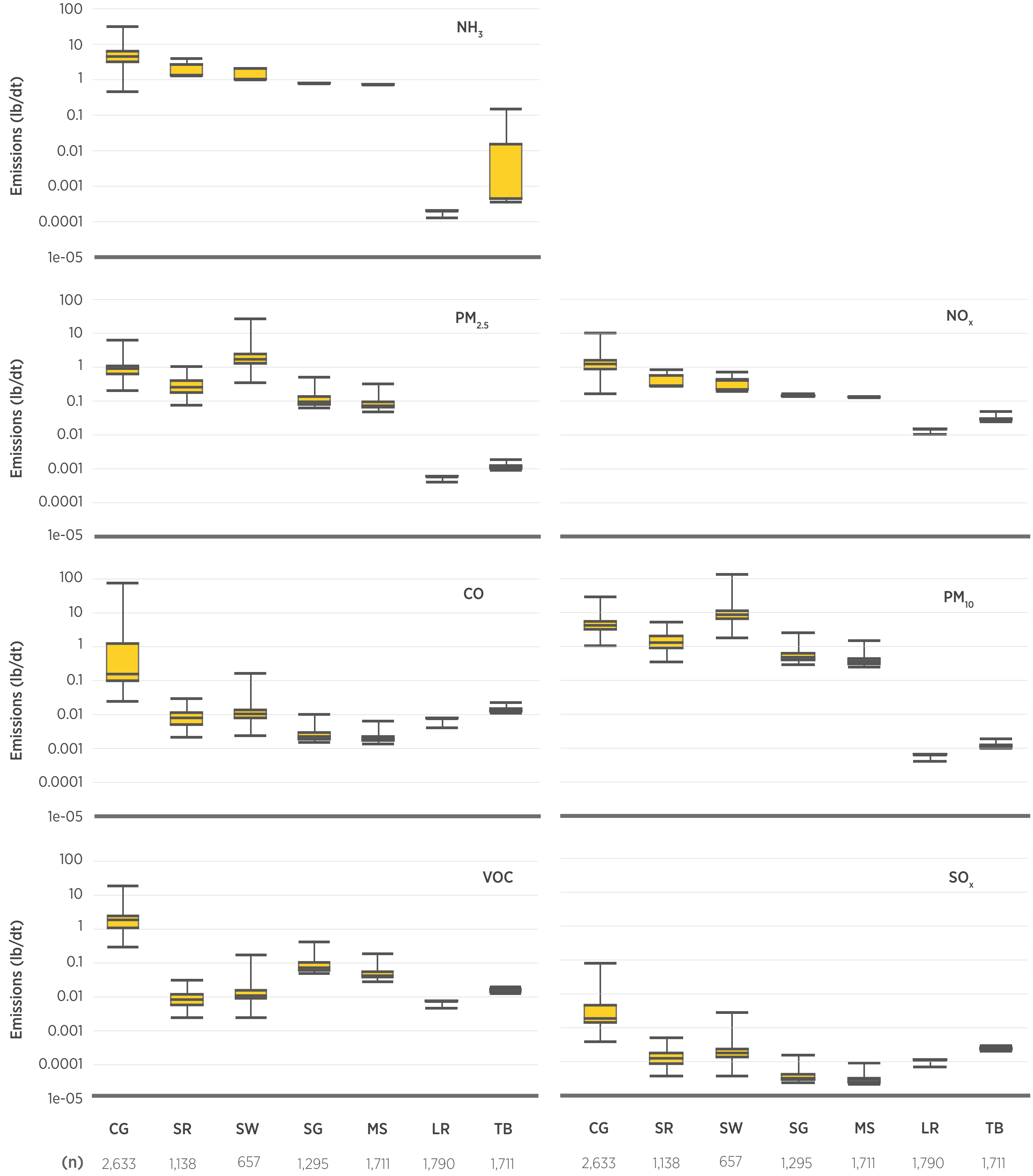
However, lower VOC emissions would not necessarily translate to lower air quality and human health impacts because fuel combustion, chemical (e.g., herbicide) application, and biomass drying emit very different VOC species and therefore may result in varying impacts on air quality. Given a lack of data (e.g., EPA's NEI reports non-speciated VOC emissions for herbicide applications), it is beyond the scope of this work to estimate speciated VOC emissions for these emission sources.

Unlike agriculture where one budget is assumed for each county for each crop, in forestry, several budgets are used in each county for whole-tree biomass from multiple wood types and forestry land types. Variation in whole-tree biomass emissions is due to variability in estimated county-level yields in each county, as well as variability in the equipment operations for establishment and harvest in each county (DOE 2016).

Among the feedstocks evaluated and shown in figure 9.2, logging residues would be estimated to have the lowest air pollutant emissions per unit of biomass for NH_3, NO_X, VOC, $PM_{2.5}$, and PM_{10}. However, it is important to note that $PM_{2.5}$ and PM_{10} emissions from logging residues and whole-tree biomass are not directly comparable to those of other feedstocks due to the lack of data on potential fugitive dust emissions for forestry activities. Still, these other emissions from logging residues are lowest among the types of feedstock due to the assumptions that no chemicals will be applied to compensate for the loss of nutrients from logging residue removal (EISA 2007) and that logging residues are ready for collection at the forest landing (i.e., no additional machinery operation is required for harvesting logging residues) (DOE 2016). Emissions of the remaining air pollutants, CO and SO_X, are higher for logging residues than for energy crops due to their relatively lower yields compared to agricultural cellulosic feedstocks.

With regard to whole-tree biomass, CO and SO_X emissions would be higher than other cellulosic feed-

Figure 9.2 | Distribution of county-level estimates (number of counties = n) of air pollutant emissions per unit of potential biomass produced in the BC1&ML 2040 scenario. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



Acronyms: dt – dry ton; lb – pounds; CO – carbon monoxide; NH_3 – ammonia; NO_x – oxides of nitrogen; PM – particulate matter; SO_x – oxides of sulfur; VOC – volatile organic compounds; CG – corn grain; LR – logging residues; MS – miscanthus; SG – switchgrass; SR – stover; SW – straw; TB – whole-tree biomass stocks due to the higher overall fuel consumption by equipment to establish, harvest, and chip whole-tree biomass. However, NH_3 and NO_X emissions would be lower relative to other cellulosic feedstocks. Only a small subset of softwood whole-tree biomass would require chemical inputs; in most counties, there were few acres established as plantations and therefore did not require chemical applications (DOE 2016). On average, whole-tree biomass has the highest annual per-acre yields relative to the other cellulosic feedstocks we have evaluated in this chapter (DOE 2016).

9.3.1.2 Biomass Supply Logistics

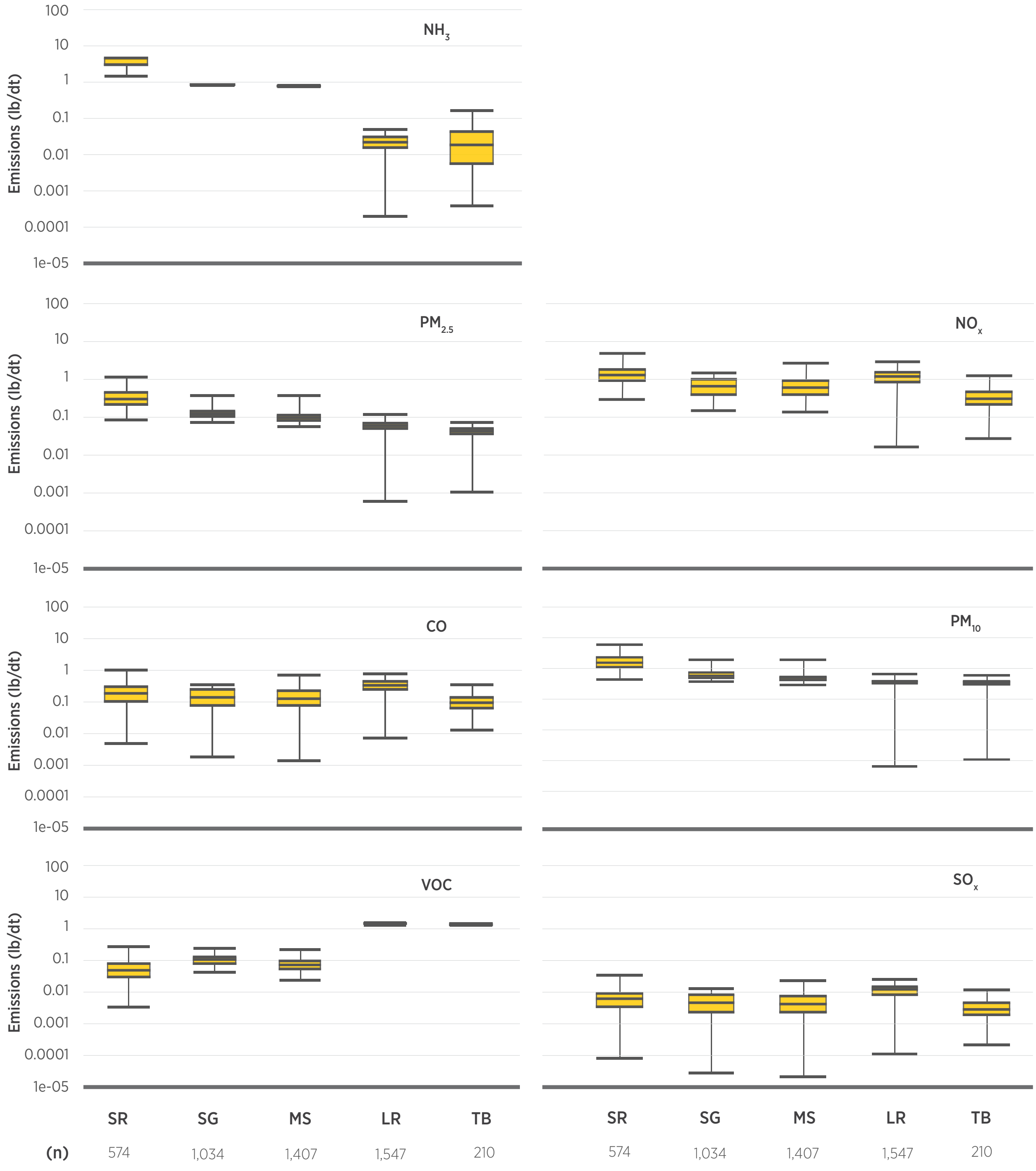
Figure 9.3 shows the estimated variation in county-level air pollutant emissions in pounds per unit of potential biomass produced and supplied to a biorefinery in the BC1&ML 2040 scenario. As noted in section 9.2.1, only a subset of feedstocks and counties (number of counties = n in the figure) are used in the logistics component of the biomass supply scenarios. For example, no corn grain or wheat straw is supplied to biorefineries in any of the biomass supply scenarios (DOE 2016). Despite this limitation, we examined the total emissions generated from potential biomass production and supply logistics for those counties and feedstocks that were represented in the biomass supply scenarios. All on-road transportation emissions are allocated to the biomass-supplying county, so these results should be considered as potentially over-estimating emissions in a county with long transportation distances.

Figure 9.3 illustrates estimated air-pollutant emissions from the BC1&ML 2040 scenario when including both production and the later supply chain elements of on-road transportation and preprocessing for several air pollutants. The most noticeable change across air pollutants is that the inclusion of on-road transportation and preprocessing would significantly increase the variability in emissions across counties. This increased variability is attributable to the distances traveled by biomass produced in a given county.

On-road transportation emissions estimated in FPEAM on a per dt basis are a major source of NO_X, CO, and SO_X emissions (see section 9.3.2), so the differences between emissions from cellulosic feedstocks become small. The most noticeable remaining difference between cellulosic feedstocks is that NO_X, CO, and SO_X emissions from logging residues would be higher than from other biomass feedstocks. High emissions from on-road transportation of logging residues are due to two factors: longer travel distances and lower truck fuel economy. Logging residues are a relatively low-cost cellulosic feedstock to produce and use at biorefineries (DOE 2016). Because of low production costs, logging residues could travel longer distances (i.e., increased transportation costs) and still fall within the $100 per unit of biomass cutoff for the supply logistics scenario. On average, a dt of logging residues priced at less than $100 per dt would travel 3–4 times farther than other cellulosic feedstocks. In the *BT16* supply budget data, the trucks transporting any woody biomass have a nearly 15% lower fuel efficiency than trucks used for other biomass feedstocks.

VOC emissions by agriculture residues and herbaceous energy crops per dt would not be significantly changed with the accounting of on-road transport because VOC emissions from pesticides dominate emissions. The inclusion of preprocessing emissions significantly increases VOC emissions for potential logging residues and additional whole-tree biomass because pesticides are only applied to softwoods in some counties.

Figure 9.3 | Distribution of county-level estimates (number of counties = n) of air pollutant emissions per unit of potential biomass that is both produced and supplied to biorefineries[8] for BC1&ML 2040 scenario. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



Acronyms: dt – dry ton; lb – pounds; CO – carbon monoxide; NH_3 – ammonia; NO_x – oxides of nitrogen; PM – particulate matter; SO^x – oxides of sulfur;. VOC – volatile organic compounds; CG – corn grain; LR – logging; MS – miscanthus; SG – switchgrass; SR – stover; SW – straw; TB – wholetree biomass.

[8] Only a subset of biomass produced is being supplied to biorefineries in the scenarios examined as a part of *BT16* and therefore, in a given county, potential biomass produced may not be used for biofuel production (DOE 2016). For example, wheat straw and corn grain are not supplied to biorefineries in the scenarios.

Emissions from transportation comprise a large portion of the estimated total emissions for whole-tree biomass. Relative to biomass production only, accounting for on-road transportation and preprocessing did not lead to significant changes in NH_3, $PM_{2.5}$, and PM_{10} emitted per unit of biomass by each feedstock. Logging residues and whole-tree biomass emissions noticeably increase when accounting for transportation and preprocessing due to the low emission from biomass production. Emissions from biomass production are low because of the limited chemical application and the lack of fugitive dust emission estimates in the forestry sector for this analysis.

9.3.2 Emissions Contribution by Activity Category

9.3.2.1 Biomass Production

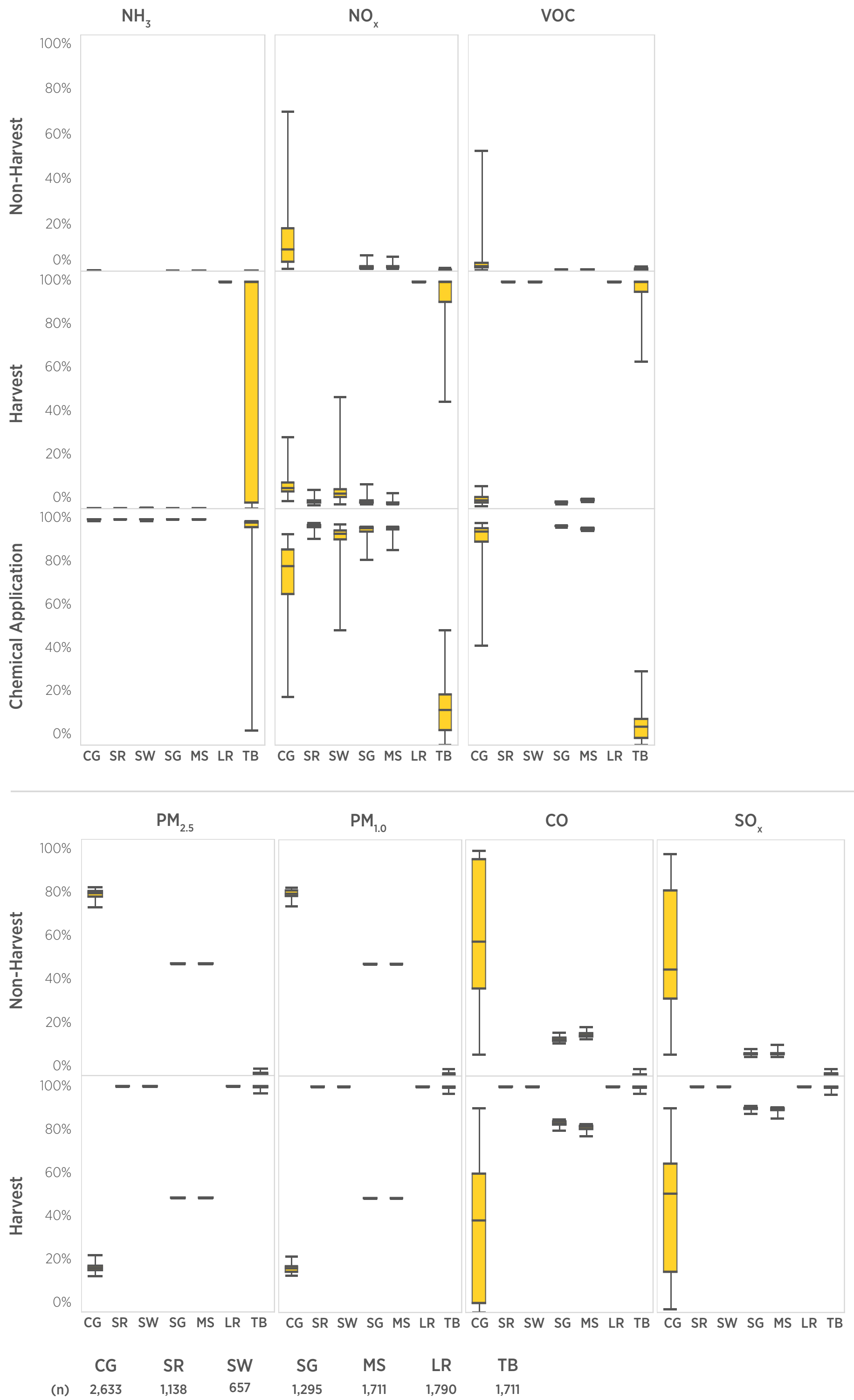
Figure 9.4 shows the distribution of each activity category's relative contribution to the projected total mass of emitted air pollutants, per pollutant and feedstock. Figure 9.4 evaluates the relative contribution of emissions by activity category for biomass production from all counties and does not include emissions from biomass supply logistics.

Figure 9.4 shows that virtually all NH_3 emissions would be attributable to nitrogen fertilizer for agricultural feedstocks, with minimal contribution from fuel use. Nitrogen fertilizer application is also the major contributor to NO_X emissions from agricultural feedstocks. NH_3 and NO_X emissions for logging residues from chemicals are zero because fertilizer inputs are not required. Many counties producing whole-tree biomass do not require nitrogen fertilizer inputs, and therefore, NH_3 and NO_X emissions would be much more variable, depending on whether or not nitrogen fertilizer is applied to whole-tree biomass in a given county.

The use of pesticides for corn grain, miscanthus, switchgrass, and whole-tree biomass on softwoods in some counties in *BT16* scenarios would contribute to the majority of VOC emissions from those counties, as shown in figure 9.4. However, variability is wide for corn grain because of considerable variation in pesticide usage among corn-producing counties. Variability in VOC emissions from whole-tree biomass is also high relative to other cellulosic feedstocks because only softwoods in some counties are assumed to require pesticides as per the budget data (DOE 2016). For stover and straw, all VOC emissions are attributable to machinery operations; this is because pesticide application is not attributed to residues but instead attributed to the conventional crop such as corn grain and wheat when using product purpose allocation.

The primary emission sources for PM_{10} and $PM_{2.5}$ are identical, so they are discussed collectively as "PM." For agricultural feedstocks, the two contributing sources of PM emissions are (1) equipment's fuel usage; and (2) fugitive dust emissions, with the latter dominating. Field preparation and tillage, planting crop maintenance, harvest, and off-road transportation all generate fugitive dust. For corn grain, harvest activities are the major contributor to PM emissions because the process of harvest and collection generates large amounts of fugitive dust. For stover and straw, fugitive dust emissions are attributable to harvest because fugitive dust from agricultural tilling (e.g., cultivating, fertilizer application) is allocated exclusively to grains (e.g., corn). Switchgrass and miscanthus are assumed to be rain-fed and require much-less-intensive tillage on a 10-year rotation, and thus, PM emissions are split between non-harvest and harvesting activities (DOE 2016). A method for estimating fugitive dust emissions for whole-tree biomass was not found in the literature; all PM emissions are from equipment fuel use. This data gap is discussed further in section 9.4.2.3.

Figure 9.4 | Distribution of county-level estimates (number of counties = n) of the fraction of aggregated emissions from three categories of emitting activities. Estimates are for potential biomass produced for the BC1&ML 2040 scenario. Blanks indicate no emissions from that activity category for that feedstock and pollutant. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



Acronyms: CO – carbon monoxide; NH_3 – ammonia; NO_x – oxides of nitrogen; PM – particulate matter; SO_x – oxides of sulfur; VOC – volatile organic compounds; CG – corn grain; LR – logging; MS – miscanthus; SG – switchgrass; SR – stover; SW – straw; TB – wholetree biomass.

Equipment fuel use accounts for all CO and SO_X emissions across all feedstocks. Corn grain emissions are highly variable, reflecting the regional variability in fuel type used by irrigation equipment (USDA 2009). Switchgrass and miscanthus require establishment only once in their multiyear rotations and do not require irrigation. As a result, harvest is responsible for most CO and SO_X emissions compared to non-harvest activities for those feedstocks. CO and SO_X emissions associated with non-harvest activities are exclusively allocated to the primary products (e.g., corn grain) rather than agricultural residues. Logging residues do not have non-harvest activities, and most CO and SO_X emissions from whole-tree biomass are attributable to harvest activities.

9.3.2.2 Biomass Supply Logistics

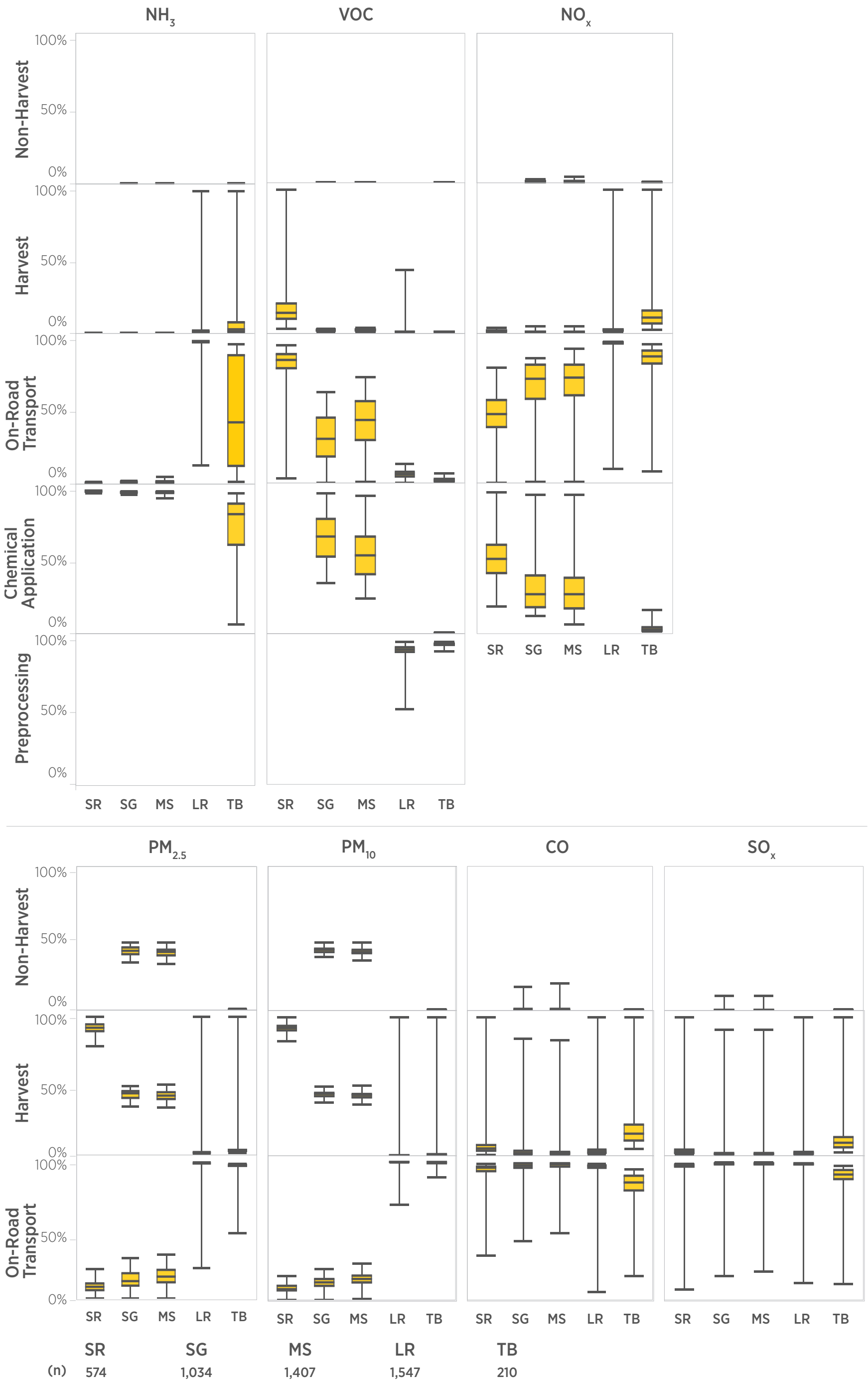
Figure 9.5 shows the distribution of each activity category's relative contribution to the total mass of air pollutants, per pollutant and feedstock, emitted in the BC1&ML 2040 scenario. Figure 9.5 illustrates the relative contribution of emissions by activity category for both biomass production and biomass supply logistics but only for the subset of biomass-supplying counties (number of counties = n in the figure) that were evaluated in the *BT16* supply-logistics scenarios. For example, no corn grain or wheat straw is supplied to biorefineries in any of the *BT16* biomass supply scenarios in this report. We examined the total emissions generated from production and supply logistics for those counties and feedstocks that were represented in the biomass supply scenarios. All on-road transportation emissions are allocated to the biomass-supplying county, so these results should be considered as potentially overestimating emissions in a county with long transportation distances.

Figure 9.5 shows that relative to other sources, on-road transportation would be a major source of many emissions—except for NH_3, PM_{10}, and $PM_{2.5}$—for agricultural cellulosic biomass. The application of pesticides was often the most important source of VOC emissions that we evaluated, but NO_X emissions from transportation were often larger for a single biomass-supplying county than emissions from fertilizer. On-road transportation is the major contributor to SO_X and CO emissions. Fugitive dust from agricultural biomass harvest activities remains the major contributor to overall PM_{10} and $PM_{2.5}$ emissions, and fertilizer application remains the major contributor to overall NH_3 emissions from biomass production and supply activities.

Relative to other sources of emissions, on-road transportation emissions would be a major, if not *the major*, source of all emissions for logging residues and whole-tree biomass in the scenarios evaluated. PM and VOCs are the exceptions because fugitive dust from whole-tree biomass was not evaluated and chemical application in the forestry sector was limited to softwoods based on the *BT16* budget data. The major source of VOC emissions from logging residues and trees are drying and preprocessing, but conclusions from these results should be constrained as noted in section 9.4.2.1 because of the limits of available, robust emission rate data.

Figure 9.6 shows county-level scatter plots of total distance traveled by stover to supply biorefineries and the emissions that would be generated per unit of biomass for transporting that biomass. As distance increases, emissions generally increase, as indicated by trends in figure 9.6. This figure also indicates that relative to the near-term system, the long-term feedstock supply logistics system reduces emissions for the same distance traveled through biomass densification. A regression line was fit to the data in figure 9.6. The regression shows a good fit (R-squared = 98%) for the near-term logistics system and a less good fit (R-squared = 78%) for the long-term system. Increased variability in the long-term logistics system reflects reduced emissions from fuel use and increased importance of more variable fugitive dust emissions. Fugitive dust emissions are highly variable due to the variability in assumptions about local conditions (e.g., climate, on-road traffic, silt loading) for fugitive dust estimates.

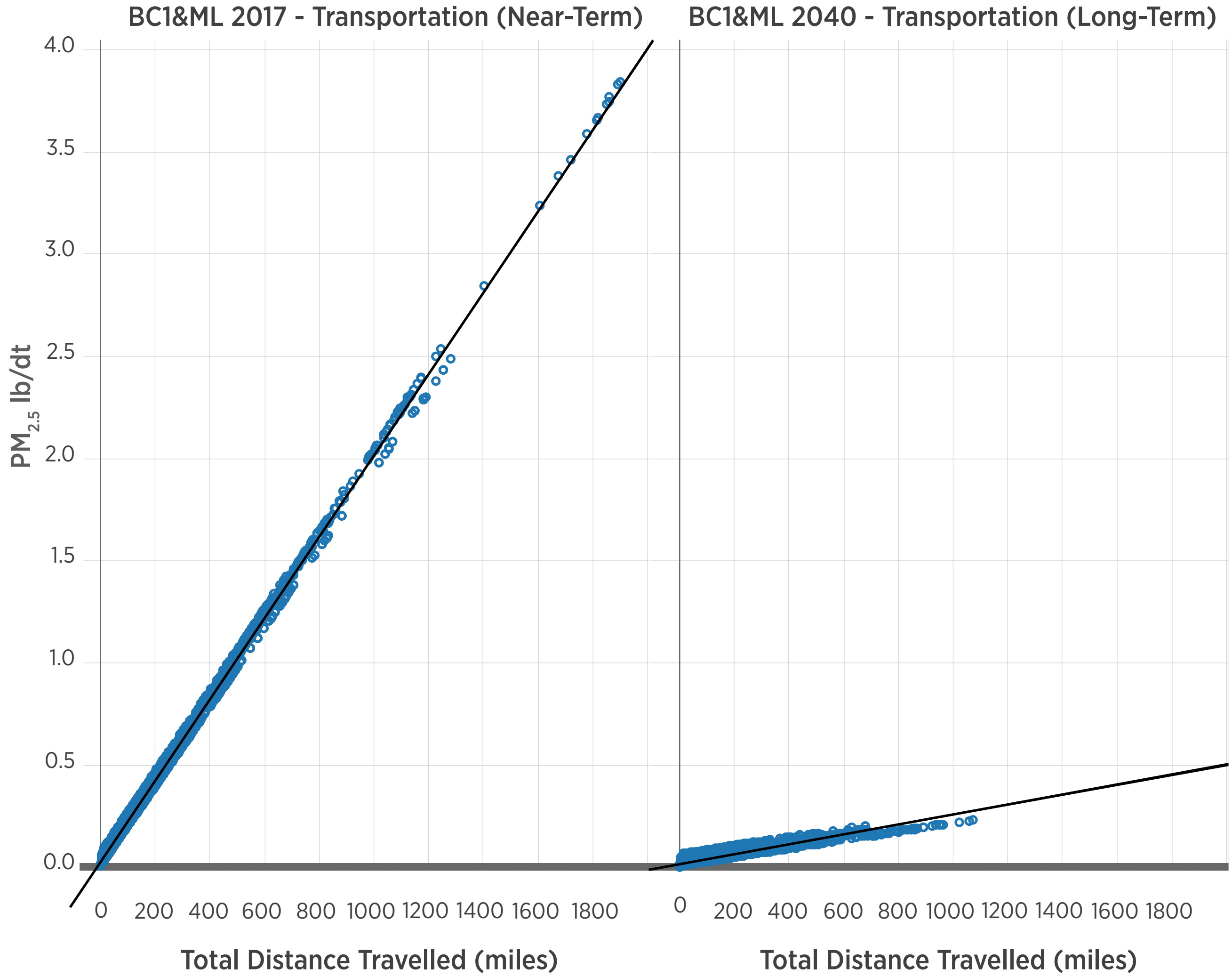
Figure 9.5 | Distribution of county-level estimates (number of counties = n) of the fraction of aggregated mass emissions from five categories of emitting activities. Estimates are for potential biomass produced and supplied[9] for the BC1&ML 2040 scenario. Blanks indicate no emissions from that activity category for that feedstock and pollutant. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



Acronyms: CO – carbon monoxide; NH_3 – ammonia; NO_x – oxides of nitrogen; PM – particulate matter; SO_x – oxides of sulfur; VOC – volatile organic compounds; CG – corn grain; LR – logging; MS – miscanthus; SG – switchgrass; SR – stover; SW – straw; TB – wholetree biomass.

[9] Only a subset of biomass produced is being supplied to biorefineries in the scenarios examined as a part of *BT16* and therefore, in a given county, potential biomass produced may not be used for biofuel production (DOE 2016). For example, wheat straw and corn grain are not supplied to biorefineries.

Figure 9.6 | County-level scatterplot of total distance traveled by cellulosic biomass being supplied to biorefineries (x-axis) and $PM_{2.5}$ emissions per dt (y-axis) for BC1&ML 2017 and 2040 near-term and long-term supply logistics scenarios.



Acronyms: dt – dry ton; lb – pounds.

9.3.3 Comparison of Estimated Emissions Inventory to the NEI and NAAQS NAAs

An increase in air pollutant emissions, especially in the context of emission growth in sectors other than biomass production, can be problematic for counties already not in compliance (so-called, nonattainment) with the NAAQS. Furthermore, owing to the importance of atmospheric transport to the local concentrations of many air pollutants, air pollutant emissions from upwind counties could further deteriorate air quality for counties already in nonattainment. Transport distances can be as large as 600 miles for precursors to $PM_{2.5}$ and 60 miles for ozone. Though the specific threshold of a potential emission increase that would be meaningful relates to local air quality, in the context of ever-tightening air quality standards, air quality managers might be concerned about relatively small increases in emissions. Our results are reported in a way that is intended to help inform air quality managers about air emissions from potential biomass production that could be translated into locally relevant decision factors.

The first panels in figures 9.7–9.9 display distributions of the emission ratios comparing the inventory to the 2011 NEI for each NAAQS criteria air pollutant based on mass air pollutant emissions estimated for the *BT16* scenarios. Results are presented for precursors to ozone, $PM_{2.5}$, and PM_{10} (table 9.3), as well as for SO_2, NO_2, and CO emissions. Distributions are shown for counties in attainment or nonattainment with the NAAQS. Any increase in emissions has the potential to contribute to air quality degradation in or upwind of a county, but of particular interest are those counties whose emission ratios are potentially greater than a threshold (Zhang et al. 2016). An emission ratio above 1% is suggested as a threshold that any county might consider as potentially significant. An emission ratio greater than 1% does not indicate that air quality degradation will occur, but that emissions in those counties warrant further analysis by air quality managers in the context of a reference scenario to determine the potential for air quality degradation in or upwind of that county. Counties in nonattainment whose emission ratios are above the suggested threshold of 1% are considered among the most at-risk for potential air quality degradation.

The maps in figures 9.7–9.9 display the emission ratios for each NAAQS criteria air pollutant along with locations of NAAs for these pollutants as of 2015 (EPA 2016d). NAAs are designated based on the currently enforced primary standards[10] for ozone (8-hour standard), $PM_{2.5}$ (24-hour and 1-year), PM_{10} (24-hour), SO_2 (1-hour), NO_2 (1-hour and 1-year), and CO (8-hour and 1-hour) NAAQS. Increases in emissions even in counties in attainment for NAAQS could impact NAAs downwind, owing to atmospheric transport.

This chapter focuses discussion on emission ratios for ozone, $PM_{2.5}$, and PM_{10} in the context of counties in NAAs. No county is out of compliance with the current NO_2 and CO NAAQS (EPA 2016b). SO_2 is not transported upwind, so we only discuss emission ratios for SO_2 in NAAs. For additional results for SO_2, NO_2, and CO, please refer to appendix 9-A, section 9A.3.1.

9.3.3.1 Counties Upwind from NAAs

Figures 9.7–9.9 show that in the BC1&ML 2017 and 2040 scenarios, about 25% of the total number of counties evaluated (~3,000) in attainment with the NAAQS for ozone, $PM_{2.5}$ and PM_{10} have emission ratios greater than 1% for each pollutant. In the BC1&ML scenarios, the upper quartile of county-level emission ratios for ozone range from 0.8% to 10% in 2017 and 0.7% to 8% in 2040. In the BC1&ML scenarios, the upper quartile of county-level emission ratios for $PM_{2.5}$ range from 0.9% to 10% in 2017 and 2% to 10% in 2040 with many counties having emission ratios above 1% in both years. In the BC1&ML scenarios, the upper quartile of county-level emission ratios for PM_{10} range from 0.9% to 8% in 2017 and 2% to 11% in 2040. We visually display all counties with emission ratios alongside those counties currently in nonattainment with applicable NAAQS because air quality in any location could be affected by emissions upwind.

Figure 9.7 shows areas in nonattainment with ozone NAAQS that are upwind (on the order of 60 miles) of multiple counties with ozone precursor emission ratios greater than 1% in BC1&ML 2017 and 2040 scenarios. In 2017, these areas include the city of Chicago, Illinois (eleven counties); Cincinnati, Ohio (nine counties); and Columbus, Ohio (six counties). In 2040, areas with nonattainment counties adjacent to multiple attainment counties with emission ratios greater than 1% include the city of Chicago, Illinois

[10] There are also secondary standards intended to provide public welfare protection against decreased visibility and damage to animals, crops, vegetation, and buildings rather than health. These secondary standards are not considered in our analysis.

(eleven counties); St. Louis, Missouri (eight counties); and Memphis, Arkansas (three counties). The majority of these counties have potential agricultural residue production in 2017 and 2040 scenarios and energy crop production in the 2040 scenario. As a result, the emission ratios above 1% are largely attributable to NO_X and VOC emissions from fertilizer and pesticide application as well as NO_X emissions from transportation.

Figures 9.8 and 9.9 show areas in nonattainment with $PM_{2.5}$ and PM_{10} NAAQS that are upwind (on the order of 600 miles) of multiple counties with $PM_{2.5}$ and PM_{10} precursor emission ratios greater than 1% in BC1&ML 2017 and 2040 scenarios. For $PM_{2.5}$ estimated for the 2017 scenario, these upwind counties are located around the city of Louisville, Kentucky (four counties); Lane and Klameth Counties, Oregon; Lincoln County, Montana; and Shoshone County, Idaho. For $PM_{2.5}$ in 2040 these upwind counties are located around the city of St. Louis, Missouri (eight counties); the city of Louisville, Kentucky (four counties); the city of Cleveland, Ohio (two counties); and Lincoln County, Montana. For PM_{10} estimated in 2017, the upwind county is Lane County, Oregon. For PM_{10} in 2040, upwind counties include Shoshone County, Idaho, and five counties in northwest Montana. The high $PM_{2.5}$ and PM_{10} emission ratios in these areas are largely attributable to three sources: (1) the application of fertilizers and pesticides, which contribute to changes in PM precursor emissions (NH_3, NO_X, and VOC); (2) fugitive dust emissions from the use of agricultural equipment, which contribute to $PM_{2.5}$; and (3) NO_X and SO_X emissions from transportation of any biomass, which are PM precursor emissions (table 9.3).

If future biomass production sources of air pollutants are additional and do not displace current biomass production sources (see section 9.4), air pollutant emissions from these sources may pose challenges for compliance with the NAAQS in these selected areas. The emission estimates provided in this study could help inform long-term air quality planning, such as state implementation plans, which are required to consider new emission sources for future scenarios.

9.3.3.2 Counties in NAAs

Figure 9.7 shows how the locations of counties in nonattainment for ozone with emission ratios greater than 1% for ozone precursors differ by year for the BC1&ML 2017 and 2040 and HH3&HH 2040 scenarios. For the BC1&ML 2017 scenario, the nonattainment counties with emission ratios estimated to be greater than 1% are Kings and Tulane counties in California, Madison and Knox counties in Ohio, and Kane County in Illinois. The emissions in 2017 would be primarily concentrated in counties with agricultural residue production, with the exception being Knox County, Ohio, where forestry biomass would be a major contributor to ozone precursor emissions (VOC and NO_X). However, for the BC1&ML 2040 scenario, the non-attainment counties with ozone precursor emission ratios greater than 1% have shifted to St. Claire and Monroe counties, Illinois; and Crittenden County, Arkansas. In the HH3&HH 2040 scenario, the additional counties in NAA estimated to have ozone emission ratios greater than 1% are Grundy and Kendall counties in Illinois; and Madison, Clinton, Fairfield, and Knox counties, Ohio; Crittenden County, Arkansas; and Hamilton County, Texas.

Similarly, figure 9.8 shows how the locations of counties in NAAs for $PM_{2.5}$ with emission ratios greater than 1% for $PM_{2.5}$ vary by year for the BC1&ML scenarios. In 2017, these areas are in Kings, Tulare, and Merced counties, California; Lincoln County, Montana; and Shoshone County, Idaho. However, in the BC1&ML 2040 scenario, non-attainment counties with $PM_{2.5}$ emission ratios higher than 1% are Monroe, St. Claire, and Randolph counties in Illinois, as well as Franklin County in Missouri. In the HH3&HH 2040 scenario, no additional counties in NAAs are estimated to have $PM_{2.5}$ emission ratios greater than 1%.

Shifts in both ozone precursors as well as primary and secondary $PM_{2.5}$ emissions from 2017 to 2040 scenarios are due to a combination of several factors. In 2040, decreased wholetree biomass production (e.g., Knox and Lincoln counties), higher agricultural residue yields (e.g., Kings, Monroe, and St. Claire counties), and decreases in the average distance of biomass on-road transportation using the long-term logistics system (most counties) would reduce the ozone and $PM_{2.5}$ emission ratios in non-attainment counties with ratios greater than 1% in 2017 to less than 1% in 2040. Increased potential energy crop production in combination with continuing agricultural residue production, in the nonattainment counties in 2040 would lead to ozone and $PM_{2.5}$ precursor emission ratios greater than 1% in a different set of counties relative to 2017. Additional differences in emission ratios between the BC1&ML 2040 and the HH3&HH 2040 scenario are due to additional transportation to biorefineries of potential additional biomass produced.

Figure 9.9 shows how the locations of counties in NAAs for PM_{10} with emission ratios greater than 1% for PM_{10} vary by year for the BC1&ML and HH3&HH scenarios. In 2017, these areas are in Lincoln County, Montana, and Shoshone County, Idaho. However, in the BC1&ML 2040 scenario, Flathead County, Montana, is the only nonattainment county with a PM_{10} with an emission ratio higher than 1% and Flathead County, Montana, and Power County, Idaho are nonattainment counties with a PM_{10} emission ratios higher than 1% in the HH3&HH scenarios. Changes in the emission ratio across these scenarios reflect decreased forestry biomass use for energy (e.g., in Montana and Idaho) and increased fugitive dust from agricultural residues, in particular straw, in Power County, Idaho.

The emissions that would be generated in counties with emission ratios greater than 1% for ozone and PM_{10} can generally be attributed to a few primary sources. Emissions from counties with high quantities of agricultural-residue production and emission ratios greater than 1% for ozone would be largely attributable to NO_X and VOC emissions from chemical application and on-road biomass transportation. Greater than 1% PM_{10} emission ratios would be attributable mostly to fertilizer and pesticide applications, contributing to PM precursor emissions (NH_3, NO_X, and VOC), fugitive dust emissions from the use of agricultural equipment that contribute to PM_{10}, as well as NO_X and SO_X emissions from biomass on-road transportation. In addition, emissions from whole-tree biomass are largely attributable to on-road biomass transportation.

The results for emission ratios in NAAs for SO_2 differ from those for ozone and PM. For SO_2, only partial counties are in NAAs, so local air quality and transportation modeling would be needed to understand biomass transportation through NAAs in the county. For the BC1&ML 2017 scenario, only Muscatine County, Iowa, has an emission ratio that is greater than 1% for SO_2. For the BC1&ML 2040 scenario, in Muscatine County, Iowa, and Pike County, Indiana, emission ratios are greater than 1%. In the 2040 HH3&HH scenario, Muscatine County, Iowa; Pike County, Indiana; and Tazewell, Illinois, the NEI ratio is greater than 1%. The emission ratios in Muscantine, Pike, and Tazewell counties are largely attributable to transporting stover (i.e., up to 20 miles) and miscanthus to surrounding counties (i.e., up to 80 miles), respectively.

9.3.3 Comparison of Estimated Emissions Inventory to the NEI and NAAQS NAAs

An increase in air pollutant emissions, especially in the context of emission growth in sectors other than biomass production, can be problematic for counties already not in compliance (so-called, nonattainment) with the NAAQS. Furthermore, owing to the importance of atmospheric transport to the local concentrations of many air pollutants, air pollutant emissions from upwind counties could further deteriorate air quality for counties already in nonattainment. Transport distances can be as large as 600 miles for precursors to $PM_{2.5}$ and 60 miles for ozone. Though the specific threshold of a potential emission increase that would be meaningful relates to local air quality, in the context of ever-tightening air quality standards, air quality managers might be concerned about relatively small increases in emissions. Our results are reported in a way that is intended to help inform air quality managers about air emissions from potential biomass production that could be translated into locally relevant decision factors.

The first panels in figures 9.7–9.9 display distributions of the emission ratios comparing the inventory to the 2011 NEI for each NAAQS criteria air pollutant based on mass air pollutant emissions estimated for the *BT16* scenarios. Results are presented for precursors to ozone, $PM_{2.5}$, and PM10 (table 9.3), as well as for SO_2, NO_2, and CO emissions. Distributions are shown for counties in attainment or nonattainment with the NAAQS. Any increase in emissions has the potential to contribute to air quality degradation in or upwind of a county, but of particular interest are those counties whose emission ratios are potentially greater than a threshold (Zhang et al. 2016). An emission ratio above 1% is suggested as a threshold that any county might consider as potentially significant. An emission ratio greater than 1% does not indicate that air quality degradation will occur, but that emissions in those counties warrant further analysis by air quality managers in the context of a reference scenario to determine the potential for air quality degradation in or upwind of that county. Counties in nonattainment whose emission ratios are above the suggested threshold of 1% are considered among the most at-risk for potential air quality degradation.

The maps in figures 9.7–9.9 display the emission ratios for each NAAQS criteria air pollutant along with locations of NAAs for these pollutants as of 2015 (EPA 2016d). NAAs are designated based on the currently enforced primary standards[10] for ozone (8-hour standard), $PM_{2.5}$ (24-hour and 1-year), PM_{10} (24-hour), SO_2 (1-hour), NO_2 (1-hour and 1-year), and CO (8-hour and 1-hour) NAAQS. Increases in emissions even in counties in attainment for NAAQS could impact NAAs downwind, owing to atmospheric transport.

This chapter focuses discussion on emission ratios for ozone, $PM_{2.5}$, and PM_{10} in the context of counties in NAAs. No county is out of compliance with the current NO_2 and CO NAAQS (EPA 2016b). SO_2 is not transported upwind, so we only discuss emission ratios for SO_2 in NAAs. For additional results for SO_2, NO_2, and CO, please refer to appendix section 9A.3.1.

9.3.3.1 Counties Upwind from NAAs

Figures 9.7–9.9 show that in the BC1&ML 2017 and 2040 scenarios, about 25% of the total number of counties evaluated (~3,000) in attainment with the NAAQS for ozone, $PM_{2.5}$ and PM_{10} have emission ratios greater than 1% for each pollutant. In the BC1&ML scenarios, the upper quartile of county-level emission ratios for ozone range from 0.8% to 10% in 2017 and 0.7% to 8% in 2040. In the BC1&ML scenarios, the upper quartile of county-level emission ratios for $PM_{2.5}$ range from 0.9% to 10% in 2017 and 2% to 10% in 2040 with many counties having emission ratios above 1% in both years. In the BC1&ML scenarios, the upper quartile of county-level emission ratios for PM_{10} range from 0.9% to 8% in 2017 and 2% to 11% in 2040. We visually display all counties with emission ratios alongside those counties currently in nonattainment with applicable NAAQS because air quality in any location could be affected by emissions upwind.

[10] There are also secondary standards intended to provide public welfare protection against decreased visibility and damage to animals, crops, vegetation, and buildings rather than health. These secondary standards are not considered in our analysis.

Figure 9.7 shows areas in nonattainment with ozone NAAQS that are upwind (on the order of 60 miles) of multiple counties with ozone precursor emission ratios greater than 1% in BC1&ML 2017 and 2040 scenarios. In 2017, these areas include the city of Chicago, Illinois (eleven counties); Cincinnati, Ohio (nine counties); and Columbus, Ohio (six counties). In 2040, areas with nonattainment counties adjacent to multiple attainment counties with emission ratios greater than 1% include the city of Chicago, Illinois (eleven counties); St. Louis, Missouri (eight counties); and Memphis, Arkansas (three counties). The majority of these counties have potential agricultural residue production in 2017 and 2040 scenarios and energy crop production in the 2040 scenario. As a result, the emission ratios above 1% are largely attributable to NO_X and VOC emissions from fertilizer and pesticide application as well as NO_x emissions from transportation.

Figures 9.8 and 9.9 show areas in nonattainment with $PM_{2.5}$ and PM_{10} NAAQS that are upwind (on the order of 600 miles) of multiple counties with $PM_{2.5}$ and PM_{10} precursor emission ratios greater than 1% in BC1&ML 2017 and 2040 scenarios. For $PM_{2.5}$ estimated in 2017, these upwind counties are located around the city of Louisville, Kentucky (four counties); Lane and Klameth Counties, Oregon; Lincoln County, Montana; and Shoshone Count, Idaho. For $PM_{2.5}$ in 2040 these upwind counties are located around the city of St. Louis, Missouri (eight counties); the city of Louisville, Kentucky (four counties); the city of Cleveland, Ohio (two counties); and Lincoln County, Montana. For PM_{10} estimated in 2017, the upwind county is Lane County, Oregon. For PM_{10} in 2040, upwind counties include Shoshone County, Idaho, and five counties in northwest Montana. The high $PM_{2.5}$ and PM_{10} emission ratios in these areas are largely attributable to three sources: (1) the application of fertilizers and pesticides, which contribute to changes in PM precursor emissions (NH_3, NO_X, and VOC); (2) fugitive dust emissions from the use of agricultural equipment, which contribute to $PM_{2.5}$; and (3) NO_X and SO_X emissions from transportation of any biomass, which are PM precursor emissions (table 9.3).

If future biomass production sources of air pollutants are additional and do not displace current biomass production sources (see section 9.4), air pollutant emissions from these sources may pose challenges for compliance with the NAAQS in these selected areas. The emission estimates provided in this study could help inform long-term air quality planning, such as state implementation plans, which are required to consider new emission sources for future scenarios.

9.3.3.2 Counties in NAAs

Figure 9.7 shows how the locations of counties in nonattainment for ozone with emission ratios greater than 1% for ozone precursors differ by year for the BC1&ML 2017 and 2040 and HH3&HH 2040 scenarios. For the BC1&ML 2017 scenario, the nonattainment counties with emission ratios estimated to be greater than 1% are Kings and Tulane counties in California, Madison and Knox counties in Ohio, and Kane County in Illinois. The emissions in 2017 would be primarily concentrated in counties with agricultural residue production, with the exception being Knox County, Ohio, where forestry biomass would be a major contributor to ozone precursor emissions (VOC and NO_X). However, for the BC1&ML 2040 scenario, the non-attainment counties with ozone precursor emission ratios greater than 1% have shifted to St. Claire and Monroe counties, Illinois; and Crittenden County, Arkansas. In the HH3&HH 2040 scenario, the additional counties in NAA estimated to have ozone emission ratios greater than 1% are Grundy and Kendall counties in Illinois; and Madison, Clinton, Fairfield, and Knox counties, Ohio; Crittenden County, Arkansas; and Hamilton County, Texas.

Similarly, figure 9.8 shows how the locations of counties in NAAs for $PM_{2.5}$ with emission ratios greater than 1% for $PM_{2.5}$ vary by year for the BC1&ML scenarios. In 2017, these areas are in Kings, Tulare, and Merced counties, California; Lincoln County, Montana; and Shoshone County, Idaho. However, in the BC1&ML 2040 scenario, non-attainment counties with $PM_{2.5}$ emission ratios higher than 1% are Monroe, St. Claire, and Randolph counties in Illinois, as well as Franklin County in Missouri. In the HH3&HH 2040 scenario, no additional counties in NAAs are estimated to have $PM_{2.5}$ emission ratios greater than 1%.

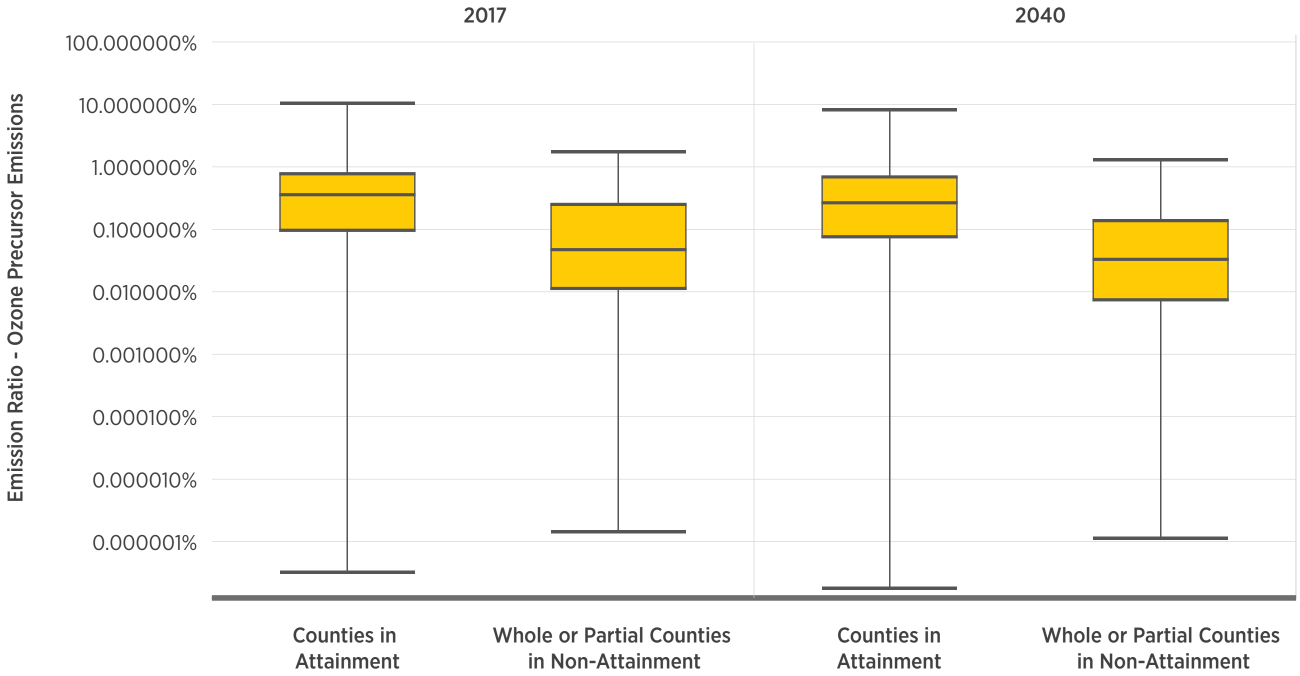
Shifts in both ozone precursors as well as primary and secondary $PM_{2.5}$ emissions from 2017 to 2040 scenarios are due to a combination of several factors. In 2040, decreased wholetree biomass production (e.g., Knox and Lincoln counties), higher agricultural residue yields (e.g., Kings, Monroe, and St. Claire counties), and decreases in the average distance of biomass on-road transportation using the long-term logistics system (most counties) would reduce the ozone and $PM_{2.5}$ emission ratios in non-attainment counties with ratios greater than 1% in 2017 to less than 1% in 2040. Increased potential energy crop production in combination with continuing agricultural residue production, in the nonattainment counties in 2040 would lead to ozone and $PM_{2.5}$ precursor emission ratios greater than 1% in a different set of counties relative to 2017. Additional differences in emission ratios between the BC1&ML 2040 and the HH3&HH 2040 scenario are due to additional transportation to biorefineries of potential additional biomass produced.

Figure 9.9 shows how the locations of counties in NAAs for PM_{10} with emission ratios greater than 1% for PM_{10} vary by year for the BC1&ML and HH3&HH scenarios. In 2017, these areas are in Lincoln County, Montana, and Shoshone County, Idaho. However, in the BC1&ML 2040 scenario, Flathead County, Montana, is the only nonattainment county with a PM_{10} with an emission ratio higher than 1% and Flathead County, Montana, and Power County, Idaho are nonattainment counties with a PM_{10} emission ratios higher than 1% in the HH3&HH scenarios. Changes in the emission ratio across these scenarios reflect decreased forestry biomass use for energy (e.g., in Montana and Idaho) and increased fugitive dust from agricultural residues, in particular straw, in Power County, Idaho.

The emissions that would be generated in counties with emission ratios greater than 1% for ozone and PM_{10} can generally be attributed to a few primary sources. Emissions from counties with high quantities of agricultural-residue production and emission ratios greater than 1% for ozone would be largely attributable to NO_X and VOC emissions from chemical application and on-road biomass transportation. Greater than 1% PM_{10} emission ratios would be attributable mostly to fertilizer and pesticide applications, contributing to PM precursor emissions (NH_3, NO_X, and VOC), fugitive dust emissions from the use of agricultural equipment that contribute to PM_{10}, as well as NO_X and SO_X emissions from biomass on-road transportation. In addition, emissions from wholetree biomass used for biomass are largely attributable to on-road biomass transportation.

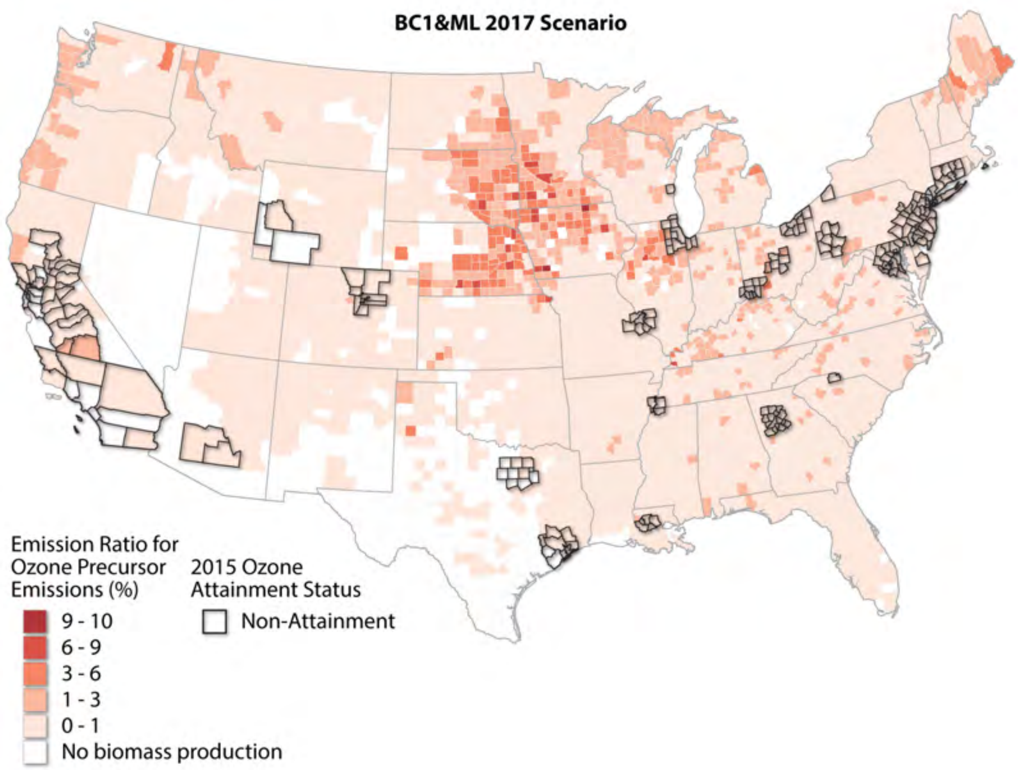
The results for emission ratios in NAAs for SO_2 differ from those for ozone and PM. For SO_2, only partial counties are in NAAs, so local air quality and transportation modeling would be needed to understand biomass transportation through NAAs in the county. For the BC1&ML 2017 scenario, only Muscatine County, Iowa, has an emission ratio that is greater than 1% for SO_2. For the BC1&ML 2040 scenario, in Muscatine County, Iowa, and Pike County, Indiana, emission ratios are greater than 1%. In the 2040 HH3&HH scenario, Muscatine County, Iowa; Pike County, Indiana; and Tazewell, Illinois, the NEI ratio is greater than 1%. The emission ratios in Muscantine, Pike, and Tazewell counties are largely attributable to transporting stover (i.e., up to 20 miles) and miscanthus to surrounding counties (i.e., up to 80 miles), respectively.

Figure 9.7 | BC1&ML 2017 and 2040 scenarios' county-level distributions of emission ratios for ozone in BC1&ML 2017 and 2040 scenarios (top frame).[11] Maps of emission ratios and non-attainment counties at the end of 2015 exceeding NAAQS standards for ozone (primary, 8-hour) (EPA 2016d)[12] are displayed in red in the 2017 (middle frame) and 2040 (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



[11] See text for a complete list of nonattainment counties with emission ratios above 1%.

[12] Includes NAA designations for the 2008 NAAQS that are still in force.


BC1&ML 2017 Scenario
Emission Ratio for Ozone Precursor Emissions (%)
9 - 10
6 - 9
3 - 6
1 - 3
0 - 1
No biomass production
2015 Ozone Attainment Status
Non-Attainment

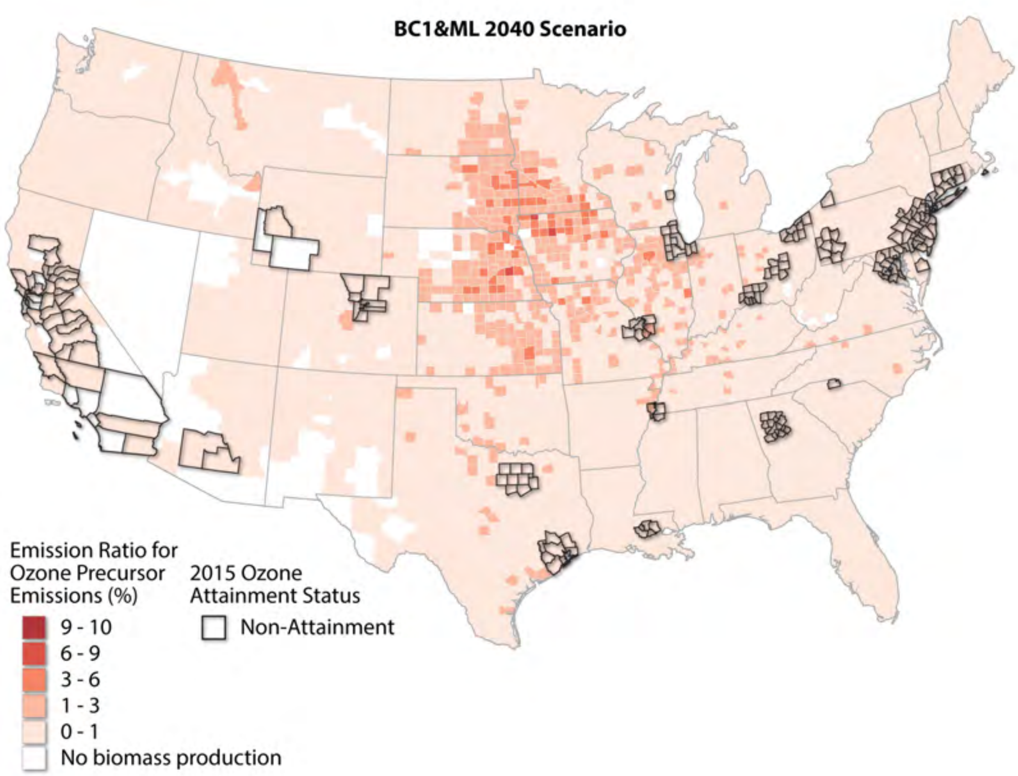
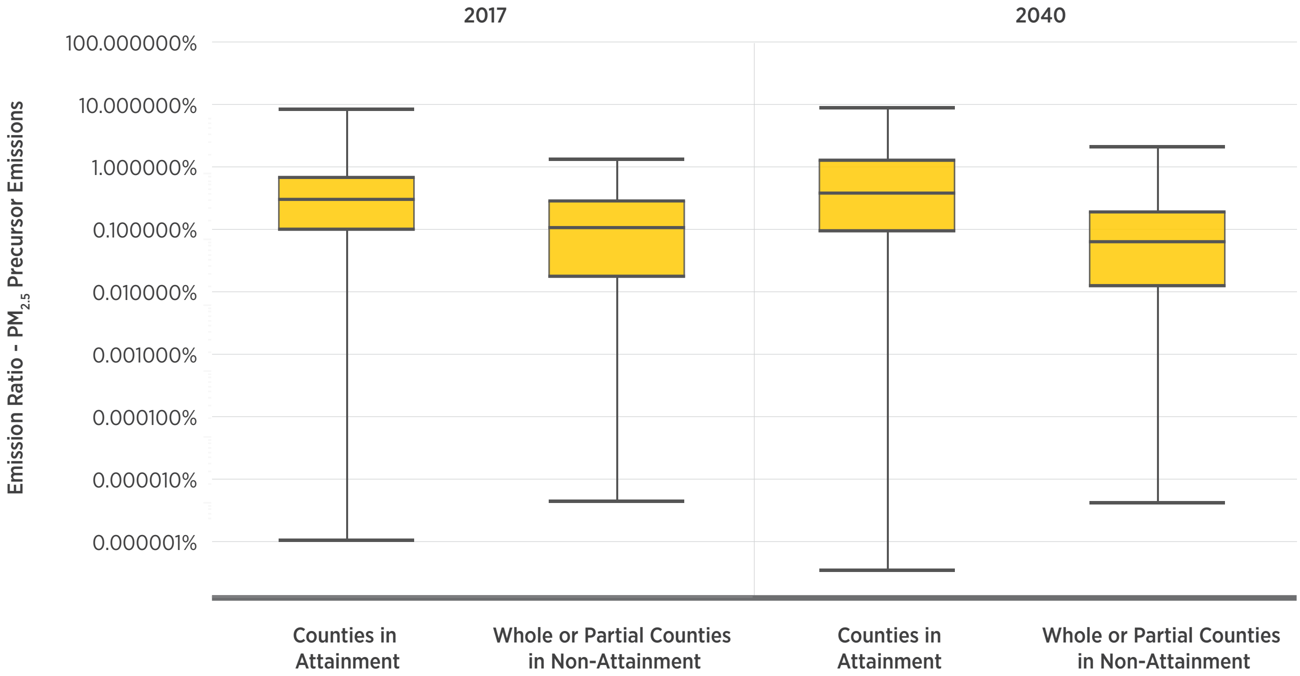
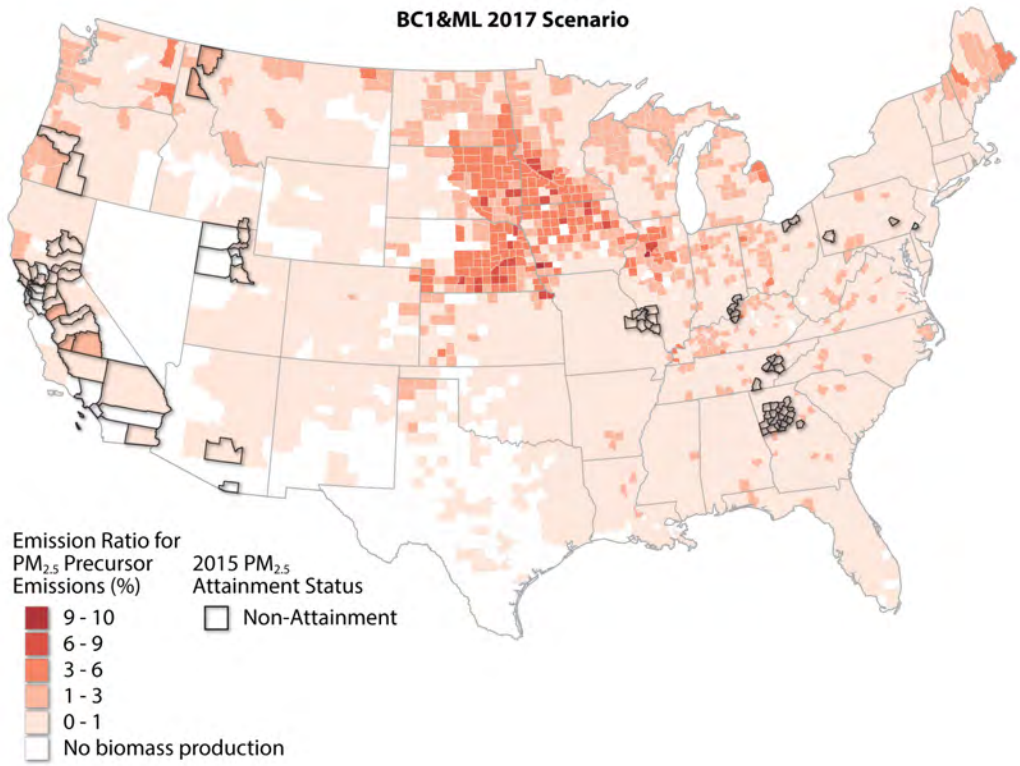

BC1&ML 2040 Scenario
Emission Ratio for Ozone Precursor Emissions (%)
9 - 10
6 - 9
3 - 6
1 - 3
0 - 1
No biomass production
2015 Ozone Attainment Status
Non-Attainment

Figure 9.8 | BC1&ML 2017 and 2040 scenarios' county-level distributions of emission ratios for $PM_{2.5}$ (top frame).[13] Maps of emission ratios and non-attainment counties at the end of 2015 exceeding NAAQS standards for $PM_{2.5}$ (primary, 24-hour and 1-year) (EPA 2016d)[14] are displayed in red in the 2017 (middle frame) and 2040 (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



[13] See text for a complete list of nonattainment counties with emission ratios above 1%.

[14] Includes NAA designations for the 1997, 2006, and 2012 NAAQS that are still in force.


BC1&ML 2017 Scenario
Emission Ratio for PM2.5 Precursor Emissions (%)
9 - 10
6 - 9
3 - 6
1 - 3
0 - 1
No biomass production
2015 PM2.5 Attainment Status
Non-Attainment

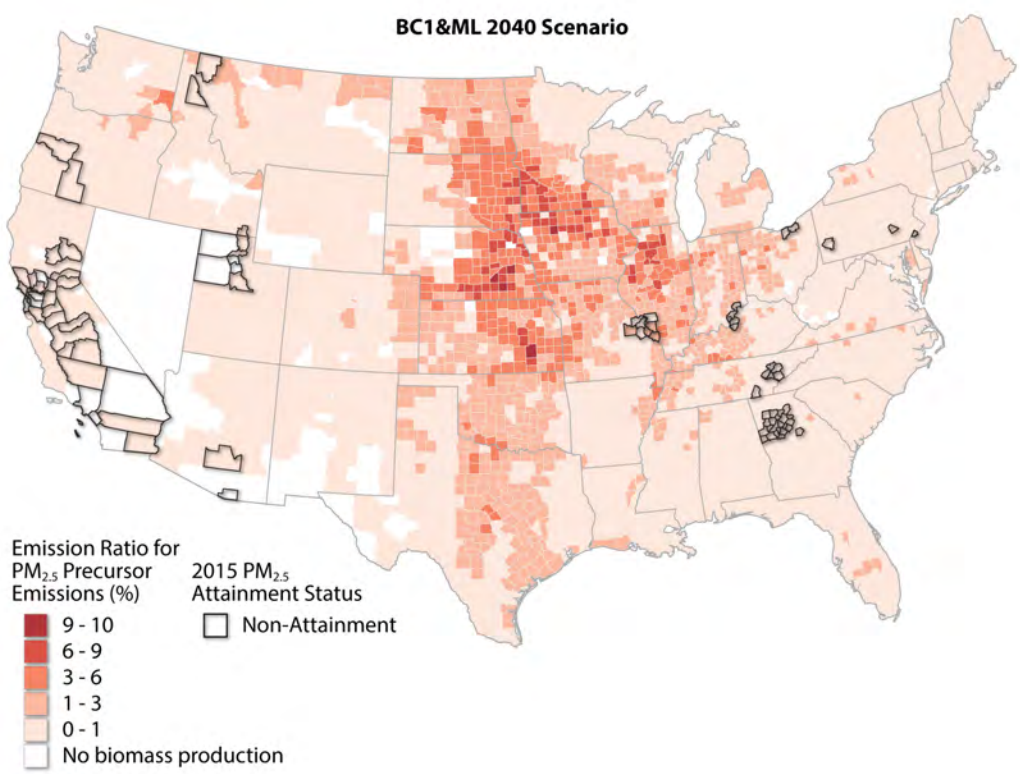
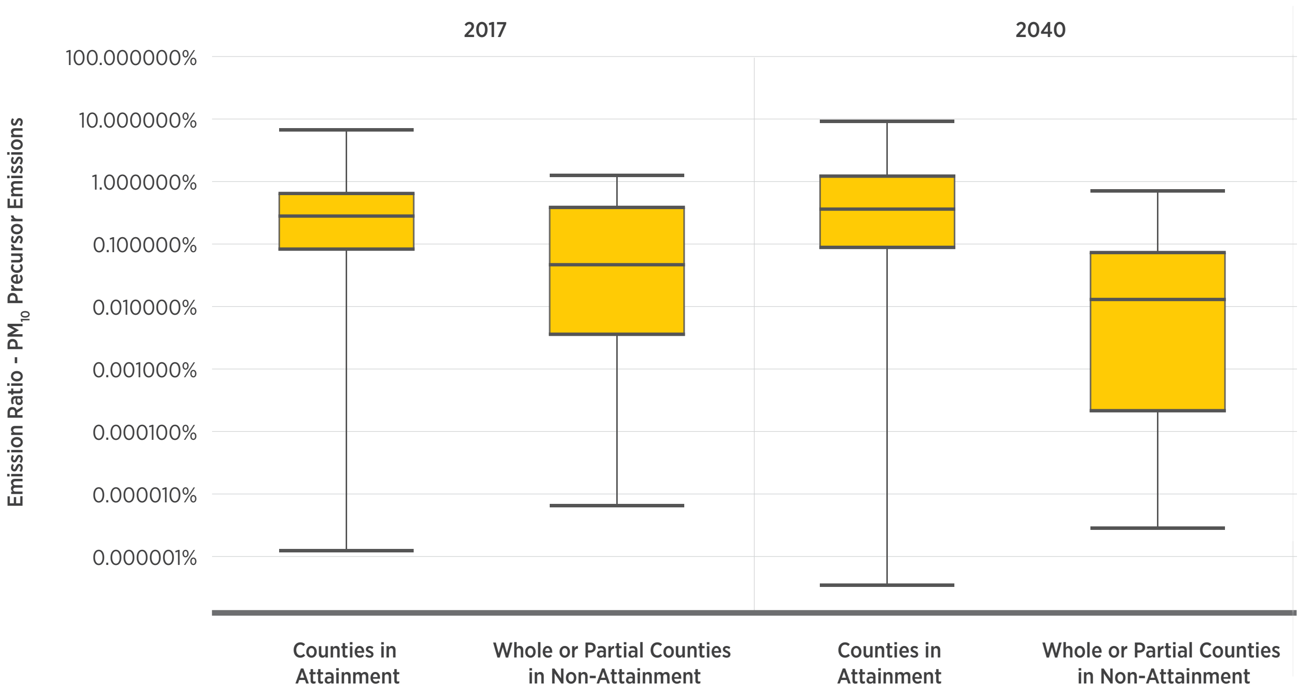
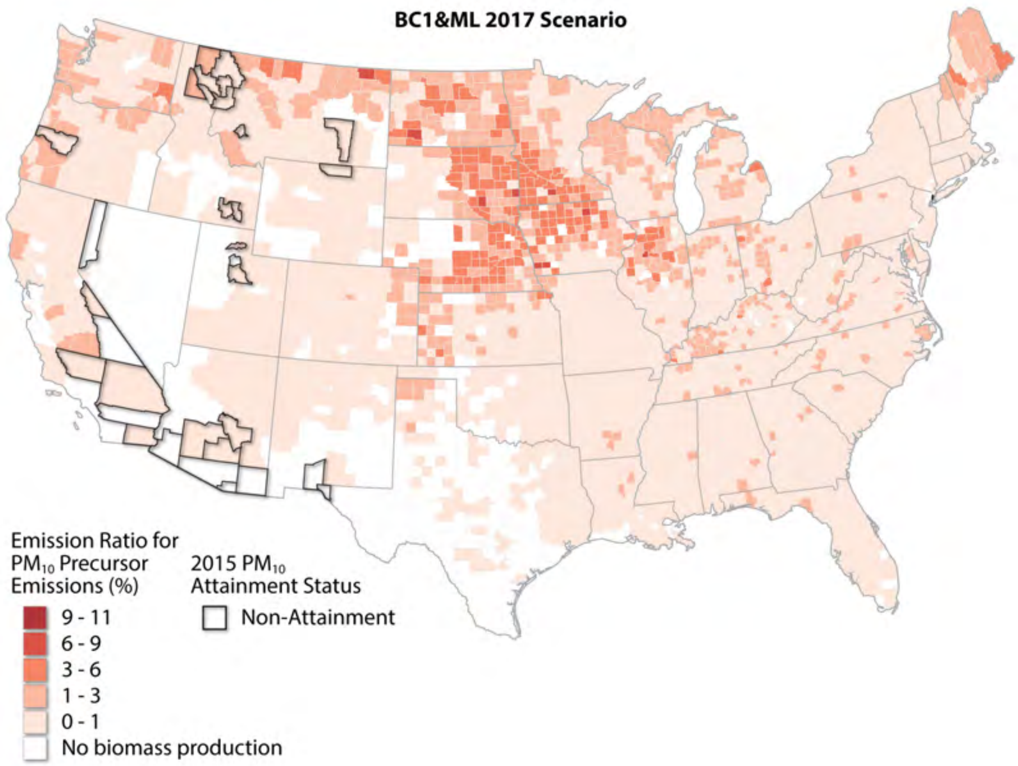

BC1&ML 2040 Scenario
Emission Ratio for PM2.5 Precursor Emissions (%)
9 - 10
6 - 9
3 - 6
1 - 3
0 - 1
No biomass production
2015 PM2.5 Attainment Status
Non-Attainment

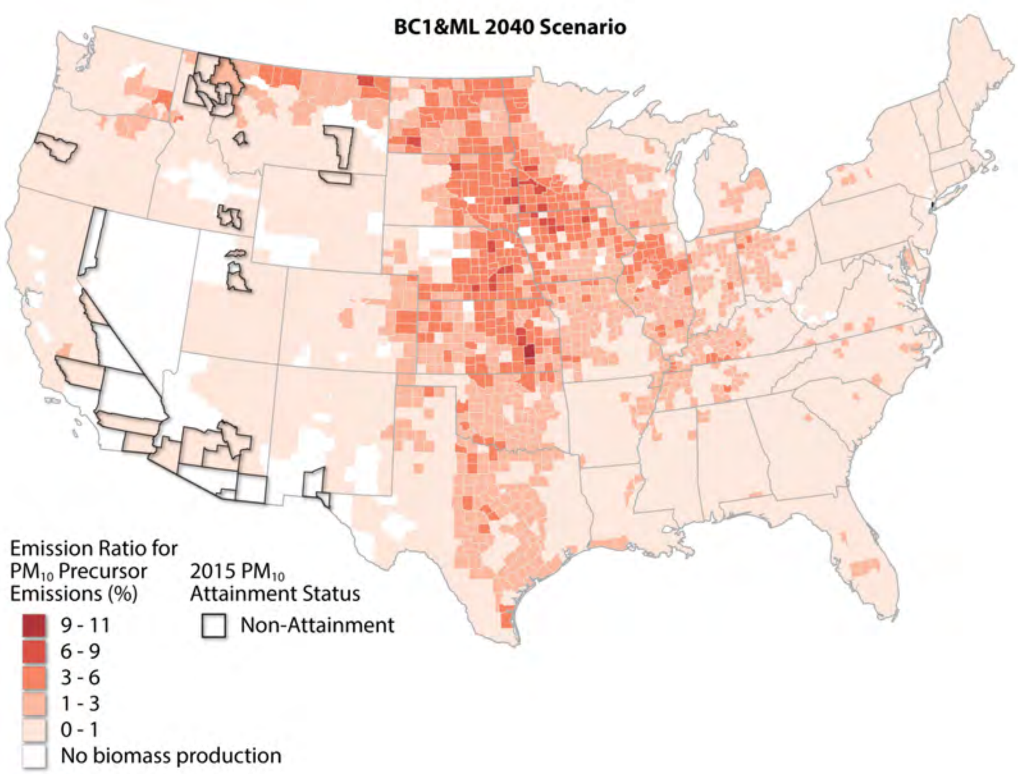
Figure 9.9 | BC1&ML 2017 and 2040 scenarios' county-level distributions of emission ratios for PM_{10} (top frame).[15] Maps of emission ratios and non-attainment counties at the end of 2015 exceeding NAAQS standards for PM_{10} (primary, 24-hour) (EPA 2016d)[16] are displayed in red in the 2017 (middle frame) and 2040 (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



[15] See text for a complete list of nonattainment counties with emission ratios above 1%.

[16] Includes NAA designations for the 1987 and 2012 NAAQS that are still in force.


BC1&ML 2017 Scenario
Emission Ratio for PM$_{10}$ Precursor Emissions (%)
9 - 11
6 - 9
3 - 6
1 - 3
0 - 1
No biomass production
2015 PM$_{10}$ Attainment Status
Non-Attainment


BC1&ML 2040 Scenario
Emission Ratio for PM$_{10}$ Precursor Emissions (%)
9 - 11
6 - 9
3 - 6
1 - 3
0 - 1
No biomass production
2015 PM$_{10}$ Attainment Status
Non-Attainment

9.3.4 Additional Discussion

In addition to comparing estimated emissions from the *BT16* scenarios to emissions in the NEI, we qualitatively discuss significant upstream emissions associated with potential biomass feedstock production and briefly discuss emissions from biomass and petroleum fuel production.

9.3.4.1 Life-Cycle Assessments

Zhang et al. (2016) compared the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) Model's (ANL 2015) national average life-cycle air pollutant emissions to FPEAM's spatially explicit inventory of emissions for biomass feedstocks. Direct comparisons are limited by the systems boundary of GREET, which also quantifies upstream emissions, and FPEAM, which estimates only direct local emissions. In addition, GREET does not include fugitive dust emissions, NH_3 emissions, or VOC emissions from pesticides or biomass preprocessing and drying. Fleet age and turnover, as well as other assumptions about equipment in the MOVES/NONROAD models that are used by FPEAM, also differ from GREET assumptions.

To identify potentially significant sources of upstream emissions, which could motivate future analysis, we qualitatively compared the results of our analysis of direct, anthropogenic emissions from feedstock production and logistics to life-cycle criteria air pollutant emissions from GREET. We reviewed GREET for activities that emit air pollutant emissions per unit of biomass of a similar or large magnitude as direct emissions sources analyzed in this chapter.

Based on the GREET model (ANL 2015), three potentially large sources of upstream emissions should be modeled in a spatially explicit fashion on a county-level basis (ANL 2015). GREET estimated that fertilizer manufacturing (primarily N-based) and transportation-related emissions are about as high as our estimated emissions from direct use of the fertilizer (ANL 2015). Other potentially large sources of upstream emissions are agricultural and forestry equipment manufacturing and maintenance (ANL 2015). This topic requires further research as GREET's equipment modeling only includes the capability to model one equipment type for biomass feedstocks, and the amortization of those emissions over the life of the equipment are not well aligned with our analysis (i.e., MOVES 2014a assumptions).

9.3.4.2 Crude Oil

Because biofuel is considered an alternative to petroleum-based gasoline and diesel fuels, another potential point of comparison is crude oil. The GREET model estimates that life-cycle air pollutant emissions from crude oil production and transport are lower than from biomass feedstocks (ANL 2015). A detailed comparison between GREET and FPEAM is not made because of the differences in system boundaries noted above. An inventory assessment of crude oil is a potential alternative that we can compare our assessment to. Existing assessments, such as one by Tessum, Hill, and Marshall (2014), also indicate that crude oil production generally emits fewer air pollutant emissions than biomass feedstocks on the basis of miles traveled using the fuel. However, studies (e.g., Tessum, Marshall, and Hill 2012) have shown that biofuels could have lower life-cycle criteria air pollutant emissions than their counterpart petroleum-based fuels. This is primarily attributable to the benefits of coproducts considered in life-cycle assessments (LCAs), such as electricity produced from the biofuel conversion process, displacing products derived from fossil fuels.

A detailed and specific analysis of crude oil is beyond the scope of this chapter. A more constrained inventory assessment of crude oil produced, supplied, and used in the United States would be feasible, but it would be misleading to compare that to biomass without taking a life-cycle approach because the emission sources in the biofuel life-cycle differ significantly from those in the petroleum fuel life-cycle. In addition, in order to understand "net" impacts of biofuel compared to petroleum, an integrated analysis would be needed to estimate when, where, and how much fuel might be displaced. Although a limited emissions inventory of only crude oil and the associated integrated analysis would not allow for a complete comparison between biomass and crude oil, it would help identify counties where air quality could improve because of reduced production of domestic crude oil and transportation of that oil. One benefit of biomass production for biofuels is that emissions from feedstock production and transportation, as well as emissions from biorefineries, are likely located in rural counties. U.S. petroleum refineries are largely located in or near urban areas, and therefore the exposure of populations and resulting health effects could change, depending on the fuel's supply chain. Consequently, there could be a complex and new pattern of air quality considerations when considering net emissions from a high biofuels penetration scenario.

9.3.4.3 Preprocessing Emissions Resulting from Electricity Usage

To test the significance of excluding upstream emissions from the *BT16* air pollutant emissions inventory, we performed a sensitivity analysis to estimate emission rates resulting from the electricity use of biomass preprocessing equipment. Other than the on-site wood chipper for processing whole-tree biomass, preprocessing equipment exclusively uses electricity. It is important to note that emissions from the generation of the electricity used in the preprocessing would, in general, not be emitted near the point of use. In addition, because the source of generated electrons is not known, we cannot specifically pinpoint the location of the upstream electricity generation emissions.

Table 9.5 summarizes air pollutant emission rates from electricity used with preprocessing equipment, assuming a U.S. grid mix; regional grid mixes are further discussed in the appendix section 9A.2.3. Emission estimates are based on the sum of preprocessing equipment electricity use from equipment budgets used in *BT16* volume 1 and the methodology documented in appendix section 9A.2.3. The long-term supply logistics system uses more than four times as much electricity as the near-term system. Thus, upstream emissions from electricity generated to supply power to preprocessing equipment used in 2040 would be far higher than those associated with 2017 feedstock preprocessing. However, while electricity consumption is much higher for long-term logistics systems than near-term ones, the net effects across the life cycle could ameliorate or neutralize any effect on increased air emissions because air emissions at the biorefinery should decrease owing to reduced preprocessing requirements (at the biorefinery) and more efficient conversion of feedstocks to fuels, etc.

Comparing table 9.5 to figures 9.2 and 9.3 indicates that the use of electricity in preprocessing equipment leads to large, nonlocal SO_X emissions because of a higher share of electricity generated from oil and coal compared to natural gas or renewable energy sources (see EFs in table 9A.6). If coal or oil plants are located in or near NAAs, it is possible that these areas will face increased challenges to comply with SO_2 NAAQS. However, with decreased use of coal and oil for electricity generation by 2040 (EIA 2016), upstream CO and SO_X emissions associated with electricity generation would be expected to decrease. If preprocessing is occurring on the site of cellulosic biorefineries, lignin could be burned to produce electricity. The use of lignin for electricity would have tradeoffs in local emissions (e.g., NO_X, PM, GHGs, SO_2) at the biorefinery, as compared to non-local emissions from electricity generation at power plants. Upstream NO_X emissions from the use of electricity in preprocessing equipment in long-term systems would be relatively low, but the emissions are higher than emissions from whole-tree biomass production due to the lack of chemical application for most whole-tree biomass. Finally, upstream VOC, PM_{10}, and $PM_{2.5}$ emissions from electricity generation would be lower than those from biomass production in the lowest emitting county.[17]

[17] This excludes forestry residues and biomass due to the lack of quantification of fugitive dust emissions.

Table 9.5 | Criteria Air Pollutant Emissions That Would Result from Preprocessing Equipment Electricity Use, Assuming a U.S. Grid Mix (DOE 2016; ANL 2013).

Feed-stock	Biomass Supply Logistics System	Total Electricity Use (kWh/dt)	NO_x (kg/dt)	VOC (kg/dt)	PM_{10} (kg/dt)	$PM_{2.5}$ (kg/dt)	CO (kg/dt)	SO_x (kg/dt)
Woody	Near-Term	40	0.021	0.00028	0.0049	0.0035	0.0036	0.053
Woody	Long-Term	190	0.099	0.0013	0.024	0.017	0.017	0.25
Herbaceous	Near-Term	36	0.019	0.00025	0.0044	0.0031	0.0032	0.048
Herbaceous	Long-Term	190	0.097	0.0013	0.024	0.016	0.017	0.25

Acronyms: kWh – kilowatt-hours; dt – dry ton; kg – kilogram.

9.3.4.4 Preprocessing Emissions of Fugitive Dust

In the estimation of the emissions inventory from the biomass production scenarios, we assume 100% dust collection efficiency for the preprocessing equipment based on both near-term and long-term supply-logistics design cases described in Idaho National Laboratory (INL) reports (2013 and 2014). However, in practice, no industrial dust collection system can achieve 100% efficiency long-term. According to EPA (1999), baghouse air pollution control technologies may not be completely effective at dust collection (i.e., 99.9% collection efficiency) due to equipment age or effectiveness of installation (e.g., system closure) or inefficiencies in the control technology (e.g., the filters).

As a result, we performed a sensitivity analysis to estimate PM emissions, assuming a 99% efficiency for the dust collection system, and compared the resulting preprocessing fugitive dust emissions to other PM emissions sources directly emitted from the biomass production and logistics processes shown in figure 9.5. We selected 99% efficiency to represent national average conditions of dust collection systems based on AP-42 (EPA 1999).

The estimated PM emissions are summarized in table 9.6 based on preprocessing throughput assumptions documented in appendix section 9A.2.5. A comparison of table 9.6 to figures 9.2 and 9.3 indicates that PM_{10} and $PM_{2.5}$ from preprocessing would be lower than emissions from agricultural biomass production from even the lower quartile of the 25th percentile. Fugitive dust emissions from whole-tree biomass would be low so these potential processing emissions would represent a large relative increase, but PM emissions from whole-tree biomass would still be lower than agricultural biomass. Fugitive dust emissions from preprocessing could become important sources of emissions if, in practice, dust collection efficiency were lower than 99%.

Table 9.6 | PM_{10} and $PM_{2.5}$ Emissions from Preprocessing Equipment, Assuming a 99% Efficiency of the Dust Collection System (DOE 2016; Krause and Smith 2006; WLA Consulting 2011; Davis et al. 2013)

Feedstock	Biomass Supply Logistics System	Preprocessing Throughput (dt/hr)	PM_{10} (lb/dt)	$PM_{2.5}$ (lb/dt)
Woody	Near-Term	8.5[a]	0.21	0.035
Woody	Long-Term	8.5[a]	0.21	0.035
Herbaceous	Near-Term	5	0.35	0.059
Herbaceous	Long-Term	6.5	0.27	0.046

[a] Two processing steps with a maximum throughput of 17 dt per 2 hours from SCM budget data (DOE 2016).

Acronyms: dt – dry ton; lb – pounds.

9.4 Discussion

The objectives of this analysis are (1) to estimate the air pollutant emissions for selected biomass production, harvest, transportation, and preprocessing scenarios; (2) to determine spatially where these emissions would occur and how these emissions could potentially impact air quality; and (3) to identify potential opportunities to minimize potential adverse impacts.

9.4.1 Implication of Results

Future air pollutant emissions resulting from large-scale deployment of production and supply logistics as depicted in the *BT16* scenarios, if realized and additional (rather than displacing other agriculture or forestry activities), could yield increases in emissions that could pose challenges for areas to attain the NAAQS. The implications of air emission estimates presented in this chapter are discussed in this section in regard to feedstock comparison, potential areas where emissions might increase, sources of emissions, and opportunities for emission mitigation.

9.4.1.1 Feedstock Comparison

For biomass production on a per-unit-of-biomass basis, agricultural residues are likely to lead to lower air pollutant emissions than agricultural crops because of tillage and field establishment activities (other than fertilizer application) not being allocated to the residues. However, because harvest and collection of agricultural residues are additional activities beyond those required for growing and harvesting grains, the emissions from these "extra" activities are more likely than energy crops to represent additional emissions. The production of residues requires that the primary crop be grown.

While switchgrass and miscanthus have higher emissions than agricultural residues, the production of these two energy crops is estimated to generate lower emissions than corn grain (the most commonly used feedstock for biofuel at present) on a per-unit of biomass basis due to their greater yield. Relative to agricultural residues, growing energy crops may replace corn grain and other conventional crops, and therefore the "net" change in air emissions will be much smaller than the emissions resulting from growing and harvesting the energy crops. In fact, if switchgrass and miscanthus replace annual crops, it is possible that this displacement would lead to reductions in air emissions.

For biomass production, logging residues and whole-tree biomass are estimated to produce NH_3, NO_X, and VOC emissions that are similar to or lower than agricultural biomass feedstocks (on a per-unit-of-biomass basis) because of overall lower chemical application. Total PM emissions from logging residues and trees are not comparable to those for other feedstocks because the fugitive dust emissions from whole-tree biomass are not quantified. However, equipment use leads to CO and SO_X emissions from whole-tree biomass that are similar per dt to agricultural feedstocks (with logging residues leading to fewer emissions owing to the allocation assumptions). Since *BT16* volume 1 assumes that the land base for forestry does not change in *BT16* scenarios, the equipment for whole-tree biomass is the same as that for conventional forestry, so changes from conventional uses to energy uses are less likely to change emissions from the production of whole-tree biomass.

On a per-unit-of-biomass basis, on-road transportation is a major source of NO_X, CO, and SO_X emissions, so the differences between various cellulosic feedstocks for a pollutant would shrink because emissions vary by transportation distance rather than feedstock type. The most noticeable remaining difference between emissions from different cellulosic feedstocks is that NO_X, CO, and SO_X off-site transportation emissions from logging residues are higher than emissions from other biomass feedstocks. Low logging residue production costs allow for longer distances (i.e., increased transportation costs) and still fall within the $100 per dt cutoff for the supply logistics scenario. Relative to biomass production only, NH_3, $PM_{2.5}$, and PM_{10} emitted per unit of biomass by each feedstock remains similar across feedstocks when accounting for on-road transportation and preprocessing. Relative to biomass production, whole-tree biomass NH_3, $PM_{2.5}$, and PM_{10} emissions would increase because of the limited chemical application and the lack of fugitive dust emission estimates for whole-tree biomass production in this analysis.

9.4.1.2 Potential Areas Where Emissions Might Occur

This analysis identifies counties in attainment with NAAQS for ozone, $PM_{2.5}$, and PM_{10} that are estimated to have emission ratios greater than 1%. About 25% of the counties evaluated for ozone, $PM_{2.5}$, and PM_{10} would have emission ratios above 1% with emission ratios reaching about 10% and 11% in BC1&ML 2017 and 2040 scenarios, respectively. An emissions ratio above 1% is suggested as a threshold that any county might consider a potentially significant increase in emissions that would warrant further attention by air quality managers in anticipating potential air quality degradation or degradation upwind of that county. Counties in nonattainment whose emission ratios are above the suggested threshold of 1% are considered among the most at risk for substantial air quality degradation.

Another important consideration for contextualizing the results of this analysis is that long transport distances could result in precursor pollutants being emitted upwind of counties without significant cellulosic feedstock production. These transportation emissions could impact the emissions in upwind counties. For instance, the emissions from many Midwest and Corn Belt counties, despite largely being in compliance with the NAAQSs, could contribute to concentrations of $PM_{2.5}$ or ozone in other states downwind. The emissions inventory developed here can be further utilized for air quality modeling by creating temporal profiles and chemical speciation for each emission source. This would help determine the air quality and human health impacts of potential biomass feedstock production. Alternatively, the emissions inventory can be coupled with an air quality screening tool such as EPA's Co-Benefits Risk Assessment (COBRA) screening model to evaluate important changes in emission concentrations and potential changes in human health (EPA 2015b).[18]

[18] Other potential screening tools include InMAP (Intervention Model for Air Pollution) (Tessum et al. 2016), APEEP/AP2 (Air Pollution Emission Experiments and Policy analysis model) (Muller and Mendelsohn 2011), and EASIUR (Estimating Air pollution Social Impact Using Regression) (Heo and Adams 2016).

The estimated air pollutant emissions inventory indicates that the potential changes in ozone and PM2.5 precursor emissions (i.e., 2011 NEI ratio) from *BT16* cellulosic biomass production and supply in nonattainment counties are greater than 1% of the NEI in a few counties. Specifically in the BC1&ML 2017 scenario, there are nine counties in nonattainment for ozone, $PM_{2.5}$, PM_{10}, or SO_2 NAAQSs located in California, Ohio, Illinois, Iowa, Montana, and Idaho. Also in the BC1&ML 2040 scenario, there are eight counties in nonattainment for ozone, $PM_{2.5}$, or SO_2 NAAQSs located in Arkansas, Iowa, Illinois, Missouri, Indiana, and Montana. In the HH3&HH 2040 scenario, there are 17 counties in nonattainment for ozone, $PM_{2.5}$, or SO_2 NAAQSs located in Arkansas, Illinois, Missouri, Ohio, Iowa, Indiana, Texas, Montana, and Idaho. Emission ratios in these counties that are greater than 1% of NEI emissions indicate that there could be increased challenges for these counties to meet NAAQS under the scenarios. When comparing the 2017 scenario to the 2040 scenario, counties that may experience a greater increase in air emissions would shift geographically because of the change in county-wide biomass production (such as the type and quantity of biomass feedstocks produced).

9.4.1.3 Sources of Emissions and Opportunities for Emission Mitigation

The results of the inventory indicate several potential improvements that could mitigate risks to NAAs. The emissions estimated here for cellulosic biomass feedstocks could be further mitigated through the application of several emission reduction strategies. A comparison of the BC1&ML 2040 and HH3&HH 2040 scenarios indicates that much more biomass could be produced with only a marginal increase of about 1%–2% in emission ratios nationwide.

The use of more efficient equipment or equipment that requires fewer passes in NAAs could reduce the risk of changing local air quality by decreasing emissions per acre planted or harvested, as well as per unit of biomass produced or supplied. This is important as potential equipment improvement from 2017 to 2040 is not captured in *BT16*. For example, this analysis illustrates that the long-term feedstock supply logistics system itself could also reduce emissions per mile traveled through feedstock densification. In addition, using biomass more locally or using more fuel-efficient long-distance transportation methods (e.g., rail) could potentially decrease emissions from long-distance truck transport.

The use of less-intensive tillage practices for conventional agricultural crops such as corn grain would, in part, offset additional emissions from the harvest, collection, and transport of agricultural residues in NAAs. Furthermore, while the use of waste biomass (e.g., yard wastes and construction and demolition wastes) was not examined in this chapter, its use could also lower estimated emissions based on its lack of chemical application, tillage, and harvest activities.

Finally, constraining biomass grown or the types of biomass grown or collected (e.g., crop residues) in counties in NAAs or geographically near counties at risk for being in nonattainment is another potential mitigation strategy for potential future biomass production and supply. For example, agricultural residues are more likely to lead to emissions that are in addition to emissions from corn grain cultivation than energy crops that might replace conventional crops. Another option is that for some pollutants, such as PM, an option to prevent emissions moving upwind could be establishing buffer vegetation near agricultural lands. For example, research has shown that vegetation in forested areas can potentially remove 80%–100% of particulate emissions (Pace 2005).

9.4.2 Limitations of This Study

BT16 volume 1 estimates potential biomass production in the future. There is inherent uncertainty associated with evaluating feedstocks not currently produced at a commercial scale. As a result, the estimates of potential improvements to current crops and the comparative analysis across feedstocks could also change in the future. While our modeling of the practices and inputs for cellulosic feedstock production uses the best available information, the lack of long-term, commercial-scale production of cellulosic feedstocks (especially dedicated energy crops like switchgrass) leads to uncertainty in our results.

In addition, the focus of this analysis is on air pollutant emissions potentially resulting from the increased biomass production and supply under particular *BT16* scenarios. We do not model changes in emissions relative to a reference scenario or the impacts of these emissions on local or regional air quality. In the context of these limitations, our analysis should be revisited as experience with and data for biomass feedstocks improve and as the development of emission-estimating methods matures. Hence, results presented here should not be interpreted as predictions of changes in air pollutant concentrations within certain counties as the consequence of the potential biomass production and supply scenarios analyzed. Instead, the results of this study are intended to illustrate potential impacts of increased biomass production and to motivate further study when deemed appropriate relative to potential changes in other source categories and to NAAQS attainment strategies.

The following sections cover an important but not exhaustive set of limitations associated with this analysis.

9.4.2.1 Limitations of the Scenarios and the Inventory Approach

The estimated air pollutant emissions inventory only includes emissions based on three scenarios of potential biomass production and supply modeled at specific farmgate, roadside, and delivered prices for 2017 and 2040. These scenarios are neither optimized for yield increases nor for mitigating air pollutant emissions. Hence, opportunities exist to minimize air emissions from biomass production and supply.

A lack of data on the shares of cellulosic biomass feedstocks used for different markets (e.g., power, biofuel, export) at subnational levels (county- or state-level) limits county-level comparisons of transportation emissions from biomass production. Future assessments of feedstock allocation by end use at a subnational level are critical for more accurate estimates of local and regional air pollutant emissions from cellulosic feedstock production.

Changes in the allocation approach could significantly affect the estimated emissions for multi-product feedstock production systems. While we consider product-purpose allocation to be the most appropriate approach for analyzing residues due to a current lack of a commercial market for these residues (Wang, Huo, and Arora 2011), alternate allocation methods might become more relevant if residues become a commodity in the future and play a role in farmers' crop selection. In addition, our study employs an attributional LCA approach, which tracks the physical flows directly associated with the system being investigated, and hence, does not include emissions (or avoided emissions) outside of the system boundaries. Examples of emissions outside the system boundaries include changes in biogenic emissions associated with land management changes or the use of biomass, such as logging residues, which would otherwise be burned.

Direct modeling of future changes in air quality resulting from potential large-scale biomass production requires the estimation of criteria air pollutant emissions in a business-as-usual scenario relative to the high-potential biomass production scenario. Although the inventory approach for evaluating cellulosic biomass feedstocks allows us to gain some insights about the large-scale deployment of these feedstocks, there is uncertainty associated with actual changes in emissions, due to the lack of a business-as-usual scenario for emissions sources in agriculture or other local industries contributing to local emissions.

The EPA AP-42 emissions calculation methodology that we used to evaluate fugitive dust from road transportation (EPA 2006) also has some uncertainties associated with calculations. The fugitive dust equations are empirically developed, and the range of source conditions on which the equations are based align well with evaluating scenario transport on both paved and unpaved roads. However, the equations require data on silt loading, and state average values are utilized due to lack of other available data. This analysis also does not include the precipitation correction factor, as it has not been rigorously verified by EPA. Please refer to the appendix section 9A.2.5 for more details on the underlying assumptions for these fugitive dust equations.

9.4.2.2 Geographic Resolution

The geographic resolution of the air pollutant emissions inventory is limited in several key respects. Biomass production and supply data from POLYSYS, ForSEAM, and SCM runs are reported on a county basis. However, data sources and methods for estimating EFs and emissions are often not highly spatially resolved. Many EFs are estimated based on measured data for the United States or U.S. regions. The ability to estimate emissions nationwide using county-level MOVES runs is constrained by computing limitations (i.e., length of a MOVES run for a single county) and data at the county level.

NO_X emissions from the soil are primarily produced as part of the nitrogen cycle. Many factors impact this cycle, and the estimate of the net NO_X emission attributable to nitrogen fertilizer application is highly variable and depends on soil temperature, moisture content, pH, N availability, organic matter content, type of nitrogen fertilizer, application method, and type of vegetation. For this analysis, we assume that anhydrous ammonia, ammonium nitrate, urea, ammonium sulfate, and N solutions are applied to corn grain, stover, and straw, and the shares of these five nitrogen fertilizers are estimated based on USDA's survey data (USDA 2010; USDA 2011). We further assume that EFs do not vary by locations and soil conditions. This assumption has limitations because EFs could vary significantly. A recent study (Oikawa et al. 2015) finds that NO_X EFs in high temperature agricultural systems are higher than the values commonly used (typically between 1% and 2%). Oikawa et al. estimate that the NO_X EFs range from 1.8% to 6.6% in the Imperial Valley in California, regardless of fertilizer type and application method. We acknowledge that our estimates of N-fertilizer-induced NO_X emissions are highly uncertain. Although it is beyond the scope of this work, improvement can be made when better data (e.g., field-specific data) and tools are available for developing spatially explicit NO_X emission estimates.

NH_3 is released into the atmosphere following the application of nitrogen fertilizers. Similar to NO_X emissions induced by nitrogen fertilizers, the volatilization of NH_3 depends on the type of fertilizer used, soil properties, and meteorological conditions. Based on EPA's method, we used the mean value of the EF specific to a given type of nitrogen fertilizer in our calculation. However, it is worth noting that these average EFs do not reflect the variations in local conditions. Including uncertainty in the analysis could provide additional insights into how soil properties and meteorological conditions would affect the magnitude of NH_3 emissions due to nitrogen fertilizer application.

Agricultural fugitive dust emissions were determined based on work by the California Air Resources Board (CARB) (2003) and Gaffney and Yu (2003). Emissions factors were developed for different activities and crops. These estimated emissions factors come from a study done by the University of California, Davis in the San Joaquin Valley that measured PM10 emissions for harvest operations occurring from 1994–1998. The study performed a total of 149 tests across different operations, crops, soils, equipment, and time of year. Measurements of similar types of operations were averaged to produce composite emissions factors. This methodology does not account for variability outside of California, and assumptions used to translate emissions to crops not covered in this study contribute to the uncertainty of the results. Data collected are also relevant to equipment that is almost 20 years old, and the data do not include updates in equipment. In addition, emissions factors were not determined for all crops, so proxies were recommended. It is assumed that harvest activities are unique to a crop, so all operations associated with the harvest of each crop were combined into a single emissions factor.

Biomass preprocessing and drying are another important source of VOC emissions from woody feedstocks. The EFs available from EPA (2002) are not designed for application to a specific set of county-level conditions, but rather for application to specific pieces of equipment for some limited mixtures of wood. A county-level analysis of VOC emissions would require knowing emissions factors for particular wood mixes, as well as for equipment used in each locality. These methodologies are also based on conventional uses of wood, and therefore, the specifications for drying equipment do not match the INL design reports (2013 and 2014) and are likely overestimated given the higher temperatures used for these equipment types in AP-42 (EPA 2002).

Air pollutant emissions from the transportation of biomass were all allocated to the originating county of potential biomass production. This assumption was made because short transportation distances located within a single originating county were common and pathing data for longer transportation distances was unknown. The results of these assumptions are that emissions from transportation in some counties are likely highly inflated, and emissions from surrounding counties are lower than they would be if pathing data were available. The spatial resolution of transportation fugitive dust is also limited because road conditions at a county level are not well specified, so we used data at either the state or national level. See section 9A.2.5 for further discussion of uncertainties around transport fugitive dust.

9.4.2.3 Major Gaps in Emission Sources for Further Research

Several important data and methods gaps in our criteria air pollutant emissions inventory for biomass feedstocks require further research and development. The fugitive dust generated during forestry activities was not included due to a lack of data. The transport to and from the logging sites is covered in the chapter, but fugitive dust emissions from logging and other feedstock management activities is not included in this analysis. We contacted the Consortium for Research on Renewable Industrial Materials (CORRIM), which has done extensive research on impacts from the logging industry, and to CARB, which has done extensive research on fugitive dust emissions. At this time, neither organization had any documentation or information about research on fugitive dust from logging industry activities. However, this gap may not have a significant impact on our results because research has shown that vegetation in forested areas can potentially remove 80%–100% of particulate emissions (Pace 2005).

Emissions from biomass burning were omitted from this analysis due to a combination of a lack of spatially resolved, business-as-usual burning conditions assumed in the *BT16* analysis and barriers to developing county-specific emission estimates. Open combustion of whole-tree biomass produces large amounts of smoke that is composed of variable amounts of carbon, tars, liquids, and numerous gases. The exact composition of the smoke released from whole-tree biomass combustion is related to the temperature of the fire (rate of heat release), as well as the composition of the biomass (i.e., conifer vs. hardwood), and how the biomass was treated and/or gathered. This is particularly evident for PM emissions. PM size distributions from prescribed forest burning have been described in Radke et al. (1990) and Ward and Hardy (1984). The main emissions from open burning of whole-tree biomass are PM, CO, and VOCs. The ranges for emissions vary from 6 to 16 g/kg for $PM_{2.5}$, 28 to 226 g/kg for CO, and 1 to 9 g/kg for methane VOCs (Ward and Hardy 1984; Sandberg and Ottmar 1983).

This study did not include the biogenic emissions attributed to the agricultural and whole-tree biomass feedstocks assessed. For example, VOC emissions related to the growing and/or cutting of biomass were excluded from this analysis. Biogenic emissions are those pollutants that are emitted from natural sources such as trees and other plants, including crops. Biogenic emissions vary depending on a number of physiological plant attributes such as leaf size and density, growth characteristics, and aerial distribution. Accounting for biogenic emissions requires high-resolution spatiotemporal data. CARB has begun to support empirical research aimed at developing such a database for the State of California (CARB 2013).

Only anthropogenic emissions are tracked in our analysis. Studies (Shapouri et al. 2010; Eller et al. 2011) have shown that cultivation of agricultural feedstock crops used for biofuel (e.g., switchgrass, short rotation coppice) generates biogenic VOC emissions and could result in changes in surface ozone and secondary organic aerosol concentrations, which in turn would have an impact on local air quality. Biogenic emissions from feedstock production and harvest could be considered in future research. Such analysis should consider the net change to biogenic emissions if biofuel feedstocks are grown on lands previously used for another purpose, as well as any emissions associated with the change from one land type to another. Accounting for biogenic air emissions from biomass crops would require such detailed data for both the biomass crop and the crop or vegetation being replaced. Future efforts may consider accounting for such emissions sources pending data availability.

[19] NH_3, NO_x, VOCs, $PM_{2.5}$, PM_{10}, CO, and SO_x.

9.5 Summary and Future Work

County-level air emission inventories were developed for seven non-GHG, regulated air pollutants[19] under scenarios in which agricultural biomass production and whole-tree biomass production are expanded. Emissions were estimated for the BC1&ML 2017, BC1&ML 2040, and HH3&HH 2040 scenarios. These inventories consider emissions from field preparation through harvest, including chemical application and on-farm (or on-forest) transportation, along with transportation for a selected portion of feedstock to the biorefinery. The results of this analysis indicate that although the estimated air pollutant emissions per unit of biomass vary by county and pollutant, they are generally lower for cellulosic feedstocks than for corn grain. However, this study also shows that the emissions that would result from increased biomass feedstock production could pose challenges for local compliance with air quality regulations. Upstream air emissions (e.g., emissions associated with fertilizer production) and air emissions avoided by displacing other products or fuels with biomass-derived products or fuels were beyond the scope of this study. However, emissions reductions from displacement or upstream emission may be substantial and should be the focus of future study.

Based on the scenarios and assumptions employed, producing cellulosic feedstocks would emit lower quantities of six evaluated pollutants (all except particulate matter) per dt of feedstock in the majority of U.S. counties, as compared to producing corn grain. As summarized in table 9.7, for agricultural feedstock production, chemical application is a major source of emissions. The majority of NH_3 and NO_x emissions would be attributable to nitrogen fertilizer application, and VOC emission would be attributable to pesticide application, respectively. For logging residues and whole-tree biomass, the major sources of NH_3 and NO_x are generally harvest and non-harvest fuels use, and the major source of VOC emissions is preprocessing. The two contributing sources of PM emissions for all feedstocks are fuel combustion and fugitive dust emissions. SO_x and CO emissions are emitted primarily by equipment used to harvest all feedstocks. When off-site transportation and pre-processing activities are included, they become the major source of many emissions—all except for NH_3, PM_{10}, and $PM_{2.5}$—for all feedstocks.

The variability in county-level emissions estimates suggests that certain practices and production locations would result in much lower emissions than others. Higher yields, lower tillage requirements, and lower fertilizer and chemical inputs are important factors that contribute to lower air emissions. A comparison of the BC1&ML 2040 and HH3&HH 2040 scenarios indicates that much more biomass could be produced with only a marginal increase of about 1%–2% in the emission ratio comparing the inventory to the 2011 baseline emissions from the NEI. The use of either more efficient equipment or equipment that requires fewer passes would reduce emissions from fuel use and fugitive dust from soil disturbance. The application of emission reduction strategies (e.g., higher yielding seed varieties, energy crops with high nutrient use efficiency, more efficient farm engines, and wider adoption of less intensive tillage practices) could mitigate the potential increase in emissions from *BT16* scenario activities. This analysis illustrates that the long-term feedstock supply logistics system itself could reduce emissions per mile traveled through feedstock densification. In addition, using biomass more locally or using more fuel-efficient long-distance transportation methods (e.g., rail) could potentially decrease emissions from long-distance truck transport.

Future air pollutant emissions, if realized and additional (rather than displacing other agriculture or forestry activities), represent increases in emissions that could challenge certain areas in attaining the Clean Air Act's NAAQS. For the *BT16* scenarios analyzed, about 25% of the counties currently in attainment for ozone, $PM_{2.5}$, and PM_{10} NAAQS are estimated to emit direct and precursor criteria pollutants above 1% of 2011 NEI. These emissions could pose challenges for some areas to meet the NAAQS in the context of population and economic growth. Emissions in areas currently in attainment could also pose challenges for surrounding areas. For example, long-distance transport of ozone, $PM_{2.5}$, and PM_{10} direct and precursor emissions means that downwind counties without significant cellulosic feedstock production could be affected by biomass production from upwind counties. For instance, emissions from Midwest and Corn Belt counties that are in compliance with the NAAQS could contribute to increased concentrations of PM or ozone in downwind counties that struggle to comply with the NAAQS.

Table 9.7 summarizes nonattainment counties for relevant NAAQS where the total potential mass emissions from biomass were above 1% of the NEI for a county. While the absolute increase in mass emissions under *BT16* scenarios is estimated to be small in these areas (a few percent of the current NEI baseline emissions, see discussion above) relative to current attainment counties, these emissions are more likely to pose challenges to meeting the Clean Air Act's NAAQS in the context of population and economic growth.

The emission estimates provided in this study could help inform future air-quality planning, such as state implementation plans, which are required to consider new emission sources for future scenarios. They could also be coupled with air-quality screening tools to evaluate potential changes in emissions concentrations, to assess potential human health impacts, and to develop constraints (i.e., excluded lands) for future scenarios related to biomass production. Beyond air quality assessments this research can help identify locations where constraints on or emission mitigation strategies for biomass production and supply could be explored in future modeling.

Table 9.7 | NAAs Where Total Mass Emissions Relevant to Certain NAAQS Could Increase Relative to a 2011 Baseline from the NEI as a Result of *BT16* Potential Biomass Production and Supply Logistics Scenarios.

NAAQS Primary Standards	BC1&ML 2017 NAA w/ Emission Ratios >1%	BC1&ML 2040 NAA w/ Emission Ratios >1%	HH3&HH 2040 NAA w/ Emission Ratios >1%	NAAs Upwind of Counties with Emission Ratios >1%[a] Across Scenarios	Major Emission Source(s)
2008 Ozone (8-hour)	Kings, CA Tulane, CA Madison, OH Knox, OH Kane, IL	St. Claire, IL Monroe, IL Crittenden, AR	Grundy, IL Kendall, IL Monroe, IL St. Claire, IL Madison, OH Clinton, OH Fairfield, OH Knox, OH Crittenden, AR Hamilton, TX	Chicago, IL (11 counties) St. Louis, MO (8 counties) Memphis, AR (3 counties) Cincinnati, OH (9 counties) Columbus, OH (6 counties)	Chemicals Transportation
1997, 2006, and 2012 $PM_{2.5}$ (24-hour and 1-year)	Kings, CA Tulane, CA Lincoln, MT Merced, CA Shoshone, ID	St. Claire, IL Monroe, IL Randolph, IL Franklin, MO	St. Claire, IL Monroe, IL Randolph, IL Franklin, MO	St. Louis, MO (8 counties) Louisville, KY (4 counties) Cleveland, OH (2 counties) Lincoln, MT Lane County, OR Klameth County, OR Shoshone County, ID	Chemicals Fugitive Dust Transportation
1987 and 2012 PM_{10} (24-hr and 1-year)	Lincoln, MT Shoshone, ID	Flathead, MT	Flathead, MT Power, ID	Northwest Montana (5 counties) Lane County, OR Shoshone County, ID	Chemicals Fugitive Dust Transportation
1971 and 2010 SO_2 [b] (1-hour)	Muscantine, IA	Muscantine, IA Pikes, IN	Muscantine, IA Pike, IN Tazewell, IL	N/A	Transportation

[a] Additional air quality assessment tools would be needed to determine if these counties might be impacted by adjacent emissions.

[b] For the SO_2 NAAQS, only partial counties are in nonattainment, so local air quality and transportation modeling would be needed to understand if transportation would be through NAAs.

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Appendix 9-A

Figure 9.1 and table 9.4 in section 9.2.2 of the chapter 9 summarize the main components of the Feedstock Production Emissions to Air Model (FPEAM), which we use to estimate the air pollutant emissions reported in this chapter. FPEAM was first described in Zhang et al. (2016), and appendix sections 9A.1 and 9A.2 explain the basic assumptions, equations, and input data that we used to generate the results reported in this chapter. These appendix sections also describe several updates and improvements that we made to the model, including the development of methods to evaluate additional feedstocks and new methods for estimating transportation and preprocessing emissions.

9A.1 Key Equipment Activity Assumptions

FPEAM uses the same equipment and chemical application budgets used in BT16 volume 1 (see section 9.2.1). Where applicable, FPEAM retains the following dimensions to BT16 volume 1 budget data for use in our modeling, and as such, these data elements are not discussed in the following sections:

- Year (i.e., 2017 and 2040)
- Scenario (i.e., BC1&ML and HH3&HH[1])
- County (i.e., Federal Information Processing Standard code)
- Feedstock (e.g., switchgrass or miscanthus)
- Tillage type (e.g., conventional)
- Near-term and long-term biomass supply logistics.

9A.1.1 Biomass Production – Agricultural Sector

FPEAM uses the following data elements from the agricultural biomass production budgets (DOE 2016):

- Equipment type (e.g., tractor)
- Fuel type (e.g., diesel)
- Equipment horsepower (hp)
- Rates of equipment usage (hr/ac)
- Rates of chemical application (lb/ac for crops and lb/dt for residues).

For agricultural residues, the product-purpose allocation approach (Wang, Huo, and Arora 2011) is used to estimate emissions associated with residue harvesting because residues are a byproduct of crop production. In other words, only additional inputs exclusively attributable to residue removal are allocated to residues. As a result, additional fertilizer application (assumed to be applied using the same equipment pass required for fertilizing the grain) is the only non-harvest activity associated with agricultural residues. No additional equipment is modeled for non-harvest activity associated with agricultural residues.

For corn grain, FPEAM incorporates additional equipment for irrigation, which are not included in the *BT16* volume 1 equipment data. The irrigation equipment data is based on the U.S. Department of Agriculture's (USDA's) Farm and Ranch Irrigation Survey (USDA 2009) which is administered every five years and covers all

[1] BC1&ML scenario is the agricultural base case yield growth (BC1) and the moderate housing–low wood energy (ML) forestry scenarios combined. HH3&HH scenario is the high-yield growth (HH3) and the high housing–high wood energy (HH) scenarios combined.

farms that produce $1,000 or more of agricultural products. Farms and ranches in the United States use gasoline, diesel, liquefied petroleum gas (LPG), compressed natural gas (CNG), and electricity to operate their irrigation systems. This applies to both well and surface water sources, as well as pressure and gravity irrigation systems. Although the dominant energy sources for irrigation systems are electricity (60%) and diesel (27%) (USDA 2009), the energy mix can vary by state. Since emissions from electricity may not be local, and the location of their release is difficult to determine, we only estimate the irrigation-related emissions associated with fuel use and exclude electricity.

The following tables from the 2008 survey were used to estimate state-level irrigation pumping requirements for corn grain (USDA 2009):

- Table 15 – Irrigation Wells Used on Farms: 2008 and 2003
- Table 16 – Characteristics for Irrigation Wells Used on Farms: 2008 and 2003
- Table 18 – Irrigation Pumps on Farms for Wells: 2008 and 2003
- Table 19 – Irrigation Pumps on Farms Other Than for Wells: 2008 and 2003
- Table 20 – Energy Expenses for On-Farm Pumping of Irrigation Water by Water Source and Type of Energy: 2008 and 2003
- Table 28 – Estimated Quantity of Water Applied and Primary Method of Distribution by Selected Crops Harvested: 2008 and 2003.

For each state, the following data were extracted:

- Crop (corn only)
- State
- Irrigation method (well, non-well, discharge, reservoir, and boost)
- Irrigated acres
- Amount of water used for irrigation per acre (uH_2O, acre-ft/acre)
- Fuel type (gasoline, diesel, LPG, natural gas, or electricity[2])
- Percentage of acres by fuel type and irrigation method
- Average flow (q, in gallons per minute [gpm])
- Static water depth (d, ft)
- Load factor of engine (lf, %)
- Pump efficiency (pe, %)
- Gear drive efficiency (gde, %)
- System pressure (p, in pounds per square inch [psi])
- Friction head (FH, ft)
- Velocity head (VH, ft)
- Pressure head (PH, ft).

[2] Upstream emissions from electricity use are not included, but the data on the percent of equipment using electricity is used.

County-level data were derived from the state data using county-level acreage of corn grain. Equipment activity (hr/ac) and power (hp) are calculated from USDA (2009) using equations 9A.1 and 9A.2, respectively (CARB 2006).

Equation 9A.1:

$$\frac{hrs}{ac} = \frac{u_{H20}}{q} * \frac{325851}{60}$$

Equation 9A.2:

$$hp = \frac{q * (d + PH + FH + VH)}{3960} * \frac{1}{gde * pe} * \frac{1}{lf}$$

Where the following are defined as true:

- 325,851 gal/ac-ft converts from acre-feet of water to gallons of water.
- 60 converts from hours to minutes.
- 3,960 converts minute-gallons of water to feet, where (2.31 ft/psi)*(7.48 gal/ft^3)*(60 sec/min)*(550 (lb*ft/sec)/hp)*(psi/144 (lb/ft^2)).
- *FH* is the friction head, which is assumed to be 2.54 ft.
- *VH* is the velocity head, which is assumed to be negligible.
- *PH* is the pressure head, which equals the pressure times 2.31 ft/psi.

9A.1.2 Biomass Production – Forestry Sector

FPEAM uses the following data elements from the forestry biomass production budgets (DOE 2016):

- Equipment type (e.g., skidder or chainsaw)
- Fuel type (e.g., diesel)
- Equipment horsepower (hp)
- Rates of equipment usage (hr/dt)
- Rates of chemical application (lb/ac for whole biomass).

Since logging residue collection occurs in conjunction with pre-existing logging operations (DOE 2016), a product-purpose allocation is applied to logging residues. Thus, only activities at the landing are considered in our inventory; all activities involved in getting the logging residues to the landing is attributed to the harvested logs. Since the logging residues are already assumed to be transported to the forest landing during log harvesting, the equipment list for logging residues consists only of a chipper and a loader.

9A.1.3 Biomass Supply Logistics

Biomass supply logistics include transportation and preprocessing of agricultural biomass at the farm gate and chipped wood collected at the roadside. FPEAM uses the following data elements from the biomass supply logistics (DOE 2016):

- Equipment type (e.g., truck)
- Fuel type (e.g., diesel)
- Equipment horsepower (hp)
- Rates of equipment usage (dt/hr)
- Electricity use (kWh/dt)
- Vehicle capacities (dt/trip)
- Vehicle fuel economy (mile/gal).

No deviations from the biomass supply logistic budget were made.

9A.2 Key Emission Modeling Assumptions

FPEAM uses the U.S. Environmental Protection Agency's (EPA's) Motor Vehicle Emission Simulator (MOVES) version 2014a to estimate criteria and other air pollutant emissions generated by most mobile sources of fuel use (EPA 2016a).

Consistent with *BT16* volume 1, switchgrass is produced in a 10-year rotation cycle, and miscanthus is produced in a 15-year rotation cycle, with different equipment being used for establishment and maintenance years. Because switchgrass and miscanthus are perennials, our emissions analysis assumes that 1/10 and 1/15 of any county's switchgrass or miscanthus production occurs in a single year of the 10-year rotation and 15-year rotation, respectively. For simplicity, for a given year (e.g., 2040), the emissions of a given pollutant, P, are summed up over acres in each year of the production cycle using equation 9A.3.

Equation 9A.3:

$$E_{P,\,feed,\,c} = \sum_{i=1}^{R} \frac{A \times E_i}{R}$$

Where the following are defined as

- $E_{P,feed,c}$ are the emissions of pollutant P in county c (lb/yr) for feedstock, *feed*
- A is total harvested acres in a county in a given year (DOE 2016)
- E_i is the sum of all emissions from one acre of switchgrass or miscanthus production in a given year (i) of the 10-year or 15-year cycle (E_i varies by year due to different activities and chemical requirements)
- R is rotation years.

9A.2.1 Non-Road Fuel Use Emission Estimates

Harvest and non-harvest activities require the operation of machinery for activities such as discing, tilling, and baling in the agriculture sector, and felling, delimbing, and bucking in the forestry sector. The operation of these types of equipment generates air pollutant emissions from fuel use. For agricultural and forestry non-road (or off-road) equipment, the fuel use emissions estimated by MOVES 2014a are mostly computed using EPA's NONROAD 2008a model (EPA 2016b, hereafter referred to as NONROAD). Airplanes used in one agriculture region for corn grain production (DOE 2016) are not included in NONROAD and instead use an alternative data source (San Joaquin Valley Air Pollution Control District 2012).

NONROAD was selected because the model generates emission inventories for individual counties, covers all the major air pollutants of interest (carbon monoxide [CO], oxides of nitrogen [NO_X], oxides of sulfur [SO_X], particulate matter [PM_{10}], and total hydrocarbons [THCs]) except for ammonia [NH_3] (which is calculated separately based on fuel consumption and emission factors [EFs]), and takes into account emission controls required by regulations over time (from 1970 to 2050) (EPA 2005b). In particular, the NONROAD model is designed to account for the effect of the federal emissions regulations. However, it does not cover any California emissions standards or any proposed federal emissions standards.

In addition to estimating emissions from combustion exhaust, NONROAD also estimates evaporative emissions. Evaporative emissions refers to hydrocarbons released into the atmosphere when gasoline, or other volatile fuels, evaporate from equipment (EPA 2010g). The types of evaporative emissions covered in the NONROAD model include diurnal, tank permeation, hose permeation, hot soak, and running losses (EPA 2010g).

The NONROAD model uses equation 9A.4 (EPA 2010b) to calculate combustion exhaust emissions and evaporative emissions associated with each of the six pollutants listed above (i.e., CO, SO_X, NO_X, CO_2, PM_{10}, and THC). These emissions, $E_{P,NONROAD,feed,c}$ (in lb/yr), are calculated for each feedstock, feed, each pollutant, *P*, and each county, *c*.

Equation 9A.4:

$$E_{P,\ NONROAD,\ feed,\ c} = POP_{feed,\ c} * Power * LF * A_c * EF_P$$

Where the following are defined as

- $POP_{feed,c}$ is equipment population, or the number pieces of equipment in each equipment category in county c in a given year for feedstock, *feed* (calculated using the activity rate from the equipment budgets and the production data (DOE 2016); see details below)
- Power is the average horsepower (hp) of the machinery (DOE 2016)
- *LF* is the load factor or fraction of available power (%) (EPA 2010b)
- A_c is the average annual activity of single piece of equipment in county c each year (hr/yr/piece of equipment) (EPA 2010b)
- *EF* is the emission factor (lb/(hp*hr)) (EPA 2010b).

In NONROAD, the equipment population in each county can be specified by the user and includes age distributions that vary with equipment type and scenario year (EPA 2010c). Since the type and number of machinery required for an activity varies by feedstock type, the equipment populations also vary by feedstock. We used crop budgets as described in appendix sections 9A.1.1 and 9A.1.2 and biomass production and harvested area estimates from BT16 volume 1 to compute the number and type of tractors and other equipment required by each feedstock in each county. For each feedstock, feed, the population of each type of non-road equipment in county, c, $POP_{feed,c}$ (number of pieces of equipment), is given by equation 9A.5.

Equation 9A.5:

$$POP_{feed,\,c} = HPA_{feed,\,c} * Harv_{feed,\,c} * \frac{1}{A}$$

Where the following are defined as

- $HPA_{feed,c}$ is the number of hours that the equipment is used per acre in county c for feedstock, feed (hr/ac) (DOE 2016)
- $Harv_{feed,c}$ is the number of feedstock acres harvested per year in county c for feedstock, feed (ac/yr) (DOE 2016)
- A is the average hourly activity of a single piece of this type of equipment used per year (hr/piece of equipment/yr) and varies with equipment type (EPA 2010b; see usage rate in table 9A.1).

The NONROAD program uses source classification codes to distinguish the different engine types and horsepower (hp) ranges. Table 9A.1 summarizes the non-road equipment categories in NONROAD that correspond to equipment used in the *BT16* volume 1 agriculture and forestry budgets. It is important to note that the program does not model specific pieces of equipment, but engines of varying power ranges (EPA 2005a). For example, a 135 hp tractor is modeled in a 100–175 hp range. More information on how the NONROAD model calculates these emissions may be found in the model's technical documentation (EPA 2010b).

Table 9A.1 | Average Number of Hours Non-Road Equipment Is Used per Year (Usage Rate, A) by Type (EPA 2010b).

Sector	Equipment Type	Source Classification Codes	Usage Rate, A (hr/piece of equipment/yr)
Agriculture	Diesel agricultural tractor	2270005015	475
	Diesel combine	2270005020	150
	Irrigation set (powered by gas, LPG, and CNG	22X0005060	716
	Diesel irrigation set (powered by diesel)	2270005060	749
Forestry	Diesel logging feller/bunch/skidder	2270007015	1,276
	Diesel crawler tractors/ dozers	2270002069	936
	Lawn and garden equipment chain saws <6 hp (commercial)	2270004020	303
	Diesel chipper (commercial)	2270004066	465

The NONROAD model also calculates age distributions for equipment populations by equipment type and scenario year. This calculation is necessary for the model to account for several factors that affect emissions over time, including emissions deterioration, new emissions standards, changes to technology, changes in equipment sales and total equipment population, and scrappage programs. More detailed information may be found in the NONROAD model's technical documentation (EPA 2005b; EPA 2005c; EPA 2004; EPA 2010f).

9A.2.1.1 Aerial Emissions (Corn Grain Only)

The agricultural equipment budgets from the *BT16* volume 1 assume aerial application of fertilizer in one agricultural region for corn grain. Airplanes are not included in NONROAD so FPEAM uses EF data from a California report on crop dusting emissions (San Joaquin Valley Air Pollution Control District 2012; see table 9A.2).

Table 9A.2 | Crop-Dusting Criteria Air Pollutant EFs (San Joaquin Valley Air Pollution Control District 2012).

NO_x (short tons/acre)	CO (short tons/acre)	SO_x (short tons/acre)	VOC (short tons/acre)	$PM_{2.5}$ (short tons/acre)	PM_{10} (short tons/acre)
$1.56*10^{-5}$	$6.75*10^{-6}$	$1.08*10^{-6}$	$4.17*10^{-7}$	$0.97*PM_{10}$	$1.05*10^{-7}$

The total amount of combustion exhaust emissions, $E_{aerial,p,cg,c}$ (in lb/yr), for pollutant (P) in each county (c) for corn grain, cg, are given by equation 9A.6.

Equation 9A.6:

$$E_{p,\ aerial,c,g,c} = EF_p * Harv_{c,g,c} * 2000$$

Where the following are defined as

- EF_p is the pollutant-specific emission factor from table 9A.2
- $Harv_{cg,c}$ is the number of corn grain acres harvested per year in county c (ac/yr)
- 2000 converts tons to pounds.

9A.2.1.2 NH_3, $PM_{2.5}$, and Volatile Organic Compound (VOC) Emissions

Since NONROAD does not compute the emissions of NH_3, particulate matter under 2.5 micrometers in diameter ($PM_{2.5}$), or VOCs, we computed these emissions separately using EPA conversion factors (see table 9A.3).

Table 9A.3 | Conversion Factors for Computing Emissions of NH_3, $PM_{2.5}$, and VOCs Using NONROAD Estimates, which Include PM_{10} and THCs. LHV Is the Lower Heating Value of the Fuel and$_{NH3}$ Is the EF for NH_3.

	$PM_{2.5}$	LHV (Btu/gallon)	EF NH_3 (g/mm BTU)	VOC
Diesel	0.97*PM_{10}[a]	128,490[c]	0.68[d]	1.053*THC[e]
Gasoline	0.92*PM_{10}[b]	116,090[c]	1.01[d]	0.933*THC[e]
LPG	1.0*PM_{10}[b]	84,450[c]	(not reported)	0.995*THC[e]
CNG	1.0*PM_{10}[b]	20,160[c]	(not reported)	0.004*THC[e]

[a] EPA 2010d

[b] EPA 2010e

[c] DOE 2012

[d] EPA 2015b

[e] EPA 2010a

Emissions of NH_3 ($E_{NH3,c,feed}$ in lb/yr for feedsock, *feed*, in county *c*) are estimated based on fuel consumption and are given by equation 9A.7.

Equation 9A.7:

$$E_{NH3,feed,c} = FC_{feed} * LHV * EF_{NH3} * 0.0022$$

Where the following are defined as

- *FC* is the amount of fuel consumed by the equipment used for feedstock, feed, per year (estimated from NONROAD)
- *LHV* is the lower heating value of the fuel (Btu/gal) given in table 9A.3
- EF_{NH3} is the EF for NH_3 (g/mmBtu) given in table 9A.3
- 0.0022 is the conversion from grams to pounds.

The size distribution of the particulate matter is given in NONROAD technical documentation (EPA 2010d; EPA 2010e). As shown in table 9A.3, $PM_{2.5}$ emissions are derived from PM_{10} and are distinguished by fuel type (EPA 2010c).

The NONROAD program adds THC to oxygenated compounds (alcohols and aldehydes commonly found in engine exhaust) then subtracts the methane and ethane components to get VOC (EPA 2010a). The definition of VOC excludes methane, ethane, acetone, and compounds not commonly found in large quantities in engine exhaust, like chlorohydrocarbons. Although acetone is not subtracted, it is present in smaller quantities compared to methane and ethane, and will have a negligible effect on the results (EPA 2010a; EPA 2010g). The THC to VOC conversion factors are shown in table 9A.3.

9A.2.2 On-Road Fuel Use Emission Estimates

In consultation with experts at Oak Ridge National Laboratory, we assume that all on-road (off-farm) transportation (i.e., transport of the feedstock from the farm to the depot and the depot to the biorefinery in the long-term logistics case, and transport from the farm to the biorefinery in the near-term case) will occur via a combination short-haul truck (DOE 2016). Although the exact route of travel is unknown, for our modeling purposes, we assume that it would occur within the biomass source county (see limitations discussion in chapter 9, section 9.4.2.2). FPEAM uses EPA's MOVES model, version 2014a, to estimate the emissions generated during transportation using a vehicle of this type (EPA 2016a).

In order to compute the total emissions generated in each county, FPEAM runs MOVES on the county level using the rates mode. This approach allows us to run MOVES once per state and year to compute the emission rates for the county producing the most cellulosic biomass in each state and year. The results are then post-processed on the county level by combining the appropriate state-level emission rates with the county-level transportation data. This approach allows us to compute the total emissions for each feedstock at the county level, while saving valuable computation time by running MOVES only once per state and year for all feedstocks.

For MOVES input data, we rely primarily on the default data in the MOVES database.

We use national defaults for

- Alternative vehicle and fuel technology
- Average speed distribution
- Day vehicle mile fraction
- Hour vehicle mile fraction
- Month vehicle mile fraction.

We use county-level defaults for

- Meteorology
- Fuel formulation
- Fuel supply
- Fuel usage fraction.

Default age profiles for each scenario year are created using the MOVES Age Distribution Tool (EPA 2016a). We also use data from the Federal Highway Administration (FHWA 2006) to compute the national average of the fraction of vehicle miles travelled by road type (table 9A.4).

Table 9A.4 | Crop-Dusting Criteria Air Pollutant EFs (San Joaquin Valley Air Pollution Control District 2012).

Road Type	Billion VMT by Combination Trucks (national)	Fraction of VMT by Road Type
Off network	No data	0
Rural restricted	43	0.30
Rural unrestricted	39	0.28
Urban restricted	30	0.21
Urban unrestricted	30	0.21

For computing the emission rates (i.e., running MOVES), we assume that the distance traveled by each vehicle was a default value of 100 miles. However, during post-processing the emission rates are multiplied by the actual distance traveled per trip to compute the total emissions per trip. We also assume that the source population (i.e., the type and number of vehicles) consists of a single vehicle for computing emission rates (i.e., running MOVES). However, similar to distance traveled during post-processing, the actual source population is equal to the number of trips required to transport the quantity of feedstock generated in a specific county.

With regard to the MOVES time frame, we only run MOVES for a single month, assuming that most emissions will occur during October, which is around the time when most crops would be harvested. We also assume that most activity would occur within the hours of 6 a.m. and 6 p.m., so we only run MOVES for this 12-hour period.

We use the emission rates generated by MOVES (EPA 2015a) to compute the total emissions ($E_{P,f,feed,c}$ in lb/yr) generated by the transportation of each feedstock, feed, in each county, c, according to equation 9A.8.

Equation 9A.8:

$$E_{P,f,feed,c} = \Sigma_h \Sigma_{proc} ([V_{P,proc,h,st} + P_{P,proc,h,st} + \Sigma_r \Sigma_s [D_{P,r,s,h,proc,st} * S_{r,s,h} * VMT_{feed,c,r}]]) (T_{feed,c}) * 2204$$

Where the following are defined as

- $V_{P,proc,h}$ is the rate per vehicle (in metric tons per vehicle) computed by MOVES for each pollutant, *P*, pollutant process, *proc*, and hour, *h*, for the state-level representative county, *st*, producing the most cellulosic biomass in the state where county, *c*, resides
- $P_{P,proc,h}$ is the rate per profile (in metric tons per hour)[3] computed by MOVES for each pollutant, *P*, pollutant process, *proc*, and hour, *h*, for the state-level representative county, *st*, producing the most cellulosic biomass in the state where county, *c*, resides
- $D_{P,r,s,h,proc}$ is the rate per distance (in metric tons per vehicle mile travelled) computed by MOVES for each pollutant, *P*, road type, *r*, speed bin, *s*, hour, *h*, and pollutant process, *proc*, for the state-level representative county, *st*, producing the most cellulosic biomass in the state where county, *c*, resides
- $S_{r,s,h}$ is the average fractional amount of time that a combination short-haul truck spends traveling on road type, *r*, in speed bin, *s*, during hour, *h*, of a weekday (MOVES default value)
- $VMT_{feed,c,r}$ is the number of vehicle miles a truck must travel per trip for feedstock, feed, in county, *c*, on road type, *r* (DOE 2016)
- $T_{feed,c}$ is the number of trips per year required to transport all of feedstock, feed, supplied in county, c (computed from data in DOE 2016)
- 2204 converts from metric tons to pounds.

9A.2.3 Preprocessing Fuel Use and Electricity Emission Calculations

For biomass supply logistics, the only piece of equipment that uses diesel (DOE 2016) is the wood chipper, which is only used for woody feedstocks. We use the NONROAD model (EPA 2010b) to calculate combustion exhaust emissions from wood chipping following the same general methods outlined in appendix section 9A.2.1. The exception to these methods are that the population of the wood chipper, $POP_{feed,c}$ (number of pieces of equipment used for feedstock, *feed*, in county c), is given by equation 9A.9 rather than by equation 9A.5.

Equation 9A.9:

$$POP_{feed,c} = \frac{1}{DTPH_{feed}} * Supply_{feed,c} * \frac{1}{A}$$

[3] Since we are using only diesel powered trucks, and MOVES assumes diesel fuel generates no resting evaporative emissions, the rate per profile is zero.

Other preprocessing equipment uses electricity instead of diesel. Electricity use creates non-local criteria air pollutant emissions upstream of where the electricity is used. As noted in chapter 9, section 9.2.2, we do not evaluate criteria air pollutant emissions from sources upstream of equipment or fertilizer production. However, preprocessing equipment's primary electricity use is potentially a source of non-local emissions from primary energy use. For discussion, we provide a rough estimate of emissions from electricity use in two counties: an agricultural biomass producing county and a whole-tree biomass producing county. We compare these emissions from electricity use to other emission sources in chapter 9, section 9.3.4.3.

For a single county, c, and for each feedstock category, *FC* (woody or herbaceous), the emission rate of pollutant, *P*, generated from all preprocessing activities ($ER_{elec,P,FC,c}$ in lb per dt) are calculated by summing emissions over all fuel combustion technologies, such that we have equation 9A.10.

Equation 9A.10:

$$E_{elec,P,FC,c} = \sum M_{tech} * EF_{P,tech,} * EPH_{FC,c} * 0.0022$$

Where the following are defined as

- M_{tech} is the percentage of electricity in the United States supplied by a given technology (see table 9A.5)
- $EF_{P,tech}$ are technology specific EFs (g/kWh) (see table 9A.6)
- $EPH_{FC,c}$ is the electricity used to process a dt of feedstock category, *FC*, in county, *c* (kWh/dt)
- 0.0022 is the conversion from grams to pounds.

Table 9A.5's electricity generation mix provides a general indication of the potential regional variability in these emission estimates.

Table 9A.5 | Electricity Generation Mix in 2016 of Combustion and Non-Combustion Technologies in the Eight Contiguous North American Electric Reliability Corporation Regions (ANL 2015)

Fuel	United States	Florida Reliability Coordinating Council	Midwest Reliability Organization	Northeast Power Coordinating Council	Reliability First Corporation	SERC Reliability Corporation	Southwest Power Pool	Texas Reliability Entity	Western Electricity Coordinating Council
Residual oil	0.6%	1.7%	0.2%	1.4%	0.2%	0.4%	1.3%	0.1%	0.2%
Natural gas	26.4%	60.2%	2.6%	50.1%	15.9%	18.8%	22.7%	41.3%	32.5%
Coal	40%	23.3%	61.8%	2.6%	51.3%	49.3%	56.0%	36.0%	25.4%
Biomass	0.3%	0.5%	0.6%	0.6%	0.1%	0.4%	0.0%	0.2%	0.2%
Non-combustion technologies	32.8%	14.2%	34.7%	45.3%	32.4%	31.0%	20.0%	22.3%	41.7%

Table 9A.6 | National Electricity Criteria Air Pollutant EFs in 2010 (ANL 2013)

Fuel	Technology	NO_x (g/kWh)	SO_x (g/kWh)	PM_{10} (g/kWh)	$PM_{2.5}$ (g/kWh)	CO (g/kWh)	VOC (g/kWh)
Oil	Steam turbine	4.4825	7.6442	0.1797	0.1395	0.1676	0.0216
Natural gas	Combined cycle	0.1175	0.0041	0.0009	0.0009	0.098	0.0018
Coal	Steam turbine	1.141	3.1998	0.2836	0.1994	0.1221	0.0147
Biomass	Steam turbine	0.9267	0.603	2.814	1.9763	4.7546	0.1349
Non-combustion technologies		0	0	0	0	0	0

9A.2.4 Emissions from Chemical Application

The application of fertilizers and pesticides results in the emission of several types of air pollutants, including NH_3, NO_X, and VOCs. The sections below describe the methods that FPEAM uses to calculate air pollutant emissions from each of these sources.

9A.2.5 Fertilizer Emissions

Our estimates of air pollutant emissions from fertilizer application are limited to the emissions associated with nitrogen fertilizer because no studies have yet reported the emissions of NO_X, VOC, SO_2, $PM_{2.5}$, PM_{10}, NH_3, or CO from the application of potassium and phosphorus fertilizers. However, fugitive dust emissions from applying these fertilizers are accounted for as described in appendix section 9A.2.5.

Since we do not have information about the exact type of nitrogen fertilizer that is applied to each feedstock, we consider a distribution of the five most commonly used nitrogen fertilizers: anhydrous ammonia, ammonium nitrate, urea, ammonium sulfate, and nitrogen solutions (USDA 2013). We assume these five nitrogennitrogen fertilizers are used for corn grain, stover, straw, switchgrass, and miscanthus.

Because each nitrogen fertilizer type emits different levels of NO_X and NH_3, we assume the share of each nitrogen fertilizer among total N usage is identical to that in 2010 (USDA 2010). For switchgrass and miscanthus, N solutions will likely be the primary fertilizers used in the model year (Turhollow 2011) and are assumed to be the only nitrogen fertilizers applied to switchgrass and miscanthus in this analysis. No additional nitrogen fertilizer is assumed necessary for feedstocks from the forestry sector. The fractional share of nitrogen fertilizer applied to each crop is listed in table 9A.7.

Table 9A.7 | Share of N Fertilizers, by Type (from USDA 2010), for Each Feedstock.

Type of Fertilizer	Fractional Share of nitrogen Fertilizer (N_{share}) for CG, CS, and WS	Fractional Share of nitrogen Fertilizer (N_{share}) for SG and MS
Anhydrous ammonia	0.34	0
Ammonium nitrate	0.03	0
Urea	0.25	0
Ammonium sulfate	0.03	0
Nitrogen solutions	0.35	1

Acronyms: CG = corn grain; CS = corn stover; WS = wheat straw; SG = switchgrass; and MS = miscanthus.

For all feedstocks, the fertilizer-specific EF, in pounds of NO or NH_3 per pound of N (lb pollutant/lb N) applied, are given by equation 9A.11 and equation 9A.12.

Equation 9A.11:

$$EF_{NO,F} = \left(\frac{\%V_{NO}}{100}\right)\left(\frac{30}{14}\right)$$

Equation 9A.12:

$$EF_{NH3,F} = \left(\frac{\%V_{NH3}}{100}\right)\left(\frac{17}{14}\right)$$

Where the following are defined as

- *F* is the type of fertilizer
- % V_{NO} or NH_3 is the percentage of N in the fertilizer that is volatized as NO or NH_3 (100% * lb pollutant/lb N provided in table 9A.8)
- The factors 30/14 and 17/14, respectively, convert the amount of N to NO and NH_3 via the molecular weight of the pollutant versus that of N.

Table 9A.8 | N content (EPA 2015b) and the Amount of N Volatilized as Nitric Oxide (% V_{NO} from EPA 2015b and ANL 2015) and NH_3 (% V_{NH3})from EPA 2015b and Davidson et al. 2004) for Five Types of Commonly Used N Fertilizers.

Type of Fertilizer	N Content (%)	$\%V_{NO}$	$\%V\ NH_3$
Anhydrous ammonia	82	0.79	4.0
Ammonium nitrate	36	3.8	1.91
Urea	46	0.9	15.8
Ammonium sulfate	22	3.5	9.53
Nitrogen solutions	29	0.79	8.0

For stover and straw, the amount of emissions of pollutant, *P*, from fertilizer, *F*, for feedstock, feed, in county, *c*, (in lb/yr) is given by equation 9A.13.

Equation 9A.13:

$$E_{P,F,feed,c} = Prod_{feed,c} * N_{app,feed,c} * N_{share,F} * EF_{P,F}$$

Where the following are defined as

- $Prod_{feed,c}$ is the amount of feedstock, *feed*, produced in dt in county, *c*, per year
- $N_{app,feed,c}$ is the amount of N applied in pounds per dt of feedstock, feed, in county, *c*
- $N_{share,F}$ is the share of N in fertilizer, *F*, as compared to all fertilizers in pounds of N in *F* per pound of N in all fertilizers (given in table 9A.7)
- $EF_{P,F}$ is the emission factor for pollutant, *P*, from fertilizer, *F*, in pounds of pollutant per pound of N in *F* (given by the equation 9A.11 and equation 9A.12).

For corn grain, switchgrass, and miscanthus the amount of emissions E_(P,F,c) (in lb/yr) generated by pollutant, *P*, from fertilizer, *F*, in county, *c*, is given by equation 9A.14.

Equation 9A.14:

$$E_{P,F,feed,c} = Harv_{feed,c} * N_{app,feed,c} * N_{share,F} * EF_{P,F}$$

Where the following are defined as

- $Harv_{feed,c}$ is the amount acreage of feedstock, feed, harvested in acres per year in county, *c* (DOE 2016)
- $N_{app,feed,c}$ is the amount of N applied in county, c, in pounds per harvested acre of feedstock, *feed* (DOE 2016)
- $N_{share,F}$ is the share of N in fertilizer, *F*, as compared to all fertilizers in pounds of N in *F* per pound of N in all fertilizers (given in table 9A.7)
- $EF_{P,F}$ is the emission factor for pollutant, P, from fertilizer, F, in pounds of pollutant per pound of N in *F* (given by the equation 9A.11 and equation 9A.12).

For each type of feedstock, the total amount of emissions of pollutant, *P*, associated with feedstock, feed, in county, *c*, $E_{P,fert,feed,c}$ (in lb/year), from all fertilizer application is calculated by summing emissions over all five types of fertilizers, such that we have equation 9A.15.

Equation 9A.15:

$$E_{P,\,fert,\,feed,\,c} = \sum_{i=1}^{5} E_{P,\,Fi\,,\,feed,\,c}$$

Where the following are defined as

- $E_{P,Fi,feed,c}$ is the amount of emissions of pollutant, P, from fertilizer, F_i, associated with feedsock, feed, in county, c, given by equation 9A.13 or 9A.14
- F_1 is anhydrous ammonia
- F_2 is ammonium nitrate
- F_3 is urea
- F_4 is ammonium sulfate
- F_5 is N solutions.

9A.2.6 Pesticide Emissions

The application of pesticides (e.g., herbicides, insecticides, and fungicides) results in the emission of VOCs. The estimation of emissions from pesticides is challenging due to the wide range of formulations (e.g., emulsifiable concentrate, aerosol, solution, flowable, granule), application equipment, and application type (e.g., band, broadcast, serial, spot). Although the Emission Inventory Improvement Program (EIIP) describes the preferred methodology for computing the amount of emissions generated by pesticide application (EPA 2001), this methodology requires a large amount of information that was unavailable for this study. As a result, we used the method used in the 2011 National Emission Inventory (Huntley 2015). According to this method, the total pesticide emissions, $E_{pest,feed,c}$ (in lb of VOC per year in county, c), by feedstock, *feed*, are given by equation 9A.16.

Equation 9A.16:

$$E_{pest,\, feed,\, c} = Harv_{feed,\, c} * R_{feed,\, c} * ER * C_{VOC}$$

Where the following are defined as

- $Harv_{feed,c}$ is the harvested acreage of feedstock, feed, in county, c
- $R_{feed,c}$ is the amount of pesticide applied to feedstock, feed, per harvested acre in county, c (lb pesticide/acre) (DOE 2016)
- ER is the evaporation rate (ratio; default value = 0.9)
- C_{VOC} is the VOC content (lb VOC/lb active ingredient; default value = 0.835).

9A.2.7 Fugitive Dust Emissions

We assume that there are no fugitive dust emissions from preprocessing equipment at the biorefinery in the near-term system or at the depot in the long-term system because the design cases that serve as the basis for equipment preprocessing assumptions used in Supply Characterization Model (SCM) modeling (DOE 2016) have a baghouse or other emission control system in place (INL 2013; INL 2014). Although whole-tree biomass and residue chipping are likely to generate fugitive emissions, no EFs for fugitive dust were identified for the operation.

After reviewing the literature and having discussions with regulatory experts at EPA, the California Air Resources Board (CARB), and researchers at the Consortium for Research on Renewable Industrial Materials (CORRIM), we concluded that data and methods are not currently available for estimating fugitive dust from forestry sector biomass production (chapter 9, section 9.4.2.3).

9A.2.7.1 Agriculture Harvest and Non-Harvest Activities

Agricultural activities include airborne soil PM emissions produced during the preparation of agricultural lands for planting, harvesting, and other activities. For example, dust emissions are produced by the mechanical disturbance of the soil by the implement used and the tractor pulling it (WRAP 2006).

According to research performed at the University of California, Davis, and summarized by CARB (Gaffney and Yu 2003; CARB 2003), the EFs for all types of agricultural land preparation (non-harvest) activities can be classified into one of five categories (table 9A.9). Additional EFs were also reported (by feedstock type) for harvest activities associated with three crop types (CARB 2003; table 9A.10).

Table 9A.9 | EFs for Fugitive Emissions of PM Generated by Agricultural Non-Harvest Activities (Gaffney and Yu 2003).

Category	Emission Factor (lbs PM_{10}/acre-pass)
Root cutting	0.3
Discing, tilling, chiseling	1.2
Ripping, subsoiling	4.6
Land planning and floating	12.5
Weeding	0.8

Table 9A.10 | EFs for Fugitive Emissions of PM Associated with Harvesting Cotton, Wheat, and Almonds (CARB 2003).

Harvest Operation	Emission Factor (lbs PM_{10}/acre)
Cotton picking	1.7
Cotton stalking	1.7
Cotton total	**3.4**
Wheat combining	5.8
Wheat total	**5.8**
Almond shaking	0.37
Almond sweeping	3.7
First almond pickup	36.7
Almond total	**40.8**

In consultation with agricultural experts, CARB (2003) used scaling factors to expand its analysis and approximate EFs for other crops (see table 9A.11 for scaling factors associated with FPEAM feedstocks). Since harvest EFs tend to be fairly unique for each crop, all harvest operations were combined into a single factor that included all relevant operations (CARB 2003). As a result, harvest EFs are reported per acre rather than per acre-pass. Although the scaling factors for corn grain and wheat came directly from CARB (2003), switchgrass and corn stover were not included so we assumed that these crops would be similar to corn grain (table 9A.11).

Table 9A.11 | EFs for Fugitive Emissions of PM Generated by Agricultural Harvest Activities Associated with FPEAM Feedstocks (Derived from CARB 2003).

Feedstock Type	Scaling Factor	Crop Proxy	Emission Factor (lbs PM_{10}/acre)
Corn grain	0.5[a]	Cotton total	1.7
Corn stover	0.5[b]	Cotton total	1.7
Wheat straw	1[a]	Wheat total	5.8
Switchgrass	0.5[b]	Cotton total	1.7
Miscanthus	0.5[b]	Cotton total	1.7

[a] CARB 2003

[b] Assumed similar to corn grain

We classified each of the non-harvest activities into the categories outlined in table 9A.9. We then used these EFs for each category to compute the fugitive dust emissions for each type of machinery in pounds of PM10 per acre of feedstock (see tables 9A.12–9A.16). By summing the estimated emissions generated from all field activities, we evaluated the total fugitive dust emissions associated with harvest and non-harvest activities for all feedstock types (see table 9A.17). Due to the product-purpose allocation approach that we use for corn stover and wheat straw, there are no non-harvest fugitive dust emissions associated with these two crops. We only incorporate the additional emissions that would result from additional crop harvesting activities for corn stover and wheat straw.

Table 9A.12 | EFs for Fugitive Emissions of PM for Non-Harvest Activities Associated with Switchgrass (Derived from Gaffney and Yu 2003 and National Crop Budgets in Zhang et al. 2016).

Category	Field Activity	Passes Over Field	Fugitive Dust Emissions (lbs PM_{10}/acre)
Establishment – Year 1			
Discing, tilling, chiseling	Offset disk	2	2.4
Weeding	Fertilizer and lime spreader	2	1.6
Weeding	Boom sprayer	3	2.4
Discing, tilling, chiseling	No-till drill	1	1.2
Maintenance – Years 2–10			
Discing, tilling, chiseling	Reseeding (year 2 only)	1	1.2
Weeding	Fertilizer and lime spreader	1	0.8
Weeding	Boom sprayer, 50 ft (year 5 only)	1	0.8

Table 9A.13 | EFs for Fugitive Emissions of PM for Non-Harvest Activities Associated with Conventional Till Corn Grain (Derived from Gaffney and Yu 2003 and National Crop Budgets in Zhang et al. 2016).

Category	Field Activity	Passes Over Field	Fugitive Dust Emissions (lbs PM_{10}/acre)
Weeding	Dry fertilizer spreader	1	0.8
Weeding	Chemical applicator GE 30ft	1	0.8
Weeding	Chemical applicator GE 30ft	1	0.8
Weeding	Fertilizer applicator	1	0.8
Discing, tilling, chiseling	Eight-row planter	1	1.2
Discing, tilling, chiseling	Field cultivator	1	1.2
Discing, tilling, chiseling	Tandem disk	1	1.2
Discing, tilling, chiseling	Moldboard plow	1	1.2

Table 9A.14 | EFs for Fugitive Emissions of PM_{10} for Non-Harvest Activities Associated with Reduced Till Corn Grain (Derived from Gaffney and Yu 2003 and National Crop Budgets in Zhang et al. 2016).

Category	Field Activity	Passes Over Field	Fugitive Dust Emissions (lbs PM_{10}/acre)
Weeding	Dry fertilizer spreader	1	0.8
Weeding	Dry fertilizer spreader	1	0.8
Discing, tilling, chiseling	Row cultivator	1	1.2
Discing, tilling, chiseling	Eight-row planter	1	1.2
Weeding	Chemical applicator	1	0.8
Discing, tilling, chiseling	Tandem disk	1	1.2
Discing, tilling, chiseling	Offset disk/light duty	1	1.2

Category	Field Activity	Passes Over Field	Fugitive Dust Emissions (lbs PM_{10}/acre)
Weeding	Dry fertilizer spreader	1	0.8
Weeding	Dry fertilizer spreader	1	0.8
Weeding	Chemical applicator	1	0.8
Weeding	Dry fertilizer spreader	1	0.8
Weeding	Chemical applicator	1	0.8
Discing, tilling, chiseling	Seven-row no-till planter	1	1.2

Table 9A.16 | EFs for Fugitive Emissions of PM10 for Non-Harvest Activities Associated with Miscanthus (Derived from Gaffney and Yu 2003, Mari et al. 2002, and National Crop Budgets in Zhang et al. 2016).

Category	Field Activity	Passes Over Field	Fugitive Dust Emissions (lbs PM_{10}/acre)
	Establishment – Year 1		
Weeding	Mower	1	0.8
Discing, tilling, chiseling	Offset disk	2	2.4
Ripping, subsoiling	Ripper bedder (deep tillage)	1	4.6
Weeding	Fertilizer and lime spreader	2	1.6
Weeding	Boom sprayer	3	2.4
Discing, tilling, chiseling	Potato planter	1	1.2
	Maintenance – Years 2–10		
Weeding	Fertilizer and lime spreader	1	0.8
Weeding	Boom sprayer, 50 ft (year 5 only)	1	0.8

By summing the emissions over all of the machinery used during each year, we compute the total PM_{10} per acre of feedstock harvested (tables 9A.12–9A.16). As shown, the emissions vary with tillage method for corn grain. The total harvest and non-harvest fugitive dust emissions of PM_{10} , $E_{PM10,FDharv/nonharv,c}$, (in lb/yr) for each feedstock, feed, in county, *c*, are given by equation 9A.17.

Equation 9A.17:

$$E_{PM10,\ FDharv/nonharv,\ feed,\ c} = Harv_{\ feed,\ c} * (EF_{feed,Harv,T} + EF_{feed,\ Nonharv})$$

Where the following are defined as

- $Harv_{feed,c}$ is the amount of harvested area of feedstock, feed, in county, *c*, per year (acre/yr)
- $EF_{feed,Harv}$ and $EF_{feed,Nonharv}$ are EFs (lb/acre) for feedstock, *feed*, from tables 9A.17–9A.19 by tillage type.

Based on the Midwest Research Institute (MRI 2006), we assume that the ratio of $PM_{2.5}$ to PM_{10} for fugitive dust emissions is 0.2.

Table 9A.17 | Total PM_{10} Fugitive Dust Emissions Associated with Harvest and Non-Harvest Activities for Corn Grain, Corn Stover, and Wheat Straw (Derived from Gaffney and Yu 2003, CARB 2003, and National Crop Budgets in Zhang et al. 2016).

	Corn Grain (lbs PM_{10}/ac)			Stover (lbs PM_{10}/ac)	Straw (lbs PM_{10}/ac)
	Conventional Till	Reduced Till	No-Till		
Non-Harvest	8	7.2	5.2	–	–
Harvest	1.7	1.7	1.7	1.7	5.8

Table 9A.18 | Total PM_{10} Fugitive Dust Emissions Associated with Harvest and Non-Harvest Activities for Miscanthus (Derived from Gaffney and Yu 2003, CARB 2003, and National Crop Budgets in Zhang et al. 2016).

Year	Total Emissions (lbs PM_{10}/acre)	
	Harvest	Non-Harvest
1	1.7	13
2	1.7	1.6
2–15	1.7	0.8

Table 9A.19 | Total PM_{10} Fugitive Dust Emissions Associated with Harvest and Non-Harvest Activities for Switchgrass (Derived from Gaffney and Yu 2003, CARB 2003, and National Crop Budgets in Zhang et al. 2016).

Year	Total Emissions (lbs PM_{10}/acre)	
	Harvest	Non-Harvest
1	1.7	7.6
2	1.7	2
3–4	1.7	0.8
5	1.7	1.6
6–10	1.7	0.8

9A.2.7.2 Transportation on All Roads

EPA has established methods for estimating fugitive dust emissions from road travel, which vary by road type (EPA 2006). The number of miles traveled by road type (e.g., unpaved, primary paved, and secondary paved) for biomass transportation were not available from *BT16* volume 1. As a result, we used national averages to estimate distances traveled on each road type (INL 2016). For each feedstock in each county, we subdivided the total distance traveled, *D*, during biomass supply logistics (DOE 2016) by road type based on the national average in table 9A.20.

Table 9A.20 | Biomass Supply Logistics Distances, where Total Distance Traveled (D) Is Split among Each Road Type (INL 2016).

Variable	Agricultural Feedstocks	Forestry Feedstocks
$D_{unpaved}$	$D \leq 2$ miles	$D \leq 10$ miles
$D_{secondary\ paved}$	$D > 2$ miles	$D > 10$ miles
	$D \leq 50$ miles	$D \leq 50$ miles
$D_{primary\ paved}$	$D > 50$ miles	$D > 50$ miles

9A.2.7.3 Transportation on Unpaved Roads

According to EPA (2006), for vehicles traveling on unpaved surfaces under similar conditions to those found at industrial sites (i.e., surface silt content of 1.8%–25.2%, mean vehicle weight from 2–290 tons and mean vehicle speed from 5–43 mph), the fugitive dust emission rate ($ER_{FDunpaved,st}$ in state, *st*, in lb per vehicle mile traveled) are given by equation 9A.18.

Equation 9A.18:

$$ER_{FDunpaved,st} = k\left(\frac{s_{st}}{12}\right)^{a}\left(\frac{W}{3}\right)^{b}$$

Where the following are defined as

- *k*, *a*, and *b* are empirical constants listed in table 9A.21
- s_{st} is the surface material silt content (percentage; values vary by state, st, according to EPA 2006)
- *W* is the mean weight of the vehicles on the road (3.2 tons) (FHWA 2000).

Table 9A.21 | Empirical Constants Used for Determining Fugitive Dust Emissions from Unpaved Industrial Roads (EPA 2006).

Constant	PM_{10}	$PM_{2.5}$
k (lb/VMT)	1.5	0.15
A	0.9	0.9
B	0.45	0.45

Acronyms: lb = pounds;
VMT = vehicle miles traveled

Equation 9A.18 was modified for use in FPEAM to estimate the total amount of fugitive dust emissions of pollutant, P (PM_{10} or $PM_{2.5}$), generated by transportation on unpaved roads in county, *c*, $E_{P,FDunpaved,feed,c}$ (in lb per year), for each feedstock in each biomass supply logistics system and is given by equation 8A.19.

Equation 9A.19:

$$E_{unpaved,\,FD,\,feed,\,c} = \frac{Supply_{feed,c}}{C_{feed}} * D_{unpaved,\,feed} * k\left(\left(\frac{s_{st}}{12}\right)^{ap}\left(\frac{W}{3}\right)^{bp}\right)$$

Where the following are defined as

- $Supply_{feed,c}$ is the amount of feedstock, *feed*, supplied per year in county, *c* (dt/yr)
- *C* is the capacity of the truck hauling the feedstock (dt/load)
- $D_{unpaved,feed}$ is the distance that feedstock, feed, travels in vehicle miles traveled on unpaved roads (mi) (see table 9A.20)
- s_{st} and *W* are given by equation 9A.18
- k_P, a_P, and b_P are the constants for pollutant, *P* (see table 9A.21).

9A.2.7.4 Transportation on Paved Roads

According to EPA (2011), for vehicles traveling on paved surfaces, the fugitive dust emission rate ($ER_{P,FDpaved,c}$) in lb of pollutant, *P*, per vehicle mile travelled in county, *c*) are given by equation 9A.20.

Equation 9A.20:

$$ER_{P,FDpaved,c} = k_P * sL^{ap} * W^{bp}$$

Where the following are defined as

- k_P,a_P,and b_P are empirical constants listed in table 9A.22 for pollutant, P
- sL is the road surface silt loading (g/m^2) on secondary (0.4 g/m^2) and primary (0.045 g/m^2) paved roads
- W is the mean weight of the vehicles on the road (3.2 tons) (FHWA 2000).

Table 9A.22 | Empirical Constant Used for Determining Fugitive Dust Emissions from Paved Industrial Roads (EPA 2011).

Constant	PM_{10}	$PM_{2.5}$
***k* (lb/VMT)**	0.0022	0.00054
A	0.91	0.91
B	1.02	1.02

Acronyms: lb = pounds;
VMT = vehicle miles traveled

Equation 9A.21:

$$E_{P,FDsec\ unpaved,\ FD,\ feed,\ c} = \frac{Supply_{feed,c}}{C_{feed}} * D_{sec\ unpaved,\ feed} * k_P * sL_{sec}^{\ ap} * W^{bp}$$

Equation 9A.22:

$$E_{P,FDpri\ unpaved,\ FD,\ feed,\ c} = \frac{Supply_{feed,c}}{C_{feed}} * D_{pri\ unpaved,\ feed} * k_P * sL_{pri}^{\ ap} * W^{bp}$$

Where the following are defined as

- $Supply_{feed,c}$ is the amount of feedstock, *feed*, supplied per year in county, *c* (dt/yr)
- C_{feed} is the capacity of the truck hauling feedstock, *feed* (dt/load)
- k_P, a_P, and b_P are given by equation 9A.20
- sL_{sec} and sL_{pri} are the road surface silt loading (g/m^2) on secondary and primary paved roads, respectively (see equation 9A.20)
- $D_{secpaved,feed}$ and $D_{pri\ paved,feed}$ are the distances that feedstock, *feed*, travels in vehicle miles traveled on secondary and primary paved roads (mi), respectively (see table 9A.20).

9A.2.7.5 Limitations of Transport Fugitive Dust Calculations

There are two main limitations to the paved road fugitive dust equations described above. First, these equations were derived using a regression analysis of experimental data, including 83 road tests on public, paved, and controlled and uncontrolled industrial paved roads. Second, these conditions may not be representative of the source conditions used in our analysis as performance is based on equipment used in the 1970s. The paved road fugitive dust equations were found to be of good quality using EPA's AP-42 data quality scoring system (score of A) for the range of source conditions listed in table 9A.23, which encompasses the source conditions used in our analysis.

Table 9A.23 | Comparison of Source Condition Ranges Where the Fugitive Dust Equations Are To Be Deemed of High Quality and Where Biomass Transportation Is Expected to Occur for Paved Roads (EPA 2011; equation 9A.20).

Parameter	Range of Source Conditions Where the Equations are Deemed to be High Quality	Range of Source Conditions Employed for Biomass Transportation Analysis
Silt loading (g/m^2)	0.03–400	0.045 (primary), 0.4 (secondary)
Mean vehicle weight (Mt)	1.8–38	3.2

The unpaved road fugitive dust equations were also determined empirically and are considered to be of fairly high quality by EPA (score of B) under certain source conditions. Like the paved road equations, the source conditions for the unpaved road fugitive emissions equation align fairly well with the scenario conditions (table 9A.24).

Table 9A.24 | Comparison of Source Condition Ranges Where the Fugitive Dust Equations Are To Be Deemed of High Quality and Where Biomass Transportation Is Expected to Occur for Unpaved Roads (EPA 2006; equation 9A.18).

Parameter	Range of Source Conditions Where the Equations are Deemed to be High Quality	Range of Source Conditions Employed for Biomass Transportation Analysis
Silt loading (%)	1.8–25.2	0-7.2a
Mean vehicle weight (Mt)	1.8–260	3.2

[a] EPA Unpaved Road Surface Material Silt Content Values used in the 1999 NEI (https://www3.epa.gov/ttn/chief/ap42/ch13/related/c13s02-2.html)

However, these equations do not include a reduction factor for precipitation, which is known to have an impact of fugitive dust generation. The AP-42 does provide an equation for paved road fugitive dust emissions that includes a precipitation correction term. However, this precipitation correction equation has not been rigorously verified and is considered to be of lower quality than the standard equation. As a result, we use the equation without the precipitation correction factor.

Several other limitations of the fugitive dust emission equations relate to data availability. For example, silt content varies spatially, and the data are not readily available to identify the silt content for unpaved roads for each county. As a result, we use constant values for silt content for primary and secondary paved roads and state averages for unpaved roads (EPA 2006). In addition, we use national data to estimate the fractional amount of travel that occurs on each road type. However, in practice, the distance travelled on each road type would likely vary widely on a county level.

Finally, it is important to note that we only report source emissions of fugitive dust. We do not account for the fraction of particulate matter that might be deposited or dispersed by surrounding vegetation or other roughness elements near the source. Several studies indicate that the fraction of particulate matter relevant to air quality analyses may be much smaller than the source emissions (Watson and Chow 2000; Cowherd, Grelinger, and Gebhart 2005; Pace 2005; Pardyjak et al. 2008; Janhäll 2015). Thus, if our results are used in air quality models, potential transportable fractions (e.g., fractions described in the 2011 air quality modeling platform for National Emissions Inventory (NEI) data (EPA 2015c)) should be considered.

9A.2.7.6 Preprocessing Fugitive Dust Sensitivity Analysis

As noted in chapter 9, section 9.2.2, we assume fugitive dust from preprocessing equipment to be zero due to the dust collection systems assumed to be in place in both near-term and long-term supply logistics designs (INL 2013; INL 2014). In section 9.3.4.3 of chapter 9, we discuss and compare the emissions of 99% dust collection to other sources of PM emissions. According to EPA (1999), baghouse air pollution control technologies may not be completely (or 99.9% complete) effective at dust collection due to the age of the equipment or whether a high-quality enclosed system is installed.

We estimate potential emissions of PM_{10} and $PM_{2.5}$ based on the following equations and data taken from Krause and Smith (2006). For a single county, and for each feedstock category, *FC* (woody or herbaceous), the particulate emissions from preprocessing[4] ($ER_{preprocess\ FD}$ in lb/dt) are calculated by equation 9A.23.

Equation 9A.23:

$$EF_{preprocess\ PM,\ FC} = PR_{FC} * gL * \frac{1}{7000} * Air\ Flow\ Rate * 60 * 8760$$

[4] All PM is assumed to be less than 10 μm in diameter. $PM_{2.5}$ emissions are assumed to be 17% of PM_{10} emissions (WLA Consulting 2011).

Where the following are defined as

- PR_{FC} is the processing throughput rate for the feedstock category, *FC*
- *gL* is 0.004 (grain loadings/ft^3) (WLA Consulting 2011)
- 1/7000 (lbs/grains) is a constant
- The assumed *Air Flow Rate* is 51,000[5] (scfm) (Davis et al. 2013)
- 60 (min/hr) is a constant
- 8,760 (hrs/yr) is a constant.

9A.2.8 Other Emissions from Preprocessing and Drying

The preprocessing and drying of woody feedstocks is expected to generate VOC emissions. Based on the INL (2013, 2014) design reports, which are used as the basis for biomass supply logistics in SCM modeling (DOE 2016), the near-term and long-term logistics systems use an indirect heat rotary dryer and a cross-flow dryer, respectively. They also both use a hammer mill for preprocessing.

We used EPA's (2002) VOC EFs for wood preprocessing and drying equipment to estimate these emissions. EPA provides VOC EFs for rotary dryers and hammer mills, but not for cross-flow dryers. Mechanically, the conveyor dryer in EPA (2002) most closely resembles the cross-flow dryer but not in terms of the drying temperature. Therefore, we use the conveyor dryer EFs that assume the use of a regenerative catalytic oxidizer in order to approximate potential VOC emissions. Table 9A.25 summarizes the softwood and hardwood EFs that FPEAM uses to estimate VOC emissions from drying a 50/50 split mixedwood.

For each feedstock in each biomass supply logistics system, the total amount of VOC emissions generated by drying and preprocessing in county, *c* ($E_{Drying\ and\ preprocessing,c}$ in lb/yr), is given by equation 9A.24.

Equation 9A.24:

$$E_{Drying\ and\ Preprocessing,\ feed} = Supply_{feed,\ c} * (EF_{drying} + EF_{milling}) * 2.2$$

Where the following are defined as

- $Supply_{feed,c}$ is the amount of feedstock, feed, supplied to biorefineries in county, *c* (dt/yr) (DOE 2016)
- EF_{drying} and $EF_{milling}$ are VOC emission factors (kg/dt) (see table 9A.25)
- 2.2 converts kg to lb.

[5] 51,000 (8,500 x 6 baghouse) is used for a facility.

Table 9A.25 | Softwood, Hardwood, and Mixedwood VOC EFs from EPA (2002).

Equipment	Biomass Supply Logistics System	Softwood EF (kg/dt)	Hardwood EF (kg/dt)	50/50 Mixed Wood EF (used in Eq S22) (kg/dt)
Indirect Heat Rotary Dryer	Near-Term	0.92	0.13	0.53
Conveyor[6] Natural Gas Dryer Heating and Cooling Zones	Long-Term	0.41	0.034	0.23
Flaker/Refinery/ Hammer Mill	Near-Term and Long-Term	–	–	0.52

9A.3 Supplemental Results

9A.3.1 SO_x, NO_x, and CO

Figure 9A.1 shows the locations of counties in nonattainment with the National Ambient Air Quality Standards (NAAQS) for SO_2 for the two BC1&ML scenarios. Upwind travel of SO_2 emissions is limited, so only changes in SO_2 in nonattainment areas (NAAs) are discussed in the main chapter.

No county is out of compliance with the current NO_2 and CO NAAQS (EPA 2016b), so we display maps (fig. 9A.2 and 9A.3) to illustrate the spatial distribution of county-level emission ratio. In the 2040 scenario, the maximum change in the NEI ratio for attainment counties for CO from *BT16* biomass production and supply scenarios is 3%. The maximum change in the NEI ratio for producing biomass is 18%. Counties having NO_2 emission ratios greater than 18% is the result of transporting biomass long distances to multiple surrounding counties for biofuel production. Due to the limitations of our analysis, all emissions from those long transportation distances are allocated to the biomass producing counties, and therefore interpretation of these high values is not possible with the long distance biomass traveled.

[6] VOC EFs for a cross-flow grain dryer are not available from EPA. Expert consultation indicated the conveyor dryer was a close approximate.

Figure 9A.1 | BC1&ML 2017 and 2040 scenarios' county-level distributions of emission ratios for SO_2 (top frame).[7] Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for SO^2 (primary, 1-hour) (EPA 2016c)[8] are displayed in red in the 2017 (middle frame) and 2040 (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[7] See the main text for a complete list of counties in partial nonattainment with emission ratios above 1%.

[8] Includes NAA designations for the 1971 and 2010 NAAQS. Includes NAA designations EPA maintains based on prior year standards. EPA considers older standards for certain pollutants and we follow EPA in this respect.

Figure 9A.2 | BC1&ML 2017 and 2040 scenarios' county-level distributions of emission ratios for NO_2 (top frame). Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for NO_2 (primary, 1-hour, and 1-year) (EPA 2016c)[9] are displayed in red in the 2017 (middle frame) and 2040 (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[9] Includes NAA designations for the 1971 NAAQS.

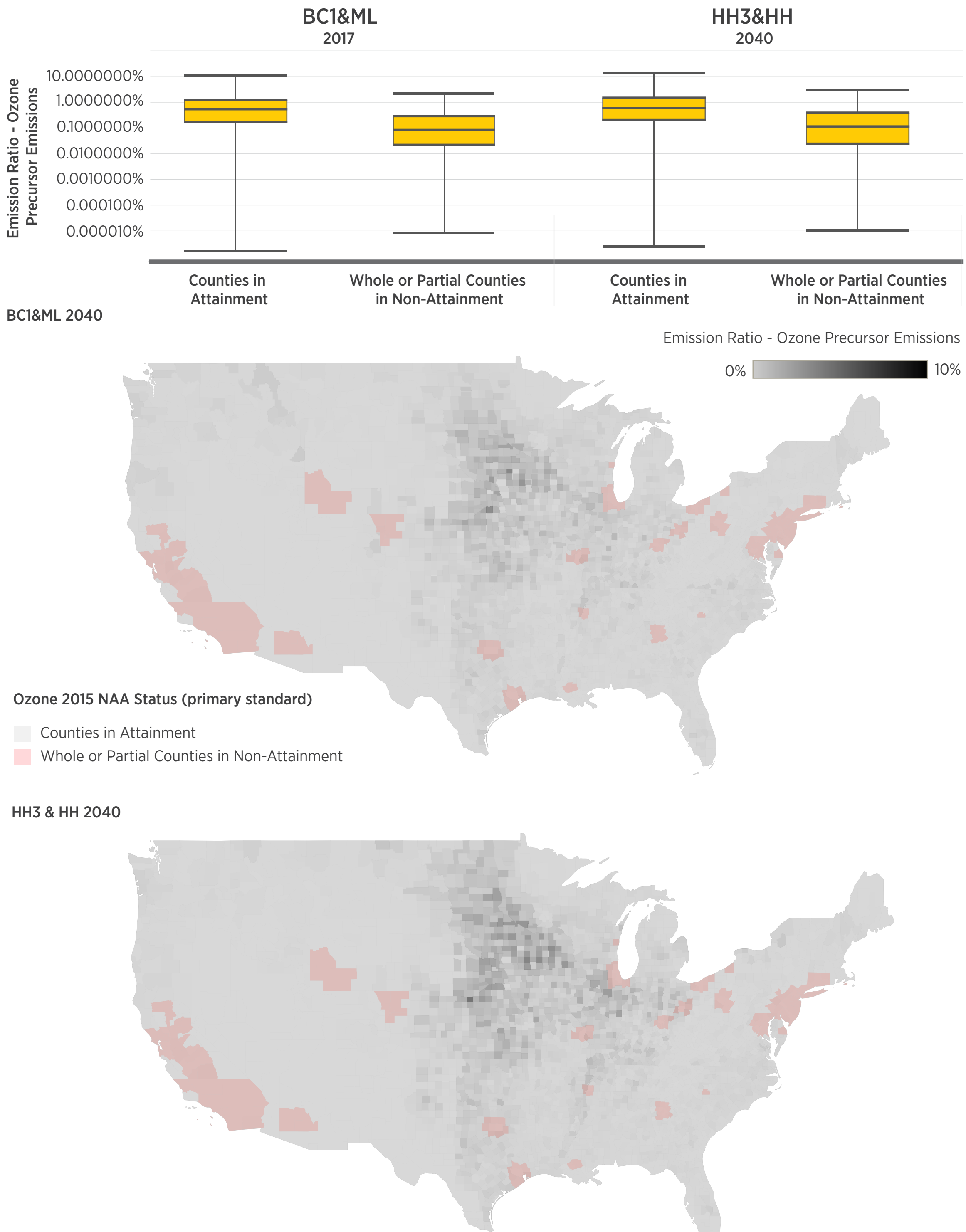
Figure 9A.3 | BC1&ML 2017 and 2040 scenarios' county-level distributions of emission ratios for CO (top frame). Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for CO (primary, 8-hour, and 1-hour) (EPA 2016c)[10] are displayed in red in the 2017 (middle frame) and 2040 (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[10] Includes NAA designations for the 1971 NAAQS.

9A.3.2 High Yield

Figures 9A.4–9A.9 compare the BC1&ML 2040 scenario to the HH3&HH 2040 scenarios.

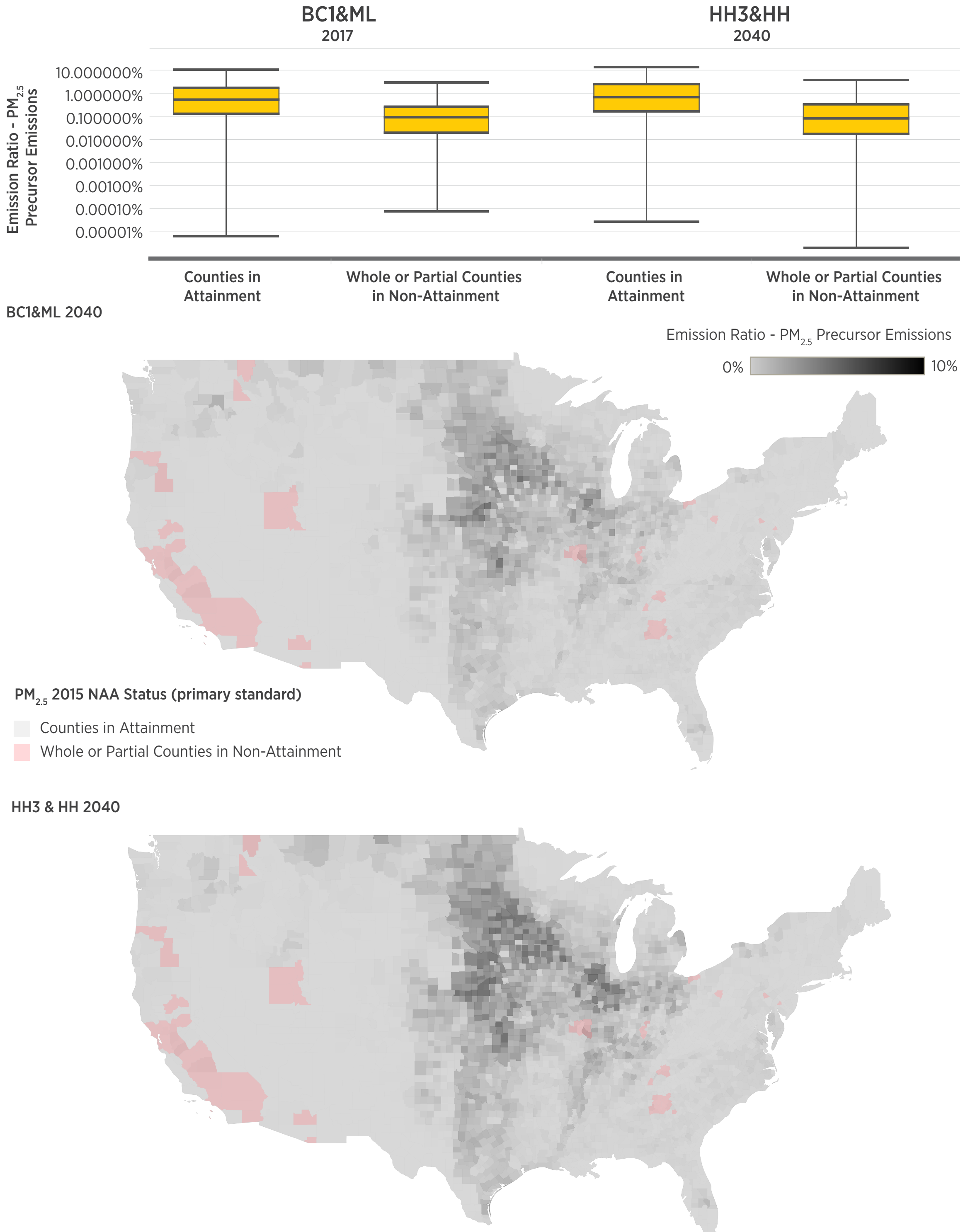
Figure 9A.4 | BC1&ML and HH3&HH 2040 scenarios' county-level distributions of emission ratios for ozone (top frame).[11] Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for ozone (primary, 8-hour) (EPA 2016c)[12] are displayed in red in the BC1&ML (middle frame) and HH3&HH (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



[11] See the main text for a complete list of nonattainment counties with emission ratios above 1%.

[12] Includes NAA designations for the 2008 NAAQS that are still in force.

Figure 9A.5 | BC1&ML and HH3&HH 2040 scenarios' county-level distributions of emission ratios for $PM_{2.5}$ (top frame). Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for $PM_{2.5}$ (primary, 24-hour, and 1-year) (EPA 2016c) are displayed in red in the BC1&ML (middle frame) and HH3&HH (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.



[13] See the main text for a complete list of nonattainment counties with emission ratios above 1%.

[14] Includes NAA designations for the 1997, 2006, and 2012 NAAQS that are still in force.

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Figure 9A.6 | BC1&ML and HH3&HH 2040 scenarios' county-level distributions of emission ratios for PM_{10} (top frame).[15] Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for PM_{10} (primary, 24-hour) (EPA 2016c)[16] are displayed in red in the BC1&ML (middle frame) and HH3&HH (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[15] See the main text for a complete list of nonattainment counties with emission ratios above 1%.

[16] Includes NAA designations for the 1987 and 2012 NAAQS that are still in force.

Figure 9A.7 | BC1&ML and HH3&HH 2040 scenarios' county-level distributions of emission ratios for SO_2 (top frame)[17] Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for SO_2 (primary, 1-hour) (EPA 2016c)[18] are displayed in red in the BC1&ML (middle frame) and HH3&HH (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[17] See the main text for a complete list of counties in partial nonattainment with emission ratios above 1%.

[18] Includes NAA designations for the 1971 and 2010 NAAQS that are still in force.

Figure 9A.8 | BC1&ML and HH3&HH 2040 scenarios' county-level distributions of emission ratios for NO_2 (top frame). Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for NO_2 (primary, 1-hour, and 1-year) (EPA 2016c)[19] are displayed in red in the BC1&ML (middle frame) and HH3&HH (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[19] Includes NAA designations for the 1971 NAAQS that are still in force..

Figure 9A.9 | BC1&ML and HH3&HH 2040 scenarios' county-level distributions of emission ratios for CO (top frame). Maps of emission ratios and nonattainment counties at the end of 2015 exceeding NAAQS standards for CO (primary, 8-hour, and 1-hour) (EPA 2016c)[20] are displayed in red in the BC1&ML (middle frame) and HH3&HH (bottom frame) maps. Box and whisker plots represent minimum, 25th percentile, median, 75th percentile, and maximum.

[20] Includes NAA designations for the 1971 NAAQS that are still in force.

9A.4 References

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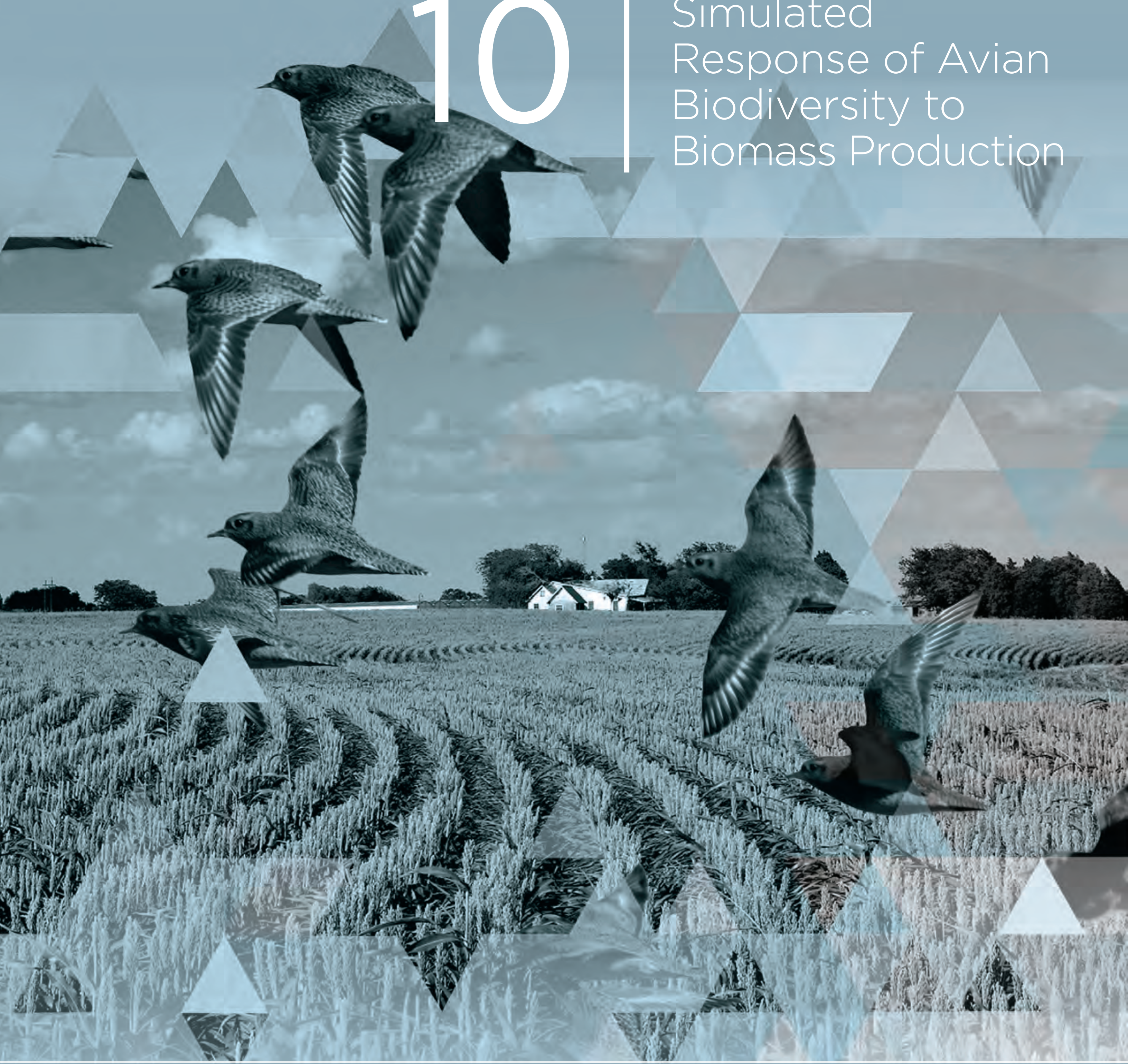
10 | Simulated Response of Avian Biodiversity to Biomass Production



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10.1 Introduction

Compared with other environmental indicators, few U.S. studies have quantified the relationships between production of biomass crops and biodiversity. The two are linked by both direct and indirect causal pathways. Indirectly, a growing bioenergy industry can delay or prevent bioclimatic stress (Cook, Beyea, and Keeler 1991a, b) to wildlife by replacing fossil energy and slowing the rate of climate warming (Dale, Parish, and Kline 2015). This is particularly true for ectotherms, including fish and other aquatic biota (Fenoglio et al. 2010). However, most public concerns center on direct linkages—specifically, how changes in land management to grow biomass will influence biodiversity. Here, we address this question, with a focus on birds that can be expanded to include other taxa in future assessments.

Public concern regarding biomass and its impact on biodiversity has been greatest in the Midwest, where 70% of diverse prairie and wetland ecosystems have been replaced by less-diverse agricultural landscapes (Samson, Knopf, and Ostlie 2004). This negative response in diversity to past changes in land management has taught us that replacing low-intensity, high-diversity land management with high-intensity, low-diversity land management is often accompanied by a reduction in species diversity (Meehan, Hurlbert, and Gratton 2010) and adds to public concern about increasing the agricultural footprint by adding biomass feedstock production. The expansion of corn grown for ethanol has also been raised as a concern for biodiversity (Brooke et al. 2009, Rashford, Walker, and Bastian 2011). However, biomass production can involve wildlife-friendly crops that mimic local native habitat (grasses and short-rotation woody crops [SRWCs]), crops that provide food (e.g., oil-seed crops for biodiesel), and more-intensive use (residue harvest) of existing croplands without expanding into less-managed land.

The analysis presented in this chapter builds on previous research. Many national-scale studies have quantified future changes in wildlife habitat, for example, in response to changes in land cover (Tavernia et al. 2013, Lawler et al. 2014) or climate change [e.g., (Matthews et al. 2011)], but few studies have considered introducing biomass feedstocks into future landscapes. In addition, many large-scale conservation-planning studies that assess the impacts of land-use change assume that all change is bad (Ando 1998, Withey et al. 2012). We relaxed this assumption by explicitly accounting for the value of biomass crops as wildlife habitat. Our approach was inspired, in part, by two earlier studies at the University of California, Berkeley. Both employed spatial optimization to determine the best places to grow bioenergy crops for wildlife and for farmer profit. In a national-scale modeling study, the number of species of concern potentially impacted by replacing pasture with perennial grasses did not increase with increased farmer profit (Evans, Kelley, and Potts 2015). However, a trade-off between biodiversity and farmer profit was evident when biomass was simulated on lands enrolled in the U.S. Department of Agriculture (USDA) Conservation Reserve Program (CRP lands) (Evans, Kelley, and Potts 2015). This study included 322 at-risk vertebrate species known to occupy cropland or grassland habitat. Evans, Kelley, and Potts estimated that 57 avian species might be influenced by conversion of cropland or pasture under a low-demand scenario (7.6 billion liters of fuel) and 119 species might be influenced under a high-demand (22.7 billion liters) scenario. They estimated that 44 avian species might be influenced by conversion of CRP land under a low-demand scenario (7.6 billion liters of fuel) and 85 species might be influenced under a high-demand (22.7 billion liters) scenario. Stoms et al. (2012) allocated biomass crops to lands across the State of California to maximize wildlife habitat and minimize land rents. Feedstocks included irrigated row crops (sugar beets); dryland grain crops (wheat, barley); irrigated grain crops (corn, sorghum, safflower, canola, and camelina); and irrigated perennial grasses. Perennial grasses supported the largest number of wildlife species, followed by irrigated grain crops, dryland grain crops, and sugar beets.

The analysis presented here incorporates information from local field-scale studies into a national-scale assessment of a potential future informed by spatially extensive biodiversity and bioclimatic data. The analysis is based on the BC1 2040 biomass supply scenario, which is described in *BT16* volume 1.

10.2 Scope of Assessment

We developed a modeling framework, Bio-EST (Bioenergy-biodiversity Estimation), to assess the change in species richness (a measure of biodiversity) associated with change in land management to grow biomass crops. This change was evaluated by comparing modeled responses of avian communities to two national-scale landscapes, a recent 2014 Cropland Data Layer (CDL-2014) and a future BC1 2040 landscape. The BC1 2040 scenario assumed a $60/dry ton farmgate price for cellulosic feedstocks and 1%/year yield increases (see BC1 2040 scenario in section 10.3.1).

Bio-EST considers effects on birds from changes associated with growing dedicated energy crops, including perennial grasses, annual crops such as sorghum, and SRWCs. Management assumptions are those of the primary studies used as the bases for analysis, as cited in the methods. For one crop (switchgrass), we compare strip and total harvest, but the effects of residue removal from agricultural lands are not considered in this chapter. The effects of forest residue harvesting (as well as harvesting of other types of forest biomass) on selected forest wildlife species is evaluated in chapter 11.

One challenge faced here was to conduct a national-scale assessment of wildlife response based on field studies that measure the avian habitat value of lands growing biomass feedstocks. The use of species distribution models at the resolution of farmer-owned fields over a national extent made it possible to estimate the distributions of a species of interest at a finer spatial resolution than is typically available with range maps or atlas data (Rondinini et al. 2006).

We focused on birds for several reasons. (1) Birds respond directly to changing vegetation composition and structure at scales relevant to management. Consequently, bird responses to bioenergy croplands have been relatively well studied compared to those of other taxa. (2) Conservation-planning studies have highlighted birds as showing strong responses to land-use change compared with other taxa (Lawler et al. 2014). (3) As a group, birds enjoy high public interest. Bird watching at backyard feeders, enjoyment of birds during outdoor activities, and hunting of game birds are common recreational activities. Consequently, the conservation status of avian fauna generally ranks high among public biodiversity concerns (Batt 2009). Earlier in the past century, native prairie was replaced by agricultural land, leading to substantial declines in birds that depend on grasslands and shrub-lands (Askins et al. 2007, Samson, Knopf, and Ostlie 2004).

Our analysis included many species on the 2008 list of Birds of Conservation Concern (U.S. Fish and Wildlife Service 2008). These include highly valued game species, such as the red-necked pheasant (*Phasianus colchicus*); species with special conservation status, such as Henslow's sparrow (*Ammodramus henslowii*), and the upland sandpiper (*Bartramia longicauda*), as well as more common species, such as the American robin (*Turdus migratorius*) and red-winged blackbird (*Agelaius phoeniceus*). In our analysis, we included species with narrow habitat requirements, such as grassland-obligate and forest-obligate birds (habitat specialists). We also include species with more generalized habitat requirements that use edges or open woodland/savannah and species found both in grassland and forest habitats (appendix A, table 1). Species in our list represent different spatial life histories, including neo-tropical migrants, North American migrants, and year-round resident species (appendix A, table 1) that breed in the eastern United States.

10.3 Methods

Our approach was to estimate and compare avian richness associated with a current year (2014)[1] and a future landscape consistent with the BC1 2040 scenario. Allocation of biomass crops within counties was performed for parcels in the USDA Common Land Unit (CLU) database (USDA 2014), which includes only lands that are associated with USDA farm programs. We refer to this as "downscaling". CLU shapefiles contain agricultural parcels of varying sizes. CLU parcels are the smallest unit of land with common land cover, land management. Each parcel is delineated by a boundary, such as a fence line, road, or waterway. Biomass was not allocated to the remaining lands (i.e., those that are not in the CLU database because they are not in private ownership). The assumption is that public lands will be ineligible to transition to growing dedicated energy crops. Landscapes used in our analyses were classified into the following land use/land cover (LULC) categories (table 10.1).

We projected occupancy for bird species in a future landscape consistent with the BC1 2040 scenario that includes SRWCs, perennial grasses, and sorghum and energy cane. Species distribution models (SDMs) were developed at a resolution of a 1-km raster with assigned LULC classes from table 10.1. SDMs provide local estimates of the probability of occupancy by a species within 1-km raster pixels for the conterminous United States. If a threshold is specified (e.g., a species is considered present if probability is >0.5), probability maps can be converted into species range maps.

Another potentially important local consideration is that wildlife species that are habitat specialists, such as grassland and forest birds, tend to require a

Table 10.1 | LULC Categories, Including Commodity Crops, Matrix Lands, and Dedicated Bioenergy Crops

Conventional Crops in Cropland Data Layer	Dedicated Bioenergy Crops not in the Cropland Data Layer	National Land Cover Data Categories in Cropland Data Layer
Barley	Switchgrass	Evergreen forest
Corn	*Miscanthus*	Mixed forest
Cotton	Energy cane	Hardwood forest
Hay	Pine	Other (water, urban)
Idle	Poplar	
Oats	Willow	
Pasture/grassland	*Eucalyptus*	
Rice		
Sorghum		
Soybeans		
Wheat		

[1] This analysis used the CDL as a baseline instead of the BC1 2017 scenario that was used elsewhere in this report.

minimum habitat area to persist. For example, Blank et al. (2014) found that grassland bird densities were positively associated with surrounding grassland area. To account for this, we developed a method for quantifying wildlife habitat in current and hypothetical future landscapes; these landscapes did not count small areas that failed to exceed species-specific minimum habitat requirements as suitable because of their size.

10.3.1 *BT16* LULC Allocation

"Current" 2014 landscape: We began by assigning an initial LULC class to each CLU parcel, p, as the CDL-2014 class having the largest area within the parcel. For very small parcels, the LULC at the centroid was used.

Future BC1 2040 landscape: We downscaled future LULC categories from county-level Policy Analysis System (POLYSYS) results to USDA CLU parcels. We formulated a mixed-integer optimization to allocate the production of biomass crops to parcels within each county. The problem involved a $p \times k$ matrix of spatial decision variables, $\boldsymbol{X}$, that determined how LULC class k is allocated to parcel p within each county. Each parcel was assigned one crop (i.e., allocation of LULC classes, $\boldsymbol{X}_p$ in parcel p was constrained to be binary, [equation 10.1]). For each county, we minimized the difference between the total area converted from LULC class j to k and the total area specified by the *BT16* scenario for the county (equation 10.1).

Equation 10.1:

Let

$t_{k'\to k} \equiv$ probability of transition from class k' to k

$$x_{pk} \equiv \begin{cases} 1 \text{ if parcel } p \text{ is in class } k \text{ in future year 2040} \\ 0 \text{ otherwise} \end{cases}$$

$x^0_{pk} \equiv$ initial value for year 2014

$L_{k'} \equiv$ the set of LULC classes to which a parcel initially assigned to k' can transition, $\{k \ni t_{k'\to k} > 0\}$

(for a given parcel p, x_{pk} is defined only if $k \in L_{k'}$ where $x^0_{pk'} = 1$)

$a_p \equiv$ area of parcel p

$A_k \equiv$ total area initially assigned to class $k = \sum_p a_p x^0_{pk}$

$T_k \equiv$ target future area assigned to class $k = \sum_{k'} A_{k'} t_{k'\to k}$

$y_k \equiv$ deficit of class k

$z_k \equiv$ surplus of class k

$$\begin{array}{ll}
\text{Choose } x_k, y_k, z_k \text{ to minimize } \sum_k (y_k + z_k) \text{ subject to} & \\
\sum_k x_{pk} = 1 & \forall p \\
\sum_p a_p\, x_{pk} + y_k - z_k = T_k & \forall k \quad \text{(total area constraint)} \\
y_k, z_k \ge 0 & \forall k \\
x_{pk} \in \{0,1\} & \forall p,k
\end{array}$$

We assigned parcels by solving the mixed-integer linear programming model above using the CPLEX[2] solvers for each county. For a given county, the input data include information about each parcel, *p*, including its area (a_p) in hectares and its initial LULC class (k' s.t. $x^0_{pk} = 1$). We can calculate the total area assigned to an LULC class *k* as shown above. We are also given a transition matrix based on POLYSYS results specifying the probabilities ($t_{k'k}$) of a parcel transitioning from any particular LULC class *k*' in 2014 to another LULC class *k*. These probabilities are used to generate the expected area, T_k, assigned to an LULC class k in the future. The goal of the optimization problem is to generate a set of assignments of parcels to LULC classes (x_{pk}) such that each parcel is assigned to exactly one LULC class, and the total area of parcels assigned to a LULC class match the total-area target as closely as possible. Ideally, we would like $\sum_k A_k t_{k' \to k} = T_k$. The initial LULC class of a parcel (*k*') limits the set of possible future LULC classes to those with a positive transition probability.

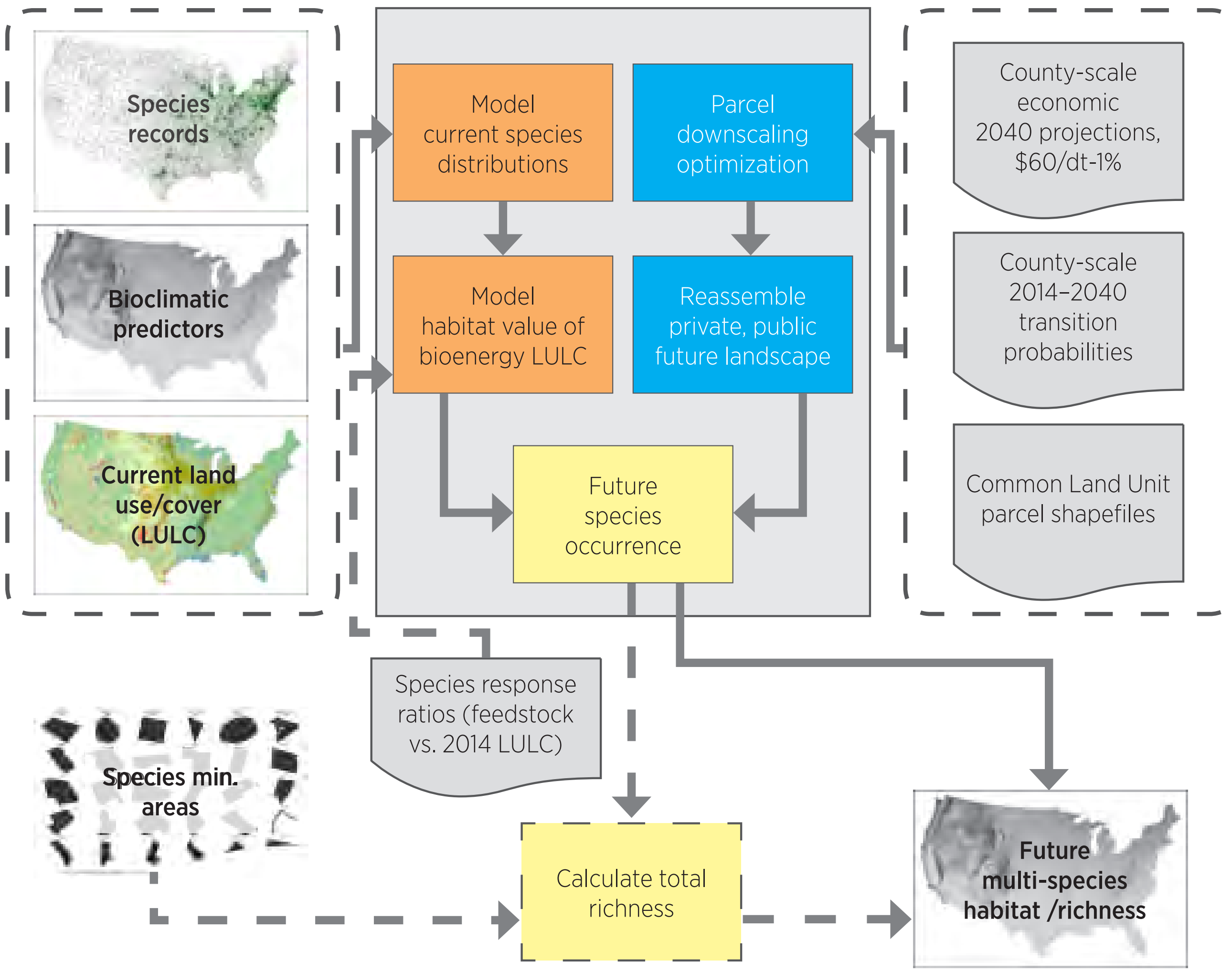
A complete future 2040 landscape was produced by overlaying the CLU parcels on the unchanged 2014 map. This was accomplished by joining parcel shapefiles with downscaled LULC in the attribute table using the "add join" tool in Arc-GIS©. The resulting layer was then converted into a raster file using the "polygon to raster" tool and merged with the original CDL-2014 map.

10.3.2 Overview

Our estimation of species richness in projected landscapes follows the process illustrated below (fig. 10.1). SDMs for each species were developed from occurrence data and landscape predictors for the year 2014 (fig. 10.1, far left). Three alternative approaches were used to project future occurrences (see section 10.3.5; fig. 10.1, middle). For each species, the result was a map projecting future likelihood of occurrence (fig. 10.1, middle) for year 2040 (fig. 10.1, far right).

[2] IBM ILOG CPLEX Optimization Studio software.

Figure 10.1 | Framework used to evaluate how bird richness might change under a future bioenergy scenario.



Our approach to estimating the habitat value of LULC that produce biomass crops depends on quantitative studies comparing habitat value of the biomass LULC and other classes. Because habitat comparisons that are required to estimate habitat value are not available for all combinations of LULC and dedicated energy crops, we present results only for bird species for which we have comparisons. We have two groups of species: (1) a set of predominantly grassland species for which comparisons of habitat value of switchgrass versus grassland were available, and (2) a set of predominantly forest and generalist species for which comparisons of bird response to SRWCs versus forest were available. These were modeled separately, with effects of changes in the geographic distribution of sorghum included for both. The approach is described in section 10.3.5.

Our primary goal in building models was to estimate the current habitat value of each parcel for different species in current and future landscapes. We developed an index of change in bird richness by summing probabilities of occupancy across species at either the grid cell or county scale.

10.3.3 Species Distribution Modeling

We used boosted-regression tree methods (Elith, Leathwick, and Hastie 2008) to develop species distribution models (SDMs) that included LULC classes and bioclimatic variables, including elevation, as predictors (fig. 10.1). The outputs of the SDMs were local estimates of the probability of occupancy by a species within 1-km raster pixels for the conterminous United States.

SDMs require data on the presences and absences of a species across a landscape to estimate the relative likelihood of occurrence in a particular location (Guillera-Arroita et al. 2015). We collected presence records for selected species. These spatially referenced biodiversity data were derived from point locations reported in the Biodiversity Information Serving Our Nation database (USGS 2013). A series of steps were required to modify these data so that the records would be useful inputs to the SDMs. First, we excluded fossil records, records without a known type, and records dated prior to 1990. Second, we accounted for the use of presence-only data, which are generally not systematically sampled and lack any substantive information about species' absences (Hertzog, Besnard, and Jay-Robert 2014). We controlled for this potential sampling bias by generating pseudo-absences from locations where similar species have been reported ("target-group background sampling" per Phillips et al. (2009)). Each SDM was based on an approximately equal number of presences and randomly sampled pseudo-absences from the target group (Barbet-Massin et al. 2012).

We used a machine-learning algorithm [boosted regression trees, (Elith, Leathwick, and Hastie 2008)] to estimate the geographic distribution of habitat suitable for each bird species based on a consistent set of bioclimatic variables and LULC data at a 30 arc-second resolution (~1 km^2) throughout the conterminous United States (Hijmans and Graham 2006). These layers were transformed from raw temperature and rainfall inputs between 1950 and 2000 to generate long-term climate measures, which are considered biologically meaningful as predictors in species distribution modeling (Booth et al. 2014). In addition to these bioclimatic variables, we included LULC class from the 2014 CDL and US Environmental Protection Agency (EPA) Tier II ecoregions as categorical variables.

We split all records into 70% training and 30% test sets, and we assessed our SDMs by their out-of-sample prediction accuracy on the test set. We excluded models not significantly more accurate than the no-information rate, or 50%, for this binary classification. All SDMs were formulated as boosted regression trees. This method produced models demonstrating a high level of accuracy here and in previous studies (Elith, Leathwick, and Hastie 2008). All SDMs were built with the "caret" package, which is used to conduct training of classification and regression tree models (Kuhn 2008) in the R computing environment (R Core Team 2014). Our SDMs predicted the probability of occurrence of a species and could also predict a binary presence (occupancy) or absence for each 1-km^2 grid cell across the conterminous United States. Summing these estimates produces an index of bird richness for each grid cell.

10.3.4 Modeling Occupancy in Extant LULC Classes

For each species, we used the SDM developed above to predict probabilities of occurrence, $P[s \mid \mathbf{x}, \mathbf{L}]$, based on spatial bioclimatic predictors and LULC at each location $\mathbf{x}$. The resulting spatial field of probabilities is used to estimate each parcel's habitat value, $P[s \mid \mathbf{x}, \mathbf{L}]$.

Our goal is to estimate the effect of each extant LULC class, k, on the probability of occurrence (or habitat value) for each species s as a function of average bioclimatic conditions in parcel p. In future landscapes, some downscaled parcels will transition to a new LULC, L_k. Let vector $\mathbf{L}$ contain: (1) $k = 1$ to r classes that are well represented in the extant U.S.

landscape and $k \geq r$ perennial grasses classes grown as feedstocks for bioenergy, and (2) $k > t$ ($t > r$) represent woody crops that are not currently well represented but that occur in future *BT16* landscapes.

Different approaches were required for LULC transitions to LULC that are well represented in the current landscape (e.g., sorghum) and for those not currently widespread but expected to increase in the future (e.g., switchgrass, miscanthus, SRWC; see fig. 10.1). We developed approaches for each of three types of LULC conversions, conversions to sorghum, conversions to perennial grasses, and conversions to SRWC):

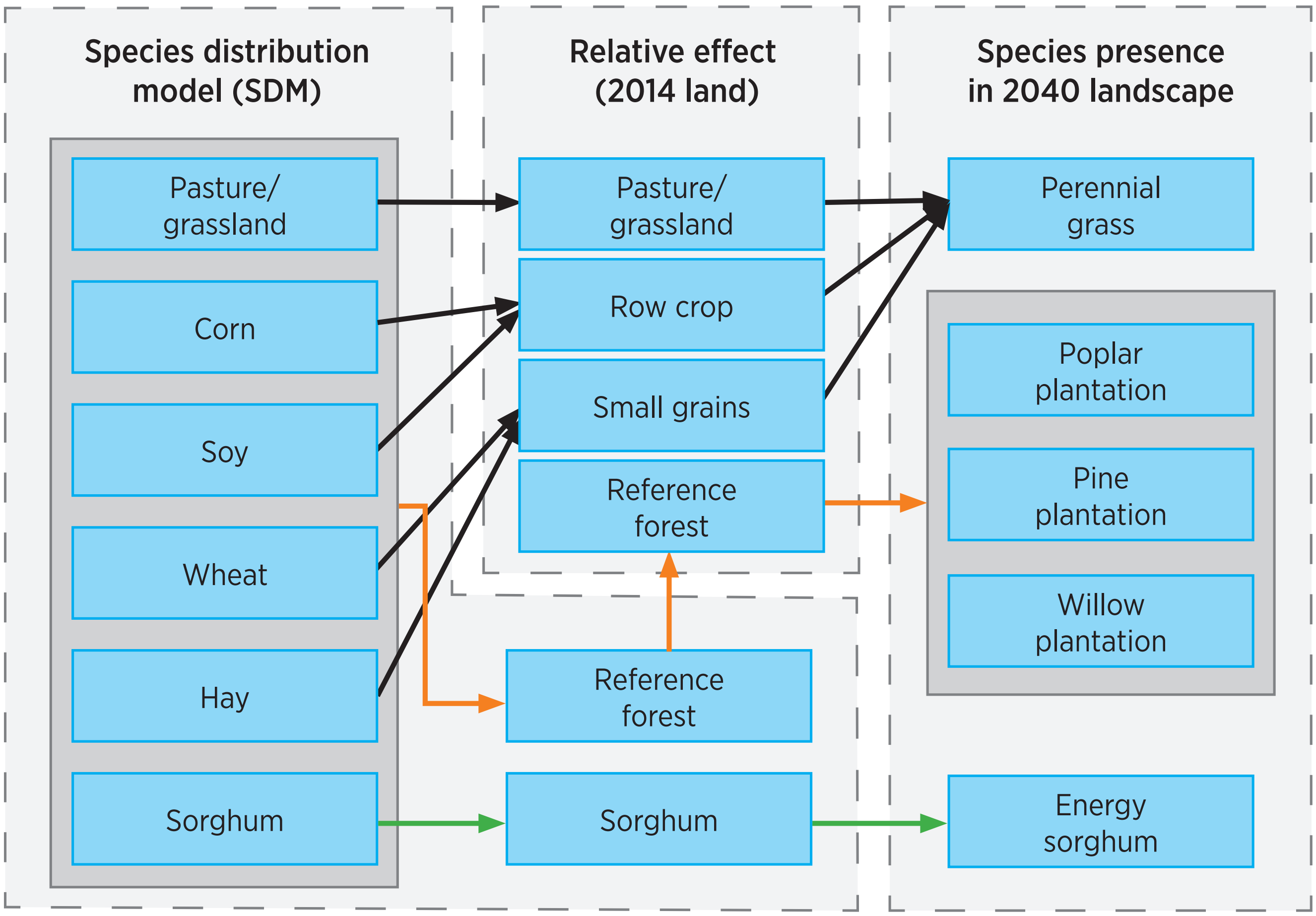
1. $k \leq r$: L_k was sufficiently well represented in the extant 2014 landscape. In this case, we could use the SDM value to estimate $P[s \mid \mathbf{x}, L_k]$. In our analysis, both sorghum and energy cane were estimated by using the SDM for sorghum.
2. $r < k \leq t$: L_k is a perennial grass that is projected to be used as a feedstock in the future, but its habitat value cannot be estimated from the current SDM.
3. $k > t$: L_k is an SRWC that is projected to be used as a feedstock in the future, but its habitat value cannot be estimated from the current SDM. Literature compares species' performance in SRWC with that in natural forest types, but not agricultural LULC.

10.3.5 Modeling Occupancy in Biomass Crops as LULC Classes

The transitions considered in our analysis are illustrated in figure 10.2. For parcels of biomass-producing LULC classes that are not currently widespread, we developed a new method for estimating habitat value. We conducted a literature review of species to find studies that compare bird densities in different LULC categories (including lands managed to produce dedicated bioenergy feedstocks). For the LULC class growing dedicated biomass crops, we compared densities under different harvest-management practices. Two meta-analyses of such studies calculated and reported response ratios in a consistent manner, reflecting the ratio of bird densities (Riffell et al. 2011, Robertson et al. 2012). However, comparative data were not available for all transitions for all species. For transitions that we were unable to model, affected CLU parcels were excluded from comparisons. We separately report results for three groups of bird species and types of LULC change from non-biomass to biomass crop: (1) predominantly grassland bird species in perennial grasses, and (2) predominantly forest birds in SRWC, and (3) generalist birds in SRWC. Birds with generalized habitat preferences are those that either prefer forest edge, those that occur both in grasslands and forest or in savannah.

In the following sections, we describe modeling pathways (fig. 10.2) for estimating occupancy in landscapes, including (1) sorghum and energy cane, (2) SRWC, and (3) perennial grasses.

Figure 10.2 | Inference for future landscape scenarios is based on literature values of relative effects of land use classes on individual species. Note that different processes are required to model transitions to switchgrass (black arrows), SRWC (orange arrows) than to sorghum or energy cane (green arrows).



10.3.5.1 General Model

Let $D^s_{(k,h)}$ denote the density of a species, *s*, in LULC class *j*. The response ratio, *RR*, of species *s*, for two LULC classes, one currently prevalent in the landscape ($i \leq r$) and one future biomass LULC ($j > r$), is given by equation 10.2, with a constant, $\delta = 0.001$, added to avoid dividing by zero in the case of zero density in the 2014 LULC.

Equation 10.2:

$$RR^s_{(i,j)} = \left(\frac{D^s_{(j)} + \delta}{D^s_{(i)} + \delta} \right), \quad i \leq r; j > r$$

We modeled the relationship between the probability of occurrence, *P*, in the old, *i*, and new, *j*, LULC using equation 10.3. This form is motivated by models that separate the observational process and represent the probability of detection as a function of abundance (Royle and Nichols 2003). The probability of detecting at least one animal, given that animals are present, is equal to one minus the probability of not detecting all animals at the site. Here, we modeled the change in detection probability by treating the individual units as groups of organisms equivalent in number to those in the original LULC.

Equation 10.3:

$$P[s \mid \mathbf{x}, L_j] = 1 - \left(1 - P[s \mid \mathbf{x}, L_i]\right)^{RR^{s}_{(i,j)}} \qquad i \le r;\ j > r$$

10.3.5.2 Sorghum and Energy Cane

We assumed that energy cane, energy sorghum, and sorghum grown for food had similar habitat value. Historical use of sorghum as habitat was modeled by the SDM. Therefore, we estimated future occurrence of birds in sorghum and energy cane directly (fig. 10.2).

10.3.5.3 SRWCs

Projecting bird occupancy in future SRWC plantations on agricultural lands required a two-step process (fig. 10.2). We estimated the habitat value of the locally prevalent forest type (fig. 10.3) before applying the forest-to-SRWC plantation response ratio (fig. 10.4). Note, this is simply an accounting trick because many studies have compared bird densities in managed LULC to densities in forest, but none have compared densities in different managed LULC including one managed for biomass crops. *In other words, transition of forest lands to SRWC was not simulated in the BC1 2040 scenario*. We obtained SDM predictions of occupancy probabilities, $P[s|\mathbf{x}, L_j]$, by creating a transitional national LULC map, where the grid cells in the baseline LULC map with SWRCs under future scenarios were substituted by locally prevalent forest types. Next, we applied the conversion from equation 10.3 using the appropriate response ratios reported for 40 birds found in forest or open woodland and edge habitat (Riffell et al. 2011).

Figure 10.3 | Dominant forest type for 1-km pixels of the conterminous United States. This information is needed to implement a two-stage estimation process of bird probability of occupancy in short-rotation woody crop plantations.

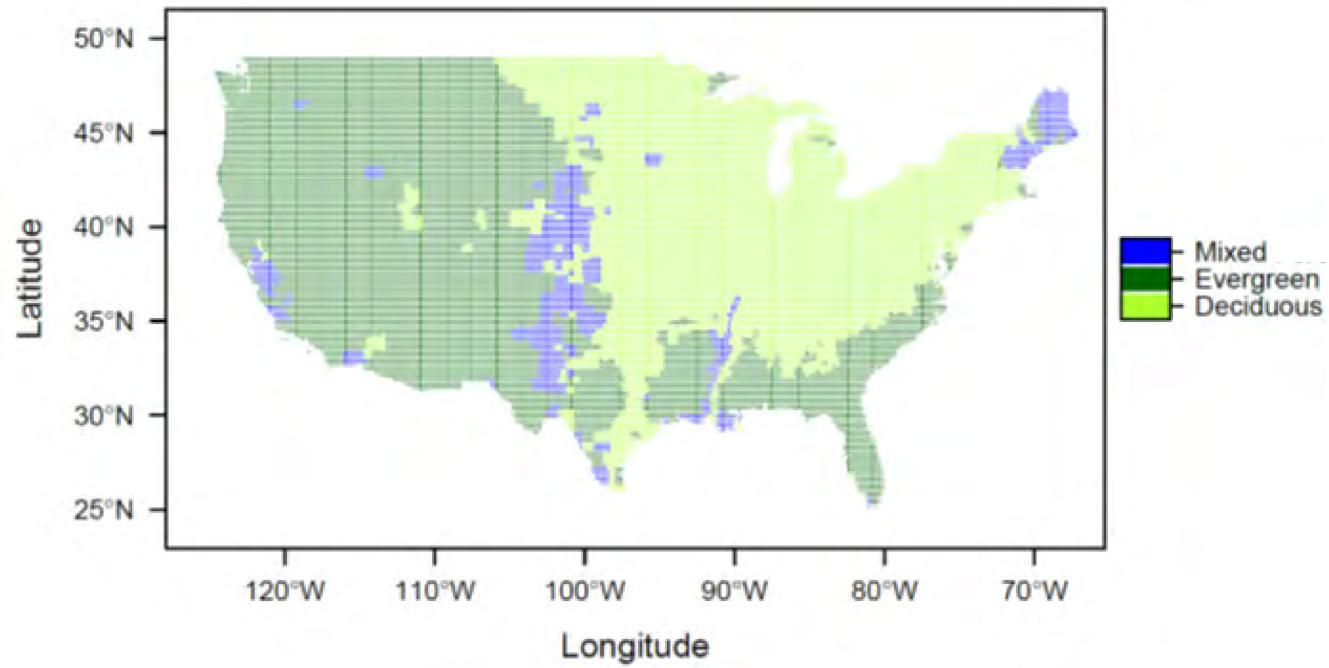
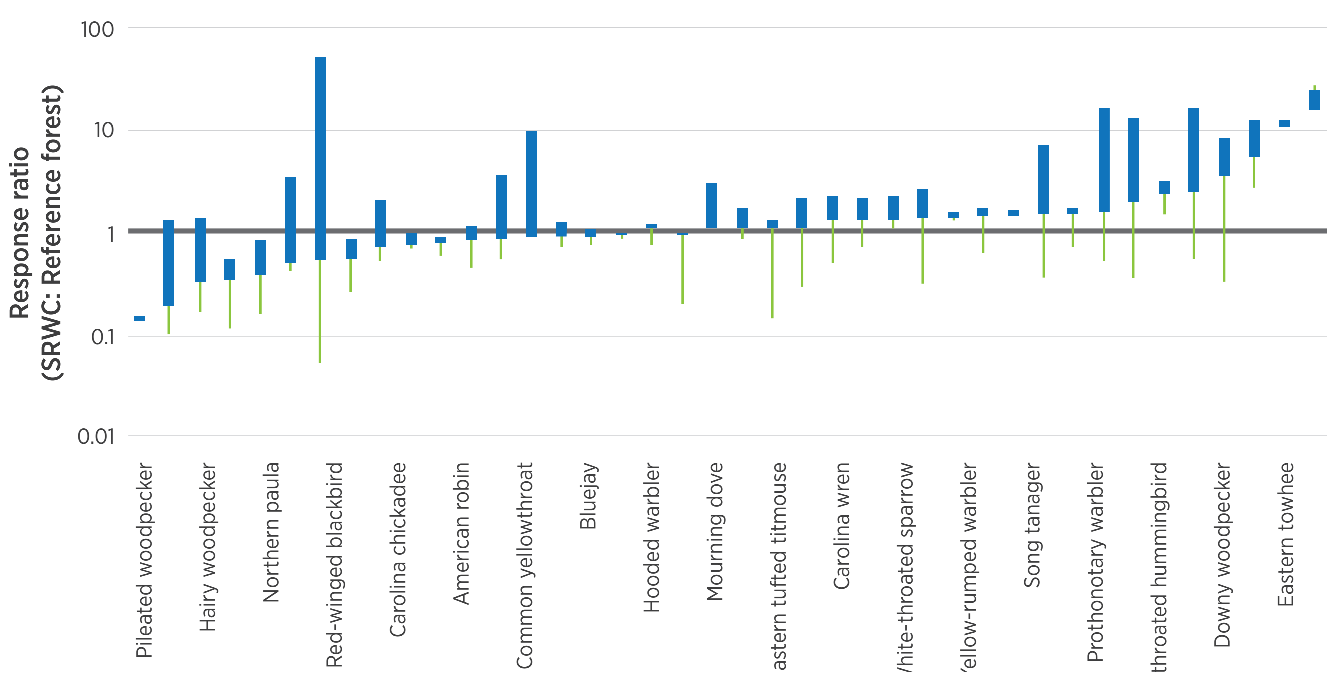


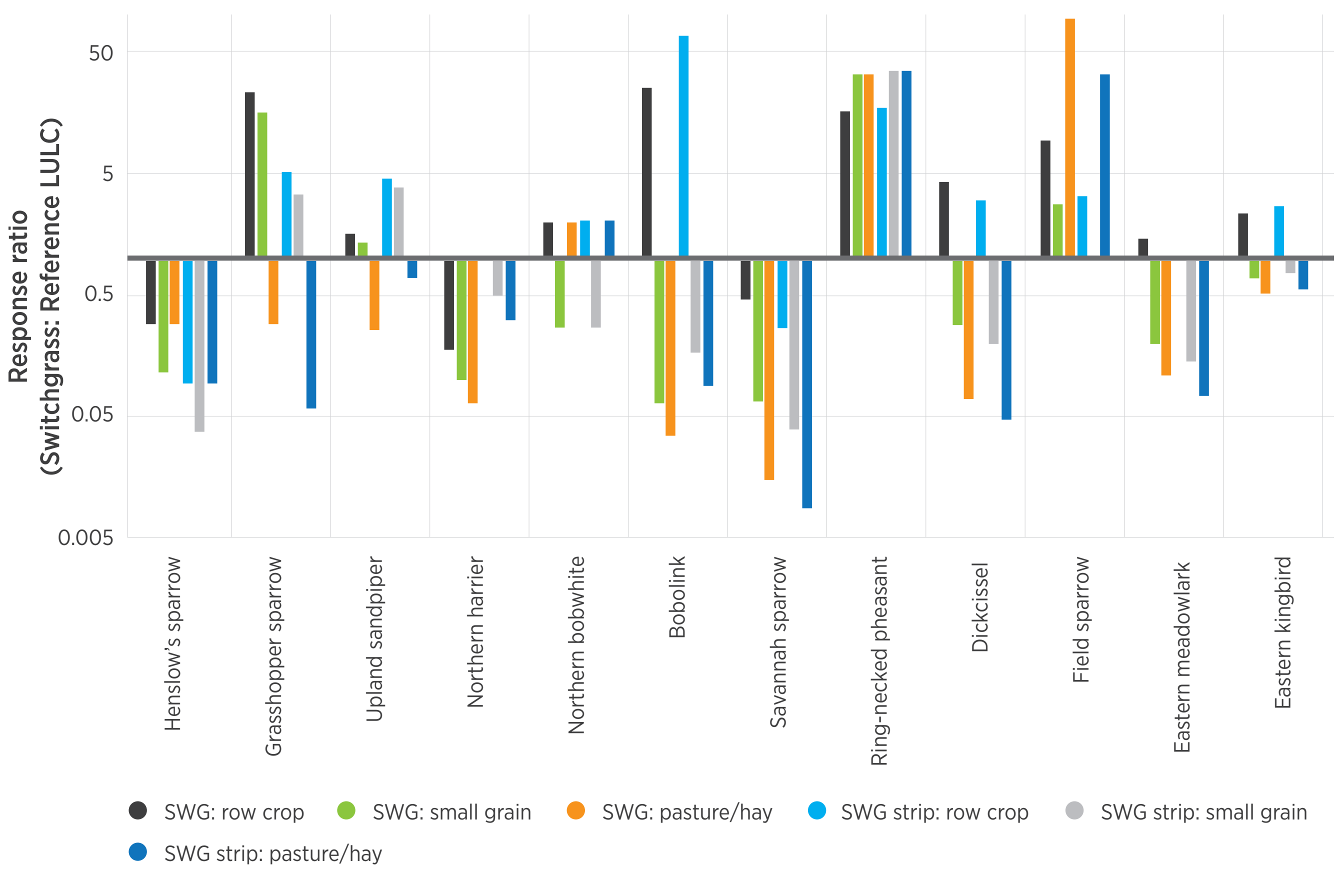
Figure 10.4 | Response ratios for forest and generalist birds to local reference forest used in two-step process.



10.3.5.4 Perennial Grasses (Switchgrass and Miscanthus)

For switchgrass, we used estimated response ratios summarized by Robertson et al. (2012) based on data collected from Fletcher et al. (2011) for 12 grassland bird species (fig. 10.5). Comparisons allowing us to model transitions to switchgrass were available for three classes of agricultural LULC: (1) pasture/grassland and hay; (2) row crops (as defined by Robertson et al. [2012]) including corn, cotton, and soybean; and (3) small grains, including barley, sorghum, rice, oats, and wheat (fig. 10.2). To account for harvest management for switchgrass, we multiplied by an additional management response ratio, i.e., the ratio of bird density in switchgrass fields harvested in a certain way to its density in unharvested switchgrass (fig. 10.5). Bird densities were reported for switchgrass fields with strip harvest and total harvest (Best and Murray 2003). Thus, for switchgrass, we have comparable densities for bird species in three extant LULC classes (small grains, pasture, and row crops) and in switchgrass managed in each of two ways.

Figure 10.5 | Response ratios for grassland birds in total- and strip-harvested switchgrass (SWG).



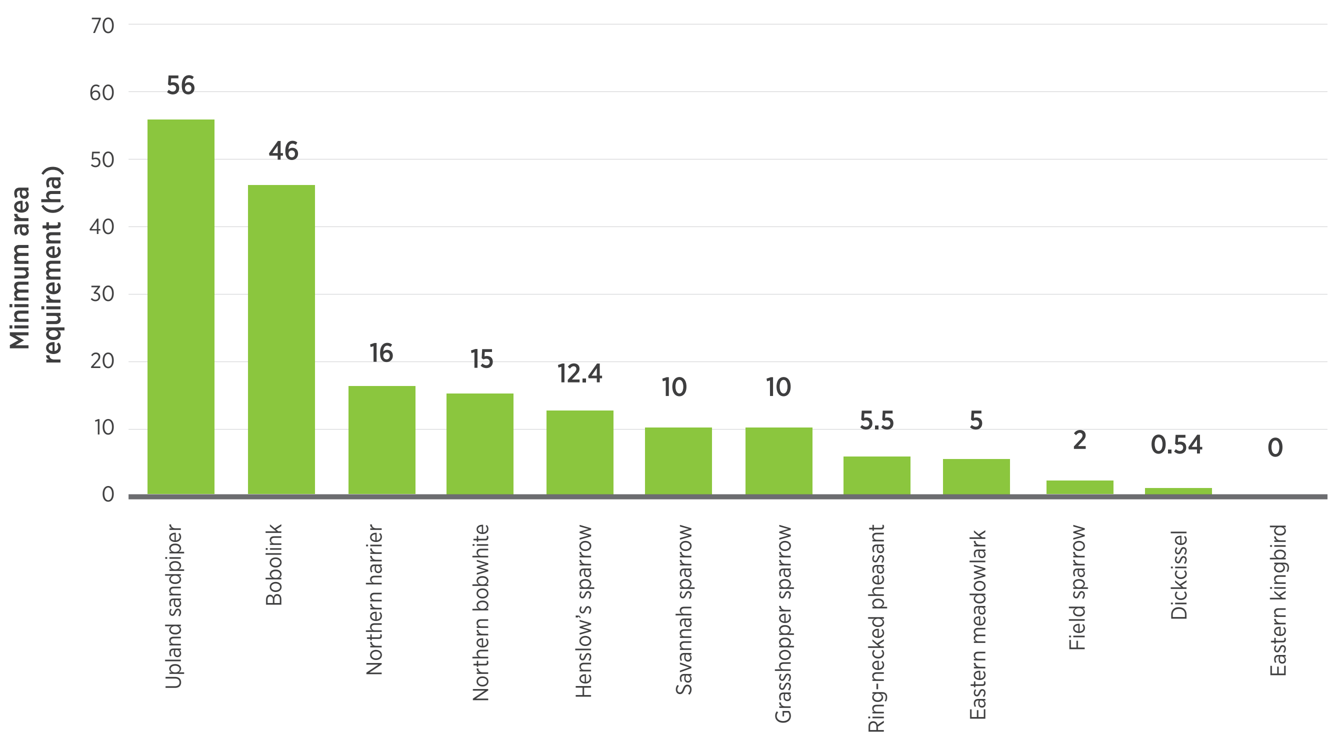
For miscanthus, we adopted a 'precautionary' approach. Published studies related to the suitability of miscanthus as a habitat are not yet available for birds in the United States (Vandever and Allen 2015). At this point, there is no evidence that miscanthus is used as nesting habitat for songbirds[4] in the Midwest, and songbird densities in miscanthus were much lower than densities in surrounding grasslands.[5] Therefore, we assumed that parcels that transitioned to miscanthus had zero habitat value.

10.3.6 Accounting for Minimum Area Requirements

A subset of species with specialized habitat needs require a minimum area of habitat to persist in habitat patches. Bio-EST can account for such area thresholds. By overlaying the patch raster and the habitat SDM and removing habitat areas associated with small patches, we accounted for area sensitivities of birds. Therefore, CLU parcels that might otherwise have had a positive probability of occupancy (habitat value) were considered unoccupied if the total area of the patch and surrounding lands (including public lands), was too small to support the species. The "raster" package in R was used to define habitat patches and to calculate patch sizes.

As a test case, we compared the estimated number of occupied grid cells for 2014 and the future map consistent with BC1 2040 for grassland birds with estimates of minimum area requirements >0 (appendix A, table 1; fig. 10.6).

Figure 10.6 | Minimum habitat area requirements for selected bird species



[4] Songbirds are in the order *Passeriniform* (i.e., 'perching birds'), which includes most grassland birds considered here. Non-passerine species considered here include the upland sandpiper and ring-necked pheasant.

[5] R. L. Schooley, University of Illinois, email to H. Jager, June 23, 2016.

10.3.7 Projecting Changes in Richness

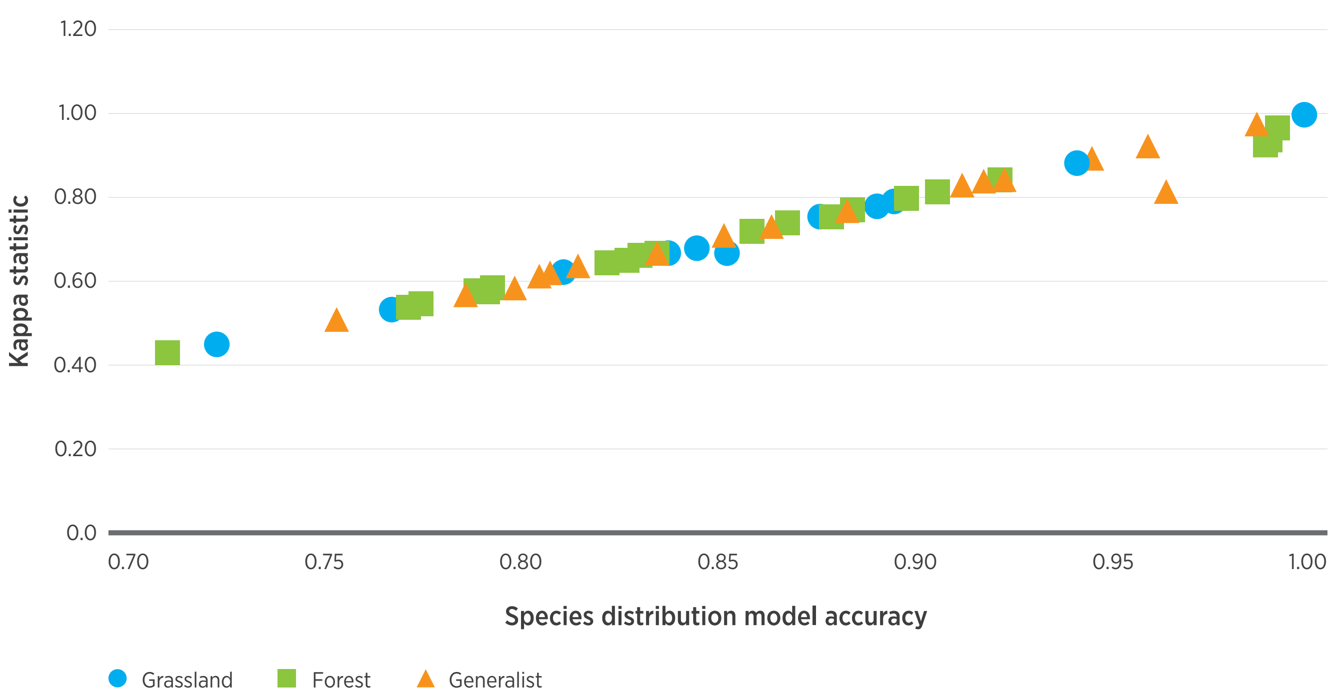
The methodology above (equations 10.2–10.3) allowed us to generate raster maps quantifying the likelihood of occurrence for each species. Richness maps result from aggregating predicted occurrences for three groups of species: a set of 12 predominantly grassland species and two sets of "forest" species for which we have data describing transitions to SRWC (see appendix A), referred to as forest specialists and generalists. For grid cells, we added the occupancy probabilities across the map to estimate the number of occupied 1-km^2 grid cells. For counties, we estimated the number of counties occupied in 2014 versus 2040 (BC1 2040), the change in the estimated number of occupied counties, and changes in richness. Analyses are reported separately for the grassland, forest specialist, and forest generalist species. Data were available to model LULC transitions to perennial grasses and energy sorghum (not SRWCs) for birds in the grassland group. Data were also available to model LULC transitions to SRWCs and energy sorghum (but not perennial grasses) for birds in the forest specialist and generalist groups. We recognize that there is some subjectivity in how these sets are defined.

10.4 Results

10.4.1 Species Distribution Modeling

Overall, the performance of SDMs was excellent. For the testing set, accuracy varied from 0.71 to 1.0 (all p-value <0.0001) across the 52 bird species modeled (fig. 10.7). Kappa statistics on the same set varied from 0.42 to 1.0 (all p-value <0.0001), with 79% of the kappa statistics (i.e., 41 out of 52 species) exceeding 0.6. Kappa values above 0.6 demonstrate substantial strength of agreement (Landis and Koch 1977).

Figure 10.7 | Minimum habitat area requirements for selected bird species



10.4.2 Minimum Habitat Area

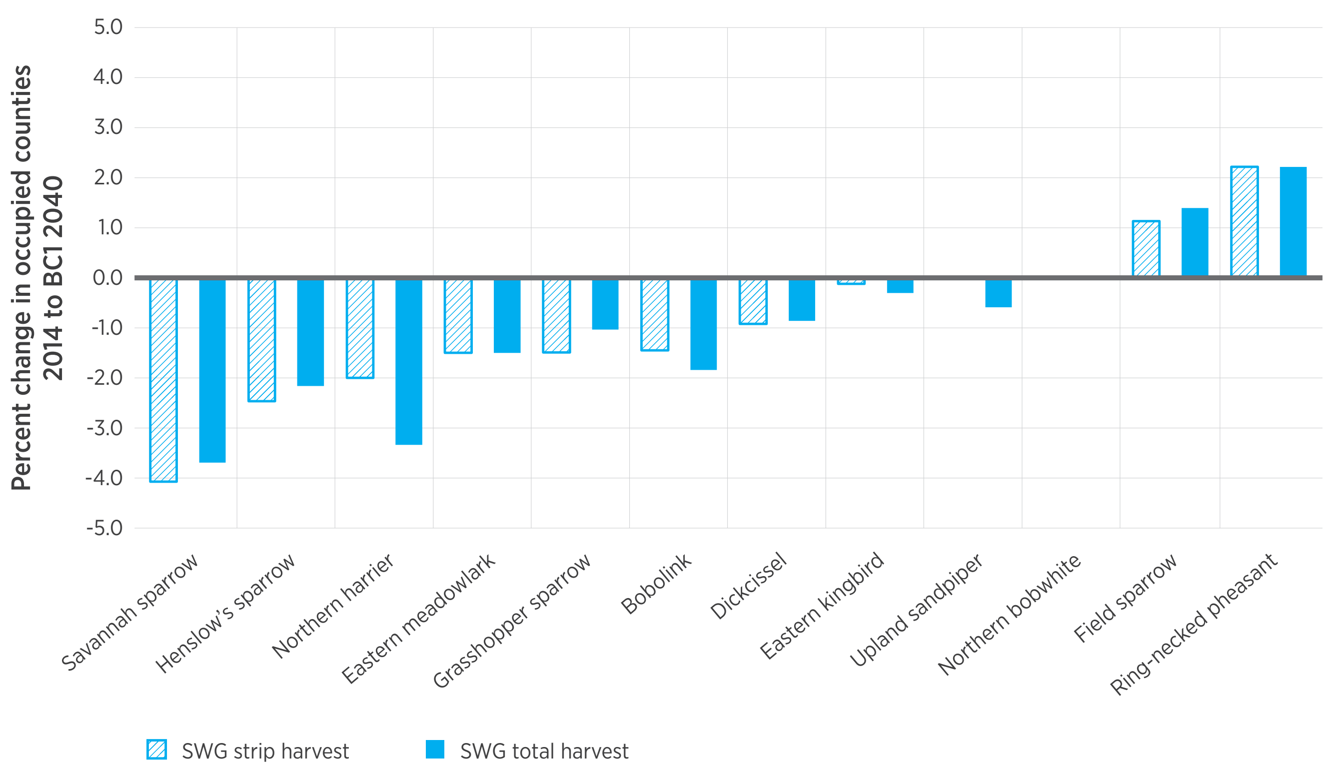
In an exploratory analysis, we removed small patches of habitat below the minimum habitat threshold of each grassland species. Because the effects were applied to future occupancy maps of both the reference 2014 case and BC1 2040 scenario, the resulting differences in range, measured in the number of counties occupied, were small and very similar for the 2014 and future landscapes (average 2.85% [SD = 0.68%] difference for 2014 map, 2.88% [SD = 0.62%] for BC1 2040 with strip harvest). Therefore, results presented here do not consider minimum habitat area.

10.4.3 Projected Changes in Richness under BC1 2040 Scenario

Our simulations excluding miscanthus showed no change in projected occupancy from the 2014 to the BC1 2040 landscape for most 1×1-km grid cells (>98% for both groups). However, in addition to lack of response to LULC change, this result is partly because we did not have information to simulate all possible transitions and partly because non-private lands were not permitted to change LULC. Decreases were projected in 0.13% of grid cells for grassland species, 1.4% for forest specialists, and 0.36% for generalists. Increases were projected in 1% of grid cells for grassland species, 0.07% for specialists, and 1.13% for generalists (fig 10.8).

Geographic patterns in grassland species reflect responses to management of agricultural lands to BC1 2040 future switchgrass (strip harvest) and energy sorghum (fig. 10.9, top row). Patterns for forest Projected decreases appear to be concentrated in the middle of the country (fig. 10.9).

Figure 10.8 | Change in the estimated percentage of counties occupied by grassland bird species between the 2014 landscape and a future landscape consistent with the BC1 2040 scenario. Results are shown for two management regimes include strip harvest and total harvest of switchgrass (SWG).



For forest birds (specialist and generalist species), no change in richness was estimated for 99.2% of grid cells between the 2014 and the BC1 2040 LULC.. Increases occurred in <1% of the grid cells for both forest generalists and specialists. Likewise, decreases occurred in <1% for both types. (fig. 10.10).

Figure 10.9 | Change in projected richness under the 2014 landscape (left column), a landscape consistent with the BC1 2040 future scenario (middle column) and differences (right column) for three groups of species. Rows display distributions for grassland, generalist, and forest specialist species. The range for differences in richness displayed by the legend row (below headers) is indicated below each map.

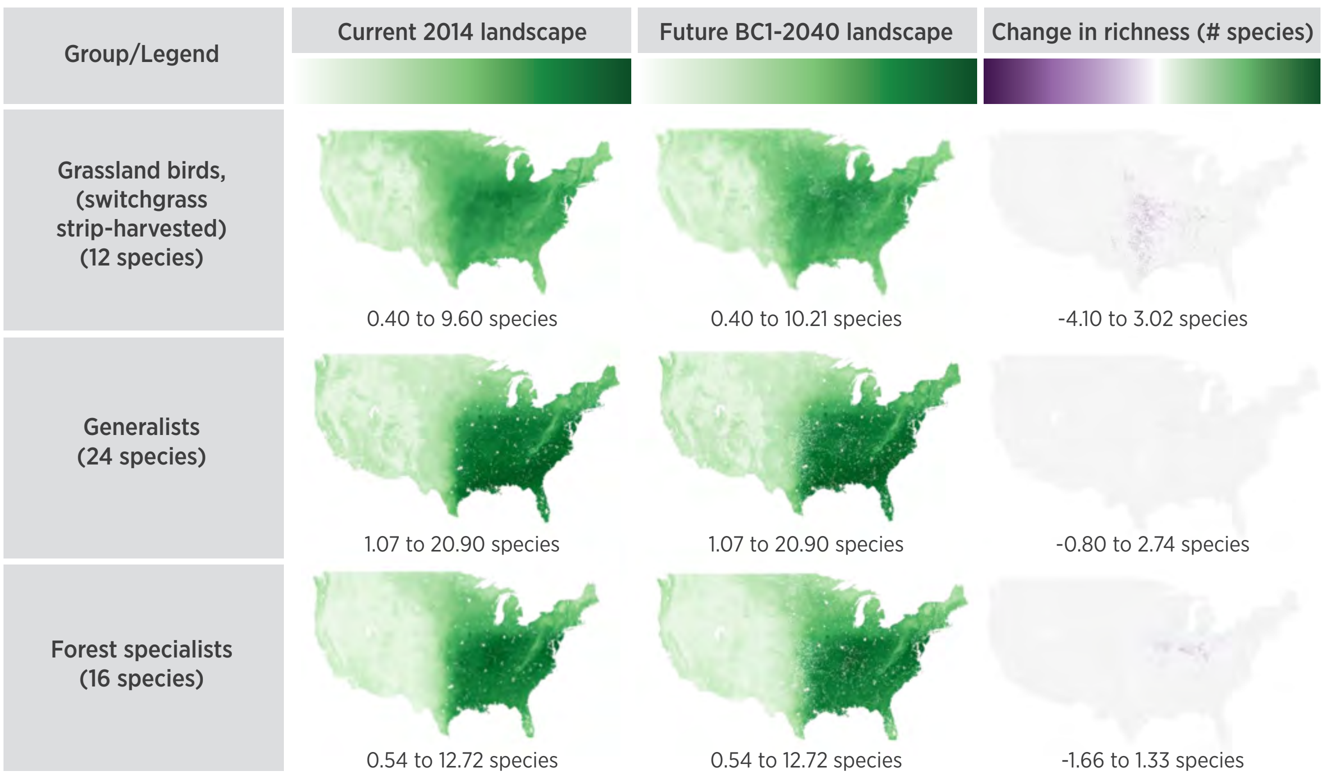
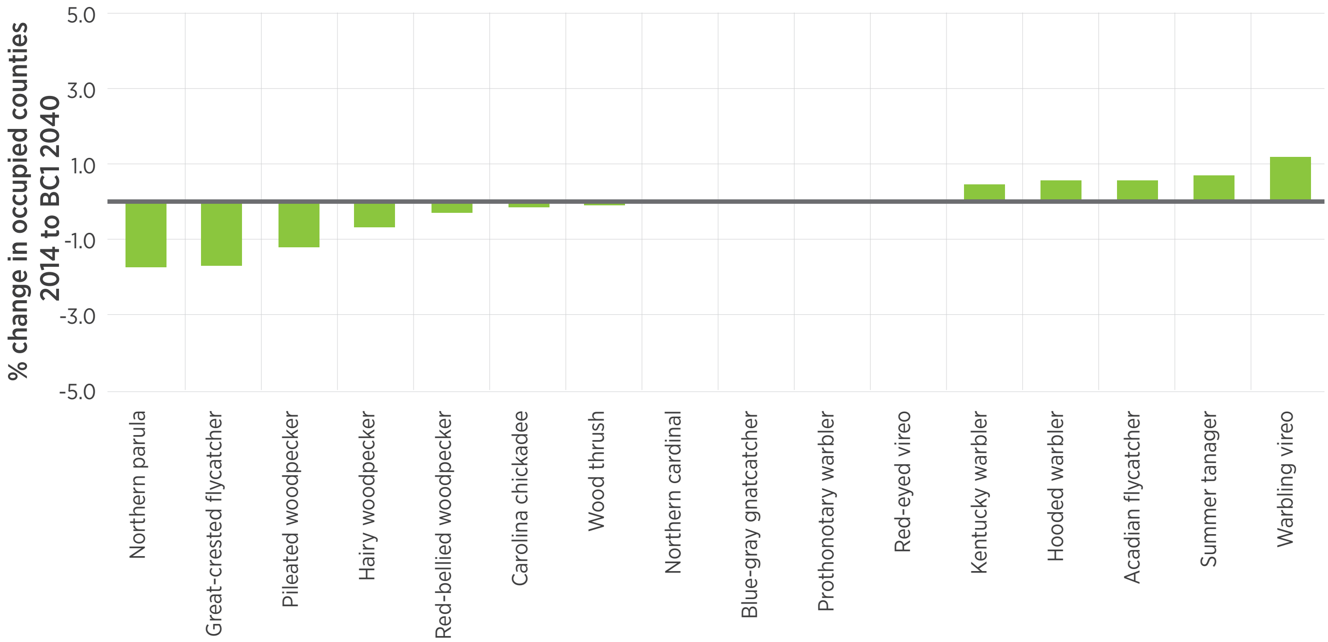
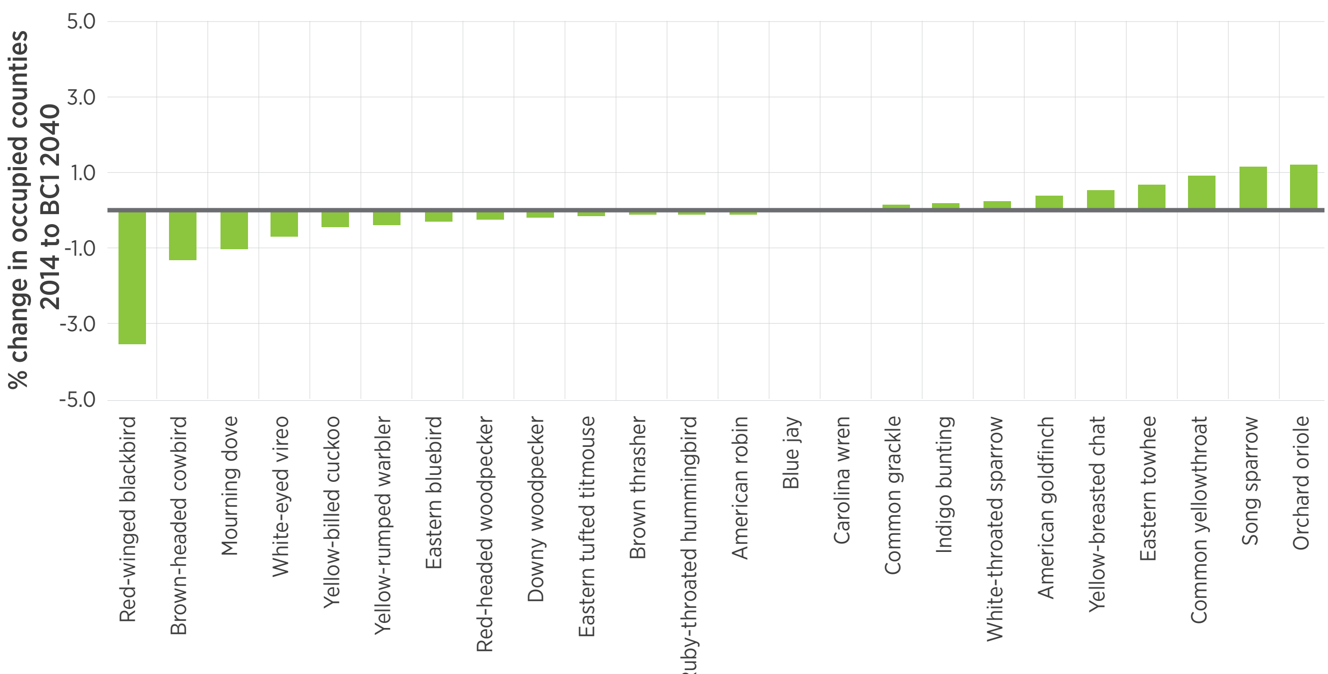


Figure 10.10 | Change in the modeled percentage of counties occupied by species designated for purposes of this analysis as a) forest specialist and b) generalist bird species between the 2014 landscape and a future landscape consistent with the BC1 2040 scenario.





To understand the LULC changes driving these results, we summarized LULC changes that would result in changes of more than 80% in richness at the grid-cell scale, all of which were planted in switchgrass in the BC1 2040 landscape. Positive changes in grassland bird richness were dominated by grid cells that were planted in cotton or corn in 2014 (1,138 grid cells), whereas negative changes were dominated by grid cells planted in pasture or hay in 2014 (14,777 grid cells). Grid cells with positive changes in generalist bird species were planted in corn (19 grid cells) or wheat (11 grid cells) in the 2014 landscape and non-coppice wood (poplar) in the BC1 2040 landscape. Grid cells that decreased in richness were in coppice wood (willow) in the BC1 2040 landscape and pasture (70 grid cells) or soybeans (20 grid cells) in the 2014 landscape. Grid cells associated with negative changes in the number of forest bird specialists were predominantly in coppice wood (willow) in the BC1 2040 landscape and in soybeans (3,057 cells) or corn (144 cells) in 2014.

10.5 Discussion

Results presented here for grassland and woodland/forest birds in the BC1 2040 scenario are consistent with our expectations about the potential costs and/or benefits of growing dedicated bioenergy crops. Among grassland birds, projections showed the potential for increases in range for ring-necked pheasant and field sparrow, and decreases (or no change) for others. It is important to note that our assumptions about miscanthus were precautionary (we assumed zero habitat value for this crop, which represented 77,821 km^2 in the BC1 2040 landscape). Interestingly, strip harvest did not consistently increase occupancy across grassland species compared with total harvest. It should be noted that grassland-obligate species are better served by patches of habitat with high area-to-perimeter ratios, i.e., blocks, not strips (Helzer and Jelinski 1999; Roth et al. 2005).

Further analysis to understand how different taxa responded could help to explain the risk or benefits to species with different life histories and habitat needs. The analysis presented here can also be extended to represent other wildlife taxa once enough comparisons of wildlife performance (e.g., density, reproductive success) in multiple food crop and biomass crop habitats have been made. For example, studies have quantified the benefits of energy crops as a habitat for pollinators (Meehan et al. 2012; Bennett et al. 2014; Bennett and Isaacs 2014) and for other beneficial insects, for example, those that provide pest-control services (Werling et al. 2011). In comparison to birds, few studies have focused on quantifying the habitat value of biomass crops for other taxa (e.g., mammals, amphibians, and reptiles).

Our analysis involved some simplifying assumptions to allow for a national-scale assessment. It uses an implicit "equilibrium" assumption. In other words, we compare a recent landscape with one potential future landscape, but not with transient influences of the crop transition on occupancy, which could incur higher, but possibly temporary, impacts. The timing of management changes might help to alleviate potential stresses caused by change. Birds are mobile taxa that could be more resilient to changes in land management than other taxa, except during mating, nesting, and incubation.

Finally, we join other ecologists by offering the suggestion that benefits to birds (and other wildlife) can be attained by implementing wildlife-friendly practices (Meehan, Hulbert, and Gratton 2010; Robertson et al. 2012; Ridley et al. 2013). Birds tend to be at their most vulnerable to disturbance by management activities during nesting. Impacts to nests can be avoided by timing farm operations prior to the summer nesting season and between harvest and the summer nesting season. Timing harvest to occur outside of the nesting season is more feasible for grasses grown for biomass than for hay and other crops that quickly lose their quality as forage for animals if harvested in the fall. Furthermore, potential for harvest after winter can be explored to provide resident birds with cover and forage during winter. In addition, using a flushing bar and raising the height of mowing equipment can help to avoid nests and animals during farm operations; and, simply harvesting from the inside out, instead of trapping wildlife in the center of a field, can be beneficial. These, and other best-management practices can help to manage bioenergy crops with an eye toward protecting biodiversity (McGuire and Rupp 2013; Brooke et al. 2009).

10.6 Future Directions

Future research can address ways to design biomass-production methods that benefit biodiversity, as well as producing feedstocks for bioenergy or other uses:

- Research is required to increase the feasibility of production systems that employ more diverse communities of plants as feedstocks, including forbs and other plants typically found in native prairie. Such plant communities have been found to support more diverse communities of insects and possibly other taxa. In conjunction with research on diverse feedstock production, research is needed to understand barriers to the conversion of complex cellulosic feedstock streams.
- This assessment relied on field comparisons of wildlife in other crops or LULC classes and biomass-producing lands. These data are needed to quantify the responses to bioenergy crops by other taxa. In particular, information about potential habitat value of miscanthus and eucalyptus is lacking. These non-native species may or may not provide similar habitat to pre-existing native vegetation.
- Research is needed to understand logistic and economic barriers that could prevent farmers from adopting practices that benefit wildlife. Some of these barriers might be overcome by developing innovative technologies (smart tractor systems) and new wildlife-friendly practices.
- The relative effects of pesticide use for bioenergy feedstocks and for other managed lands, as well as trade-offs between pesticide use and other potentially beneficial practices (e.g., tillage), have not been studied and quantified or related to wildlife performance.
- Benefits of seed-producing crops to wildlife are well known (Guthery 1997). However, the wildlife and production co-benefits of integrating production of biodiesel crops, such as soybeans and canola, with cellulosic feedstock production have not been explored.
- Future research can help to identify geographic hotspots where attention to wildlife-friendly practices is needed. In addition, trait-based guidance can be developed to guide farmers and SRWC growers toward practices that protect and support local wildlife of conservation concern.

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Appendix 10-A

Table 10A.1. | Bird species included in our analysis. References for minimum area requirements of forest, shrubland, or generalist birds include: Galli, Leck, and Forman 1976; Robbins, Dawson, and Dowell 1989; Pe'er et al. 2014; Vance, Fahrig, and Flather 2002; and Tirpak et al. 2008. References for minimum area requirements of grassland birds include: Herkert 1994; Herkert et al. 1999; Helzer and Jelinski 1999; Kobal, Payne, and Ludwig 1999; Johnson and Igl 2001; Terhune et al. 2010; USDA 1999; and Vickery, Hunter, and Melvin 1994. We defined minimum area as the area associated with a 50% probability of occupancy.

Common name	Scientific name	Primary habitat	Minimum area required (km^2)	Species considered to prefer patch edges or patch interiors	Neotropical/ North American/ Resident	Reference ecosystems			
						Forest	Corn	Prairie	Small grain
Bobolink	*Dolichonyx oryzivorus*	Mixed grassland, obligate	46	Interior	North American	•	•	•	•
Dickcissel	*Spiza Americana*	Mid-tallgrass	0.54	Interior	Neotropical	•	•	•	•
Eastern kingbird	*Tyrannus tyrannus*	Open savannah	0	Interior ground-nesting	Neotropical	•	•	•	•
Eastern meadowlark	*Sturnella magna*	Grassland obligate	5	Edge	Neotropical	•	•	•	•
Field sparrow	*Spizella pusilla*	Generalist	2	Generalist	Neotropical	•	•	•	•
Grasshopper sparrow	*Ammodramus savannarum*	Shortgrass, obligate	10	Interior	North American	•	•	•	•
Henslow's sparrow	*Ammodramus henslowii*	Tallgrass, obligate	12.4	Early succession, ground nester	Neotropical	•	•	•	•
Northern bobwhite	*Colinus virginianus*	Midgrass	16	Early succession	Neotropical	•	•	•	•
Northern harrier	*Circus cyaneus*	Grassland	15	Open	North American	•	•	•	•
Ring-necked pheasant	*Phasianus colchicus*	Tallgrass	5.5*	Edge	Resident	•	•	•	•
Savannah sparrow	*Passerculus sandwichensis*	Grassland, obligate; open fields	10	Interior	North American	•	•	•	•
Upland sandpiper	*Bartramia longicauda*	Shortgrass	56	Interior	Neotropical	•	•	•	•

*Considered area-independent, as are species with zero values listed.

Common name	Scientific name	Primary habitat	Minimum area required (km²)	Species considered to prefer patch edges or patch interiors	Neotropical/ North American/ Resident	Reference ecosystems			
						Forest	Corn	Prairie	Small grain
Acadian flycatcher	*Empidonax virescens*	Forest	15	Interior	Neotropical	•			
American goldfinch	*Spinus tristis*	Grassland, open/riparian woodland	0	Edge	North American migrant	•			
American robin	*Turdus migratorius*	Generalist, woodland/ farmland	0.2*	Open, generalist	North American migrant	•			
Blue jay	*Cyanocitta cristata*	Forest	0.8	Open, generalist	Resident/ North American migrant	•			
Blue-gray gnatcatcher	*Polioptila caerulea*	Forest	15	Interior	Neotropical	•			
Brown thrasher	*Toxostoma rufum*	Forest	0*	Early successional	Resident	•			
Carolina chickadee	*Poecile carolinensis*	Forest	0*	(cavity nester)	Resident	•			
Carolina wren	*Thryothorus ludovicianus*	Forest	2*	Generalist	Resident	•			
Common grackle	*Quiscalus quiscula*	Forest	0.2*	Edge	Neotropical	•			
Downy woodpecker	*Picoides pubescens*	Forest	1.2	Generalist, (cavity nester)	Resident	•			
Eastern towhee	*Pipilo erythrophthal-mus*	Forest	3*	Generalist/ early successional forest	Resident	•			
Eastern tufted titmouse	*Baeolophus bicolor*	Deciduous forest	2	Edge/forest-shrubland	Neotropical	•			
Great-crested flycatcher	*Myiarchus crinitus*	Forest	0.3*	Interior (cavity nester)	Neotropical	•			
Hairy woodpecker	*Picoides villosus*	Forest	24	Interior (cavity nester)	Resident	•			
Hooded warbler	*Setophaga citrina*	Forest	20	Interior, but uses gaps, understory	Neotropical	•			
Indigo bunting	*Passerina cyanea*	Forest	10*	Generalist, edge, shrubs	Neotropical	•			

*Considered area-independent, as are species with zero values listed.

Common name	Scientific name	Primary habitat	Minimum area required (km^2)	Species considered to prefer patch edges or patch interiors	Neotropical/ North American/ Resident	Reference ecosystems			
						Forest	Corn	Prairie	Small grain
Kentucky warbler	*Geothlypis formosa*	Forest	17	Interior	Neotropical	•			
Northern cardinal	*Cardinalis cardinalis*	Forest	24	Interior	Neotropical	•			
Northern parula	*Setophaga americana*	Forest	520	Woodland, shrubland	Resident	•			
Orchard oriole	*Icterus spurius*	Deciduous forest	0	Open forest, edge	Resident	•			
Pileated woodpecker	*Dryocopus pileatus*	Forest	165	Interior, (cavity nester), forages in low foliage	Neotropical	•			
Prothonotary warbler	*Protonotaria citrea*	Forest	30	Interior, (cavity nester)	Resident	•			
Red-bellied woodpecker	*Melanerpes carolinus*	Forest	7.5	Interior, (cavity nester)	Neotropical	•			
Red-eyed vireo	*Vireo olivaceus*	Forest	2.5	Interior	Resident	•			
Red-headed woodpecker	*Melaner-pes eryth-ro-cephalus*	Forest	0	Generalist	Neotropical	•			
Ruby-throated hummingbird	*Archilochus colubris*	Forest, coniferous	0	Generalist	Neotropical	•			
Summer tanager	*Piranga rubra*	Forest, deciduous and mixed	40	Streams	Neotropical	•			
Warbling vireo	*Vireo gilvus*	Forest, deciduous and mixed	0	Shrubby understory in gaps	Neotropical	•			
White-eyed vireo	*Vireo griseus*	Forest, deciduous	5.9	Generalist, shrub, pas-ture, (ground nester)	North American	•			
White-throated sparrow	*Zonotrichia albicollis*	Forest	0	Generalist, (ground nester)	Neotropical	•			

*Considered area-independent, as are species with zero values listed.

Common name	Scientific name	Primary habitat	Minimum area required (km^2)	Species considered to prefer patch edges or patch interiors	Neotropical/ North American/ Resident	Reference ecosystems			
						Forest	Corn	Prairie	Small grain
Wood thrush	*Hylocichla mustelina*	Forest	1	Interior	Neotropical	•			
Yellow-billed cuckoo	*Coccyzus americanus*	Forest generalist	24	Shrubs, cup nester	Neotropical	•			
Yellow-breasted chat	*Icteria virens*	Forest, coniferous, shrubland	0 (forest), 2.3 (shrub-land)	Shrubs, cup nester	Neotropical	•			
Yellow-rumped warbler	*Setophaga coronata*	Generalist, coniferous forest	0	Edge	North American	•			
Brown-headed cowbird	*Molothrus ater*	Grassland	0	Edge	Neotropical		•	•	•
Common yellowthroat	*Geothlypis trichas*	Forest	0	Early succession	Neotropical		•	•	•
Eastern bluebird	*Sialia sialis*	Forest edge	0	Edge	Neotropical		•	•	•
Mourning dove	*Zenaida macroura*	Open woodland, grassland	4	Edge	Resident/ North American		•	•	•
Red-winged blackbird	*Agelaius phoeniceus*	Grassland, wetland	24	Generalist	Resident		•	•	•
Song sparrow	*Melospiza melodia*	Early succession	24*	Edge	North American		•	•	•

*Considered area-independent, as are species with zero values listed.

11 | Forest Biodiversity and Woody Biomass Harvesting

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11.1 Background

With the expected increase in demand for woody biomass to help meet renewable energy needs, one principal sustainability question has been whether this material can be removed from forest stands while still conserving biological diversity and retaining ecosystem functioning (Hecht et al. 2009; Berch, Morris, and Malcolm 2011; Ridley et al. 2013). In general, biodiversity is the variety of life and can be considered at the genetic, population, species, community, and ecosystem levels (Berch, Morris, and Malcolm 2011). Biodiversity is often characterized as the number of species (or other taxonomic entity) and the relative abundance of each species in a defined space at a given time. A larger species pool is generally believed to indicate improved ecosystem functioning (i.e., health, resilience, goods, and services), especially in landscapes with intensified use (Loreau et al. 2001). Indices of species richness and evenness of their distribution (e.g., common or rare) are often used to measure local diversity and to compare the diversity across geographic areas. Relative abundance metrics, however, are not always good predictors of species importance for multiple reasons, but the scale of observation often dictates results (Godfray and Lawton 2001). More emphasis is being placed on understanding biodiversity through functional shifts in species assemblages in response to changing environments (i.e., ecosystem functioning) (Loreau et al. 2001; Hooper et al. 2005). Uncertainties exist on whether shifts in species assemblages, each with their own set of traits, influence ecosystem functioning even when biodiversity metrics may be similar.

Although seemingly simple in concept, the mechanisms driving variation and functional significance of biodiversity are complex, not well understood, and debated (Loreau et al. 2001; Hooper et al. 2005; Duffy et al. 2007; Berch, Morris, and Malcolm 2011). Besides human impacts on biodiversity that are often evaluated, abiotic factors, system variability, site productivity, and geographic location influence relationships between biodiversity and ecosystem function (Hooper et al. 2005; Verschuyl et al. 2011; Veech and Crist 2007). Biodiversity does not respond in a unidirectional manner to ecosystem changes. Spatial and temporal scale of observations and landscape context profoundly influence reported patterns of diversity and habitat relationships (Jonsell 2008; Efroymson et al. 2013; Gaudreault et al. 2016). Plus, few biodiversity studies span decades to understand temporal changes in communities (Magurran et al. 2010). Trophic-level interactions are also not incorporated often, but these interactions may have significant influence on local biodiversity (Duffy et al. 2007). For example, shifts in top predator species or an alteration to food chain length may have cascading effects across trophic levels. Thus, reporting and comparing commonly used metrics of biodiversity without considering functional components and the complexities mentioned above will not adequately provide information needed to evaluate ecosystem changes in biodiversity.

We take a coarse-filter approach in this chapter to assess effects of woody biomass harvesting on biodiversity within an ecological framework, rather than comparing biodiversity indices. We used the projected harvest acres output at the county level from the Forest Sustainable and Economic Analysis Model (ForSEAM; DOE 2016) in *2016 Billion-Ton Report* (*BT16*) volume 1 to describe changes in forest types producing feedstocks and forest age based on harvest type (i.e., thinning and clearcut) within ecoregion units that had the greatest projected harvest intensities compared to other ecoregions (see section 11.2). This approach examined forest changes within a habitat and ecological context to help identify species and areas that may be most affected by spatial variability in biomass sourcing. We used case studies of taxonomic groups or single species with life-history traits that rely functionally on dead and downed wood or changing canopy cover. This information may be used in conjunction with other biodiversity assessments completed at finer scales (e.g., state wildlife action plans, county project planning) to identify species that may be vulnerable to simulated changes and to help forest managers guide conservation of biodiversity at multiple spatial and temporal scales.

The primary mechanisms by which biomass harvesting may affect biodiversity are through (1) removal of fine woody debris (FWD) (tops and branches, diameter at breast height [dbh] <10 cm) and coarse woody debris (CWD) (generally defined as >10 cm dbh) and (2) alterations of other forest stand and landscape structural characteristics, such as reducing piles of forest residuals, expanding open-canopy coverage (i.e., young forest), and modifying landscape-scale forest age class distribution (Jonsell 2008; Riffell et al. 2011a; Verschuyl et al. 2011). Dead and decaying wood provides resources for a host of organisms dependent on this material (saproxylic) as a food or breeding substrate, and residue piles provide structure for many taxa as shelter, nesting, and foraging substrates, as well as other life history needs (Harmon et al. 1986; Aström et al. 2005; Jonsell 2008; Abbas et al. 2011). Organism responses to these changes are species specific and vary by forest type, geographic location, and spatial scale of observation.

Not much is known about importance of FWD to the conservation of biological diversity (Gunnarsson, Nittérus, and Wirdenäs 2004; Berch, Morris, and Malcolm 2011; Abbas et al. 2011). This material has been viewed as less critical for wildlife than CWD. Logging residues have been found to positively influence species richness because residues increase structural heterogeneity, cover, shelter, and food (Ecke, Löfgren, and Sörlin 2002). Residue piles can affect microhabitat complexity, especially after clearcutting (Ecke, Löfgren, and Sörlin 2002; Gunnarsson, Nittérus, and Wirdenäs 2002; Nordén et al. 2004; Aström et al. 2005), and have been shown to provide habitat for many small vertebrate species such as mice, voles (Aarhus and Moen 2005; Manning and Edge 2008), and arthropods (e.g., Coleoptera beetles) (Gunnarsson, Nittérus, and Wirdenäs 2004) at the local scale. Other species known to use residual slash include carnivores, meso-mammals, birds, reptiles, amphibians, and other invertebrates (Gunnarsson, Nittérus, and Wirdenäs 2004; Manning and Edge 2008). Less is known about the response of plants to FWD removal. Aström et al. (2005) note that species richness of mosses and liverworts that depend on dead wood can be reduced by removing logging residues in clearcuts, but residue removal effects on plant communities as a whole are most likely minimal and highly variable. Whole-tree harvesting may also impact the diversity of wood-inhabiting fungi (Nordén et al. 2004), especially on dry, nutrient-poor sites (Bråkenheim and Liu 1998).

Retaining CWD has been linked to conservation of biodiversity (Hura and Crow 2004; Aström et al. 2005; Franklin, Mitchell, and Palik 2007; McComb 2008). Species responses to CWD have been widely studied, and the abundance of some taxa has been linked to presence and amount of CWD, especially downed logs, in many regions of the United States (Loeb 1999; Maidens, Menzel, and Laerm 1998; McCay et al. 1998, Davis, Castleberry, and Kilgo 2010a). Results, however, differ among studies, and some have shown minimal response to CWD by some taxa (e.g., Mengak and Guynn 2003; McCay and Komoroski 2004; Davis, Castleberry, and Kilgo 2010b). As with FWD, response to CWD abundance appears to be species-, ecosystem-, and scale-dependent (Davis, Castleberry, and Kilgo 2010a, 2010b; Riffell et al. 2011a; Homyack et al. 2013; Otto, Kroll, and McKenny 2013), meaning that broad patterns of association between CWD, FWD, and biodiversity are complex. Additionally, results from recent studies of operational biomass-production practices in the southeastern United States suggest minimal or short-term species responses, potentially due to abundance of CWD retained on-site even after biomass harvests (Fritts 2014; Fritts, Moorman, et al. 2015; Fritts, Grodsky, et al. 2015b; Fritts et al. 2016), which reflected recommendations commonly found in some biomass-harvesting guidelines (Perschel, Evans, and DeBonis 2012).

Forest woody-biomass harvesting includes traditional forest-harvesting methods, such as thinning and clearcutting. Thinning decreases tree density, increases forest canopy gaps, and can alter abundance and diversity of mid-story trees (Artman 2003; Agee and Skinner 2005; Hayes, Weikel, and Huso 2003; Harrod et al. 2009). Thinnings can be conducted pre-commercially, commercially, or as a fuels treatment (Verschuyl et al. 2011). Because thinning reduces overstory stem density and increases light availability below the canopy, it can lead to the development of more complex understory vegetation (Doerr and Sandburg 1986; Bailey and Tappeiner 1998; Wilson and Carey 2000; Garman et al. 2001; Homyack et al. 2015). Verschuyl et al. (2011) used meta-analysis to evaluate relationships between forest-thinning treatments and forest biodiversity from 33 studies conducted across North America. They found that forest-thinning treatments had generally positive or neutral effects on diversity and abundance across all taxa, although thinning intensity and the type of thinning conducted may at least partially drive the magnitude of response.

Clearcutting associated with woody-biomass harvesting obviously changes forest stands to a state of early succession and also influences forest age distribution across a landscape. Although clearcutting negatively affects species associated with older forest structure, many species require early successional forest conditions. The extent of young forest has been declining across the United States, especially in eastern forest regions, as have population trends of birds associated with this habitat (see, e.g., Brooks 2003; King and Schlossberg 2014). Regenerating forest from clearcuts may improve habitat suitability for some declining forest interior birds (Ahlering and Faaborg 2006), and birds typically associated with mature forests seek out this early seral-stage post-fledging to take advantage of abundant fruits and seeds (Stoleson 2013).

Understanding variability of residual CWD and FWD left after clearcutting is critically important to understand how amounts may influence ecosystem processes. Many best management practices (BMPs) recommend leaving residue to provide microhabitat structure (Abbas et al. 2011). Recent studies in the southeastern United States have found that the amount of CWD left on sites after biomass harvests is higher than amounts commonly recommended in biomass-harvesting guidelines and removal effects on wildlife appear to be minimal or short term (Fritts 2014; Fritts, Moorman, et al. 2015; Fritts, Grodsky, et al. 2015; Fritts et al. 2016; Perschel, Evans, and DeBonis 2012).

This chapter describes potential forest changes and implications for biodiversity resulting from expanding U.S. national biomass production. Specifically, we assess and compare effects of potential forest biomass produced in the near term (2017) and in significantly expanded biomass-production scenarios (2040) generated in volume 1 of *BT16* at the national level. Volume 1 investigates the potential economic availability of biomass resources at the roadside using an economic supply curve approach, assuming latest-available yield and cost data. An important aspect to understand is that this assessment is evaluating potential additive effects of removing logging residues associated with conventional harvests as well as expanded whole-tree biomass harvests within the assumptions of ForSEAM (see section 11.2.2). We do not attempt to evaluate the effects of conventional harvest on biodiversity, nor do we attempt to determine landscape-level or cumulative effects due to the scale of these data (i.e., county-level) and the fact that only two points in time are being compared. However, in some cases, effects of forest woody biomass harvest may be similar to the effects of conventional harvests. Assessment results, however, can provide information to help prioritize future research needs for specific species and communities based on forest-change scenarios. Results can also foster more focused investigations on critical thresholds of biomass removal and interactions of woody biomass harvest with other anthropogenic and natural factors relative to conservation of biological diversity.

11.2 Methods

Given the geographic extent representing numerous ecological contexts contained within this assessment, it was not possible to investigate all species that rely on dead and downed wood or young forests. To refine our assessment, we used the USDA U.S. Forest Service's National Hierarchical Framework of Ecological Units developed for the contiguous United States (ECS; Cleland et al. 2007) to identify ecoregion units that are expected to supply the greatest quantities of feedstock. This hierarchical framework classifies ecological types and maps ecological units based on associations of climate, physiography, and biotic characteristics that distinguish a unit from neighboring ones. The framework incorporates energy, moisture, and nutrient gradients that regulate the structure and function of ecosystems. Within each selected ecoregion unit, we describe primary forest changes that may drive the responses of species to removing feedstock. We used the province ecoregion unit (fig. 11.1), which is at a scale of millions to tens of thousands of square kilometers; this is an appropriate scale for assessments and strategic planning. Next, based on information in the scientific literature, we discussed implications of the forest type and structure changes to biodiversity-indicator case-study species found within each selected province.

As with other chapters in this report, we used environmental indicators and, in particular, biodiversity indicators, suggested by McBride et al. (2011), which include presence and associated habitat area for taxa of special concern that may be directly affected by forest changes related to forest woody-biomass harvesting. Taxa of concern can be categorized into 6 groups: (1) rare (or could become rare) native species; (2) keystone species that have a disproportionately large impact relative to abundance; (3) bioindicator taxa that monitor the condition of the environment; (4) species of commercial value; (5) species of cultural importance, or (6) species of recreational value.

Text Box 11.1 | Definitions from *BT16* Volume 1

- Forestland—land at least 120 ft (36.6 m) wide and 1 acre in size with at least 10% cover by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated.
- Timberland—forestland that is producing, or is capable of producing, in excess of 20 ft^3 (0.57 m^3) per acre per year of industrial wood and not withdrawn from timber utilization by statute or administrative regulations.

We focused our attention on vertebrate species that depend on CWD or FWD (e.g., amphibians—bioindicators) or rely on structure of residue woody material (e.g., piles) for shelter, feeding, or foraging, such as ground nesting birds, small mammals, and furbearers (e.g., American marten); we also focused on those species that may respond to open-canopy conditions, such as reptiles (e.g., gopher tortoise—keystone species) and game species. Based on the potential forest change for each province, we selected several representative species within each of the categories above and species functional groups. By targeting species within these categories, we were better able to assess potential effects of additive biomass harvest and help identify species and ecosystems for further consideration in BMPs, strategic planning, and scientific investigations. Saproxylic organisms such as invertebrates and wood-inhabiting fungi would be the primary species impacted by biomass harvests, since they depend directly on dead wood during part of their life cycle. However, not much is known about these species, nor are there adequate data to determine their presence.

11.2.1 Scope of Assessment

For the purposes of this report, we assessed potential effects on biodiversity indicators from forest change resulting from biomass harvests on timberland (text box 11.1) under select scenarios from *BT16* volume 1. We used county-level data down-scaled from ForSEAM analysis units (see *BT16* volume 1, fig. 3.16, p. 73) to (1) summarize projected change in harvest acres between the near-term baseline (moderate housing–low wood energy scenario, ML 2017) and expanded production under baseline (ML 2040) and high-yield (high housing–high wood energy scenario, HH 2040) growth assumptions by 2040 (table 11.1); (2) spatially identify geographic areas expected to have greater harvest intensities; (3) describe forest-structure changes based on forest habitat-cover types that supply feedstock within those geographic areas; and (4) infer how these changes may affect selected biodiversity indicators using case studies of wildlife taxa that functionally depend on dead and downed wood, residue piles, or open forest canopy (i.e., young forests).

Table 11.1 | Description of Wood Energy and Housing Scenarios (modified from *BT16* volume 1, table 3.6)

Land type (million acres)	Baseline 2015	Extended Baseline 2040
Moderate housing–low wood energy (baseline), ML	Returns to long-term average by 2025	Increases by 26% by 2040
High housing–high wood energy, HH	Adds 10% to baseline in 2025	Increases by 150% by 2040

USDA, U.S. Forest Service, Forest Inventory and Analysis

The forest biomass feedstocks considered were forest residues (i.e., logging residues) and whole-tree biomass from harvests of smaller-diameter merchantable stands (i.e., biomass-only harvest). Logging residues were generated as a product from conventional harvests. Whole-tree biomass was generated from commercial and noncommercial trees of smaller-diameter merchantable stands or removal of excess biomass from fuel treatment and thinning operations designed to reduce risks from catastrophic fires and improve forest health. The harvest method, whether full-tree or cut-to-length, differed among ForSEAM analysis regions (see under each ForSEAM region below), which impacted whether logging residues stayed on-site. Under the cut-to-length harvest method, residues stayed on-site (i.e., trees are felled, delimbed, and bucked directly in the stump area and then log sections are transported to landing or roadside). Under full-tree method, the whole tree (aboveground portion) was brought to a landing for processing, and residue was recovered. Also, merchantable materials were assumed to be harvested as roundwood.

We used the center of each county to delineate whether it was included in the province ecoregion of interest. We used harvest acres as the response variable rather than volume of feedstock produced because the amount of habitat is a major metric for vertebrate species. The number of acres harvested was highly correlated with the volume of feedstock produced for logging residues ($r = 0.87$) and whole-tree biomass harvests ($r = 0.77$).

11.2.2 Relevant ForSEAM Assumptions

The following aspects of ForSEAM are important to understand because these assumptions influence spatial and temporal patterns of woody biomass-supply projections reported for each ForSEAM analysis region, which ultimately influences projections within each province ecoregion and forest type:

- As an economic model, ForSEAM compared the relative costs of raw material inputs and met demands using harvest (including stumpage) residues first, then the least expensive harvest of whole trees, and finally, higher-cost harvests of whole trees.
- The model first solved for conventional timber demands (i.e., sawtimber and pulpwood), which generated logging residues (i.e., integrated harvest). Whole-tree biomass harvests did not occur unless demand for woody biomass was not met by logging residues.
- Availability of biomass declined through time as the model captured how and when materials were harvested, meaning that a harvest in year T impacted output in year T + 1. The land category transitioned from "available" to "regenerating," and over the short duration of modeling (2017 to 2040), land was, at the most, available for harvest only one more time.
- Only timberland <0.5 miles (<0.8 km) from roads with ≤40% slope (except Inland West region) were considered available for harvest. For most counties, only up to 5% of forestland was available for harvest in the model.
- Forest cover type remained consistent, meaning there was no land-use or cover change (e.g., natural stands of softwood were not converted to plantations, and marginal agricultural lands were not converted to forest).
- Only 70% of available logging residues were recovered from clearcut full-tree harvests on timberland with ≤40% slope to incorporate BMPs (i.e., 30% residues retained on-site). No logging residues were removed on timberlands with ≥ 40% slope. During thinning operations associated with whole-tree biomass harvests, all residues were harvested under the assumption that tree breakage during harvest would result in some retention of residues.
- Only small- and mid-diameter stands were harvested as whole-tree biomass. Harvest of mature trees provided stand-regeneration opportunities (i.e., age-class distribution) and affected availability of the next generation of small- and mid-diameter removals for biomass (i.e., harvest with no thinning for the next 10–15 years following final harvest). All diameter classes (class 1, >11 inches for hardwood or >9 inches for softwood; class 2, diameter 5–11 inches for hardwood or 5–9 inches for softwood; class 3, diameter <5 inches) could be clearcut.

11.3 Results

We report projected forest change under each scenario and time at several scales, followed by potential biodiversity effects. We first report national-scale changes, followed by changes at the ForSEAM regional level, and then by the province ecoregion unit that encompasses the concentration of counties having greater harvest intensities (e.g., >5,000 acres harvested) (fig. 11.1). Within each of these scales, we report total acres harvested by each source feedstock as well as acres in young forest. We then discuss the biodiversity effects on particular taxonomic groups or individual species that could be affected by the described forest changes at each scale.

11.3.1 Conterminous United States

Overall, approximately 8.5 million total acres were harvested for forest woody-biomass under the ML 2017 scenario, with harvested acreages reduced by 51% and 61% under ML 2040 and HH 2040 scenarios, respectively. At the ForSEAM region level, total acres harvested declined under both 2040 scenarios from ML 2017 projections for all regions except the Inland West (IW), where ML 2040 totals increased by 9.6% (fig. 11.2a). Under all three scenarios, approximately half of the national woody biomass-feedstock supply was projected to be harvested on lands within the South (S) region of the United States, 51%–57% across all three scenarios (fig. 11.2a). This pattern is a result of logging residues entering the model first to meet region demands of an increasing pellet market (*BT16* volume 1, p. 43).

The counties with >5,000 acres harvested for woody biomass in the ML 2017 scenario were concentrated mostly throughout S forests, especially in Louisiana, Arkansas, Alabama, and South Carolina; in North Central (NC) forests, primarily in northern Minnesota, Michigan, and Wisconsin; in Northeast (NE) forests, primarily in Maine; in Pacific Northwest (PNW) forests, primarily in northern California and southern Oregon; and in IW forests, primarily in northern Idaho and western Montana (fig. 11.3).

Figure 11.1 | Delineation of ecoregion provinces overlaid on total potential acres harvested under the ML 2017 scenario, which had the greatest quantity of total acres harvested of all scenarios. Black letters indicate ForSEAM regions outlined by bold black lines; red numbers indicate province ecoregions. See text for descriptions.

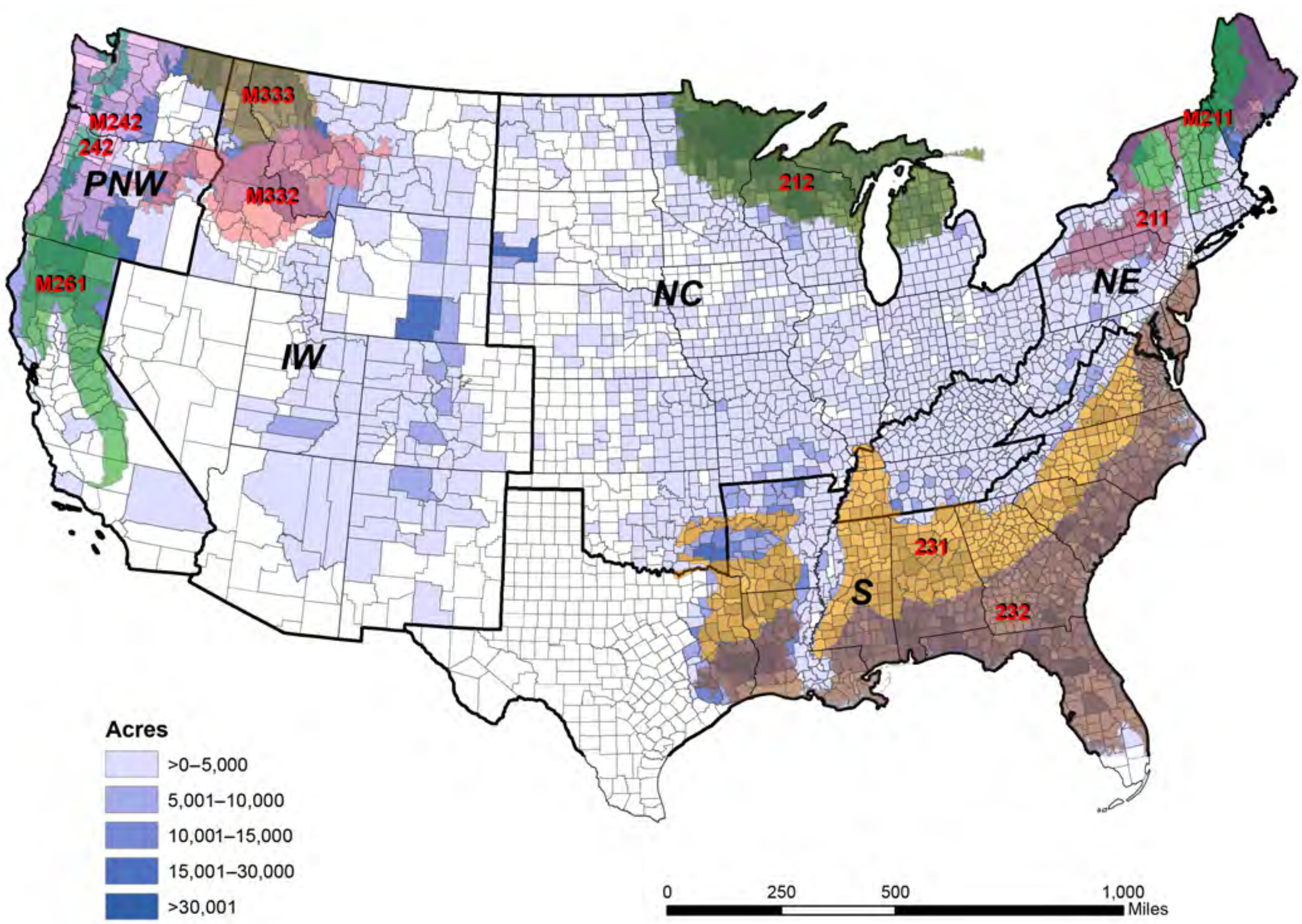


Figure 11.2 | Potential total acres harvested in each scenario—baseline (moderate housing–low wood energy) in the near term (ML 2017) and expanded production under baseline- and high-yield (high housing–high wood energy) in 2040 (ML 2040 and HH 2040, respectively)—in each ForSEAM region for (a) all feedstocks, (b) logging residues, and (c) whole-tree biomass harvests; note differences in scale.

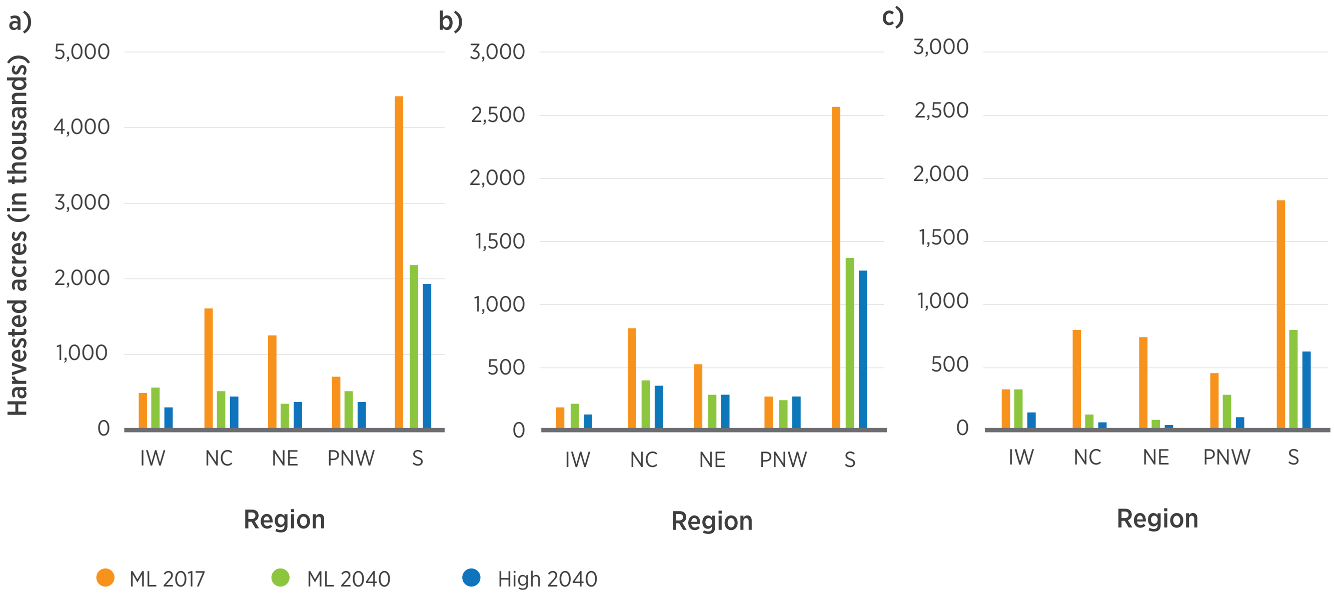
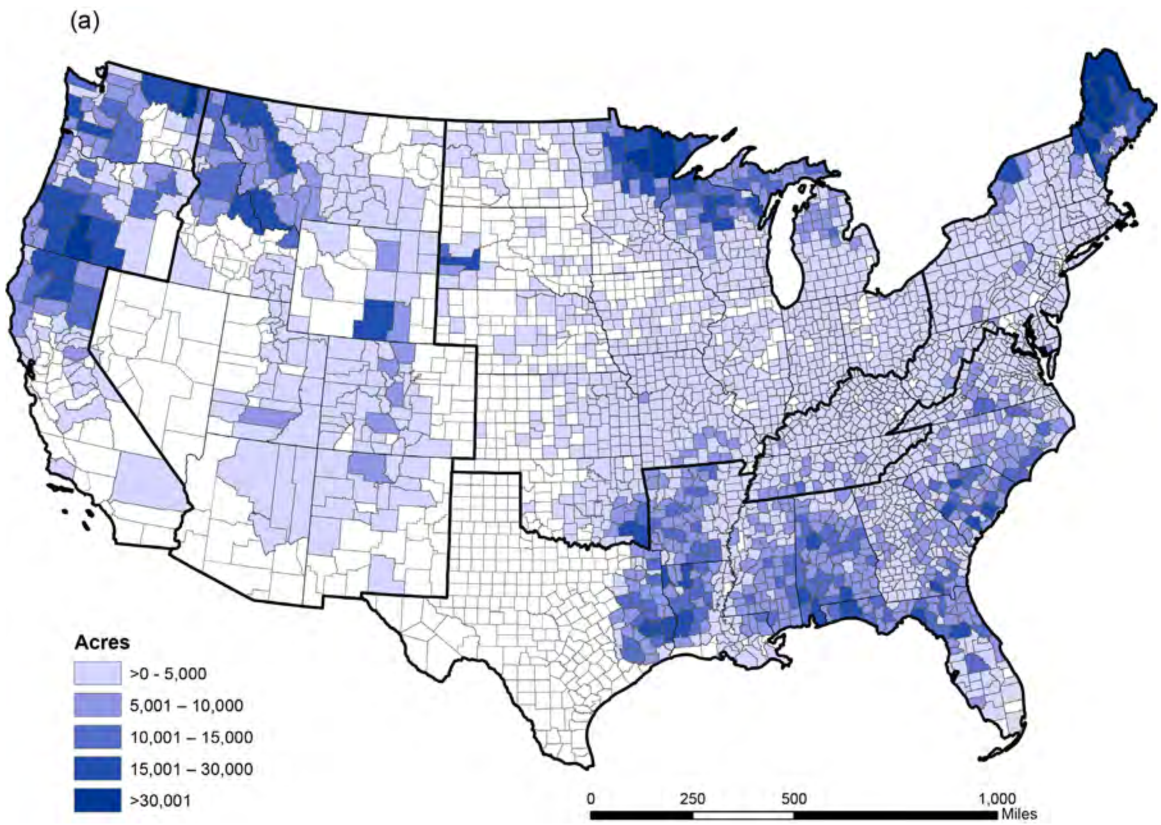
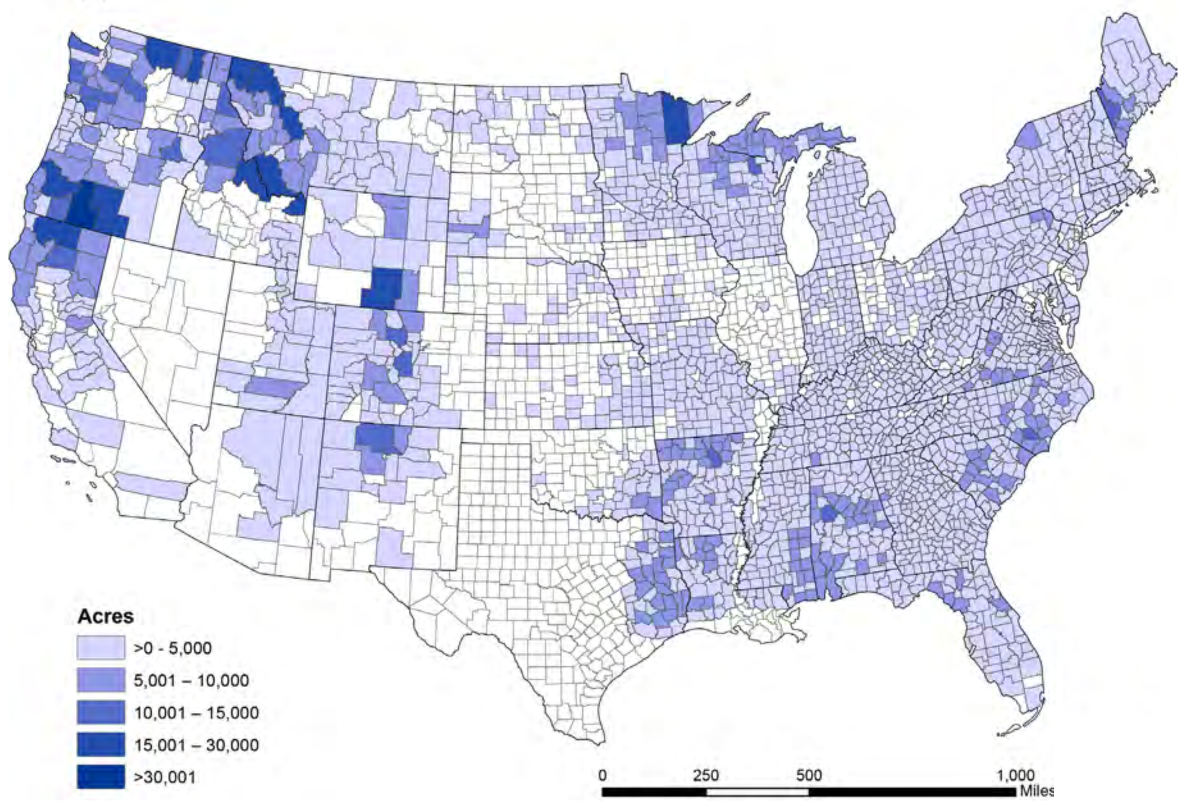


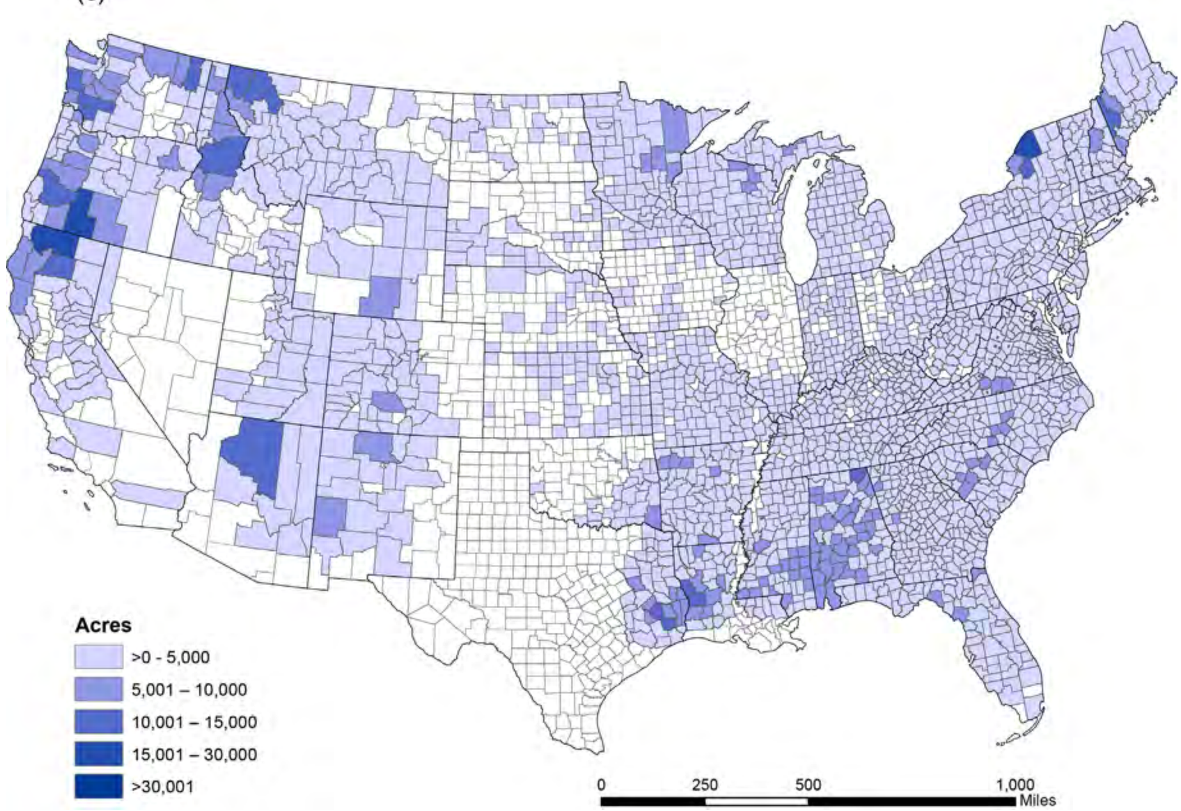
Figure 11.3 | Distribution of projected total acres from all feedstocks harvested by county under three biomass scenarios: (a) ML 2017, (b) ML 2040, and (c) HH 2040.



(b)


Acres
>0 - 5,000
5,001 – 10,000
10,001 – 15,000
15,001 – 30,000
>30,001
0
250
500
1,000
Miles

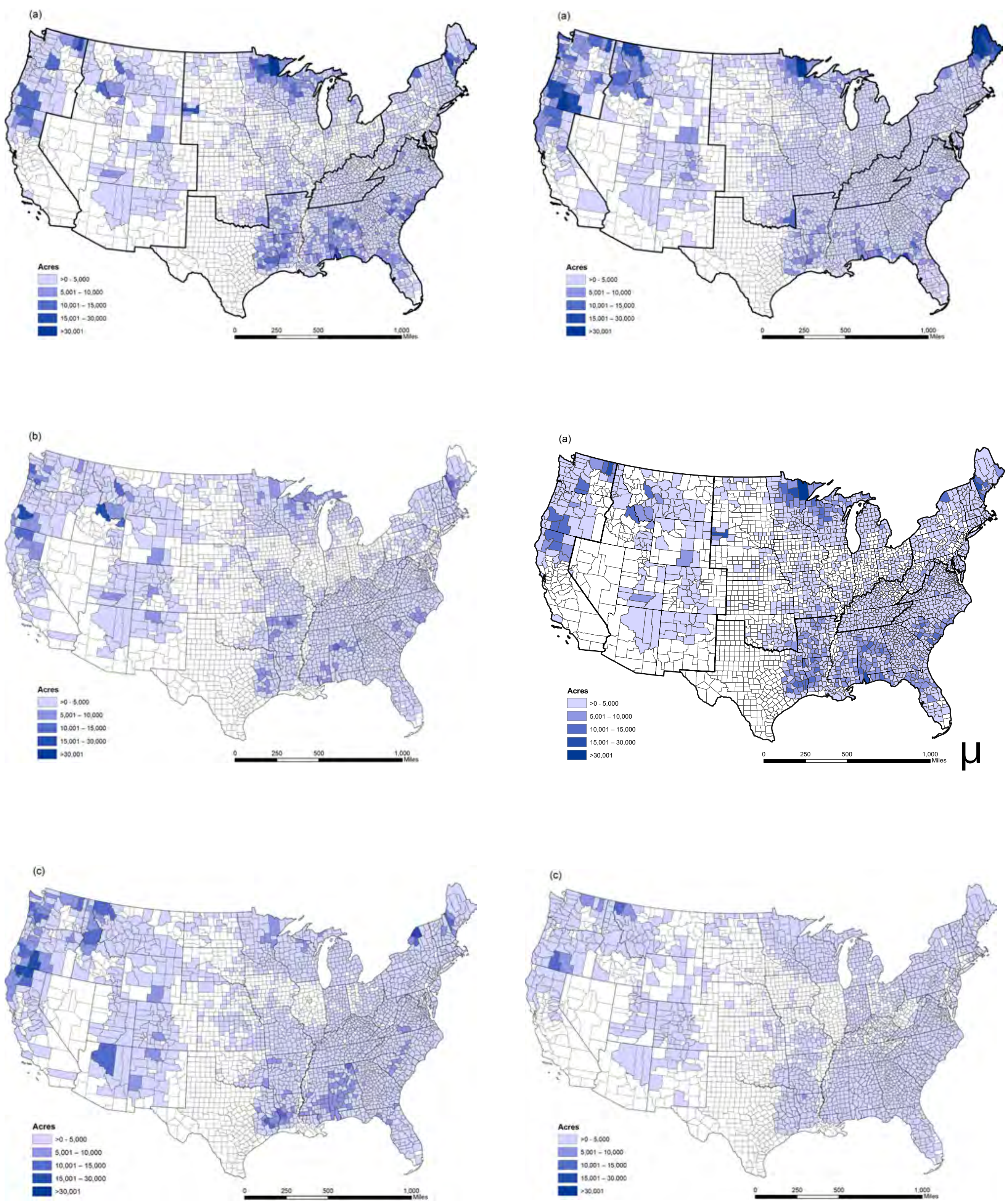
(c)


Acres
>0 - 5,000
5,001 – 10,000
10,001 – 15,000
15,001 – 30,000
>30,001
0
250
500
1,000
Miles

Logging residues remained the primary feedstock under all scenarios in the S and NC regions, while whole-tree biomass remained the primary feedstock under all scenarios in the IW region (fig. 11.2b and 11.2c). Comparing the distribution of counties with >5,000 acres among feedstock type and ForSEAM regions also shows more counties with greater acres of logging residues than whole-tree biomass harvests (fig. 11.4). Logging residues under ML baseline scenarios decreased in all regions from 2017 to 2040, except in IW where it increased by 27% (fig. 11.2b). Comparing ML 2040 and HH 2040 scenarios, logging residues declined slightly in the S, NC, and IW, while increasing slightly in the NE and PNW regions (fig. 11.2b). The distribution of counties with >5,000 acres of harvest narrows to S, NC, and PNW primarily, with several counties also in NE and IW (fig. 11.4).

Harvested acres of whole-tree biomass declined in all regions between ML 2017 baseline and HH 2040 scenarios, except for IW where acreage was constant between ML 2017 and ML 2040 baseline scenarios (fig. 11.2c). This same pattern existed between ML 2040 and HH 2040 scenarios as well. Whole-tree biomass was the primary feedstock harvested in NE in near-term (ML 2017), but logging residues became the primary feedstock under ML 2040 and HH scenarios. For PNW, whole-tree biomass was the primary feedstock under ML 2017 and ML 2040 scenarios, but logging residues became the primary feedstock under HH 2040 scenario (fig. 11.2c). The distribution of counties with >5,000 acres narrows within PNW to the northwest with few counties in the remaining regions (fig. 11.4).

Figure 11.4 | Distribution of projected total acres harvested by county for logging residue feedstock (*left*) and whole-tree biomass feedstock (*right*) incorporating clearcut and thinning harvest types under scenarios (a) ML 2017, (b) ML 2040, and (c) HH 2040. Bold black lines in top panels delineate ecoregion provinces (see fig. 11.1).



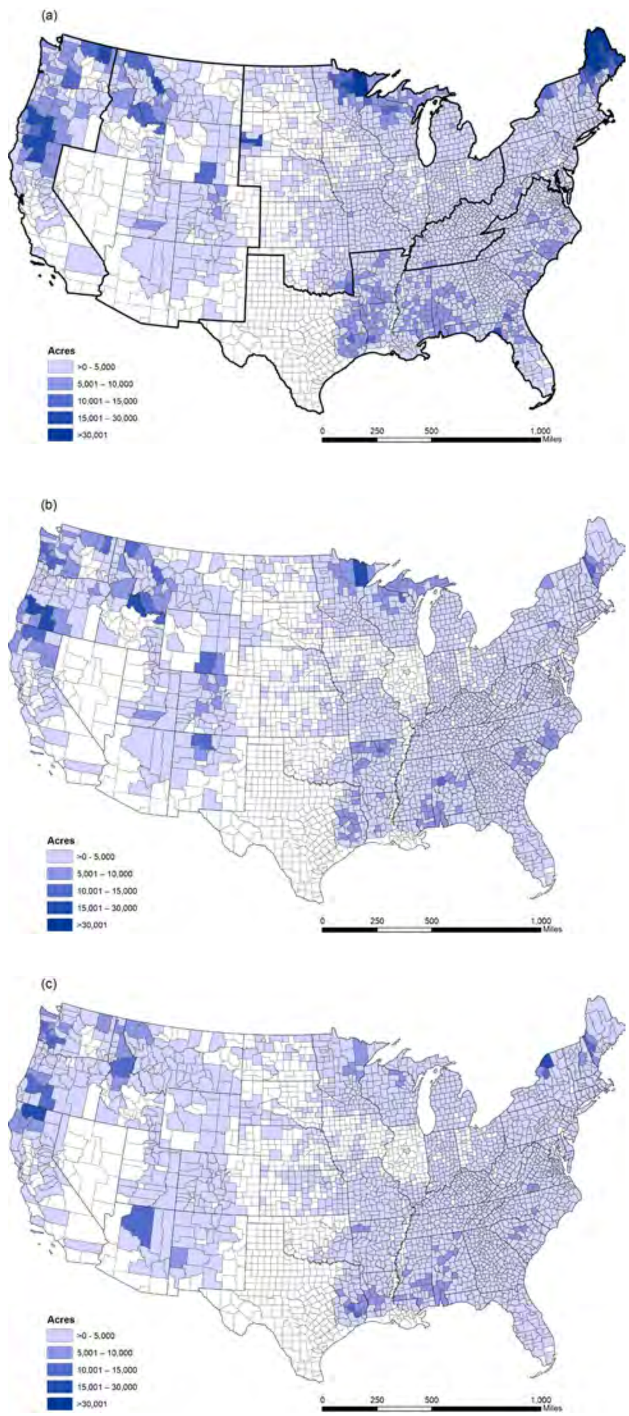
The distribution of young forests as a result of conventional and whole-tree harvest clearcutting was concentrated in the NE, upper NC, central PNW, and central S regions (fig. 11.5). However, under ML 2040 and especially HH 2040, the NE had fewer counties with large acres of clearcutting than did the Atlantic coast in the S, as well as the areas along the southern Rocky Mountains of Montana and Colorado.

11.3.1.1 Biodiversity Effects

Reduced biological diversity is caused by local extinctions, which can be a result of natural or human-induced factors. Habitat loss and fragmentation is identified as a significant driver of biodiversity loss (Reed 2004). Geographic distribution of species-endangerment patterns across the continental United States are typically unevenly distributed, concentrated into a few areas, separated along a land-cover gradient, and system-related (Dobson et al. 1997; Flather, Knowled, and Kendall 1998). Areas with greatest species endangerment have been found to be in the arid Southwest, Florida, southern Appalachia, and along the Atlantic, Gulf, and northern Pacific coastlines (Dobson et al. 1997; Flather, Knowled, and Kendall 1998). Because different factors have driven these patterns, endangered biota differ by area, with birds and reptiles driving trends in the eastern United States and aquatic vertebrates driving patterns in the western United States. Many imperiled species have faced face habitat loss associated with broad-scale processes such as urbanization, grazing, or altered natural disturbances (e.g., fire suppression), but local factors also contributed (Flather, Knowled, and Kendall 1998). By comparison, counties with greater harvest intensities in this assessment were also along the Gulf, Atlantic, and Pacific coasts, but were not concentrated in southern Appalachia, and there were only a few counties in the southwestern United States with high harvest intensities (fig. 11.5).

Because timberland area remained constant in ForSEAM, and other processes that may drive biodiversity patterns besides habitat area were not incorporated into ForSEAM, we mainly selected indicator species based on life-history characteristics and

Figure 11.5 | Distribution of projected young forest acres after logging residue and whole-tree harvests using clearcuts harvest type under (a) ML 2017, (b) ML 2040, and (c) HH 2040.



habitat associations for potential forest type changes within the ecoregion under consideration. In general, some characteristics of extinction-prone species are low reproductive rate; feeding at high trophic levels; large body size; limited or specialized nesting or breeding habitat; restricted or patchy distribution; poor dispersal ability; and low population densities but large individual ranges. Due to their life-history characteristics and occurrence in all regions of this assessment, amphibians may be a group of species that show biodiversity effects of woody biomass harvesting at national to local scales (see text box 11.2).

11.3.2 South Region

Overall, 51%–57% of the projected total acres harvested for woody biomass occurred in the S under ML 2017 and HH 2040 scenarios (fig. 11.2a). Total acres harvested declined approximately 50% from ML 2017 to ML 2040 and HH 2040, but the acres harvested were only 12% lower in HH 2040 than in ML 2040. Logging residues were the primary feedstock under all scenarios (fig. 11.2b and 11.2c), and were harvested from approximately the same proportion of land under ML and HH scenarios (58% and 63% in ML 2017 and ML 2040,

Text Box 11.2 | Case Study: Lungless Salamanders (Family Plethodontidae)—Bioindicators

Due to permeable skin, amphibians are considered environmental bioindicators. Amphibians are often abundant in ecosystems and play an important functional role as apex predators in detrital food webs (Davic and Welsh 2004). However, amphibians are declining across the nation at a projected rate of 3.79% per year, which may result in half of occupied sites becoming locally extinct within the next 19 years (Grant et al. 2016). No strong region-specific driver or single cause has been shown to account for this decline; local factors appear to have a stronger influence on viability (Grant et al. 2016). Reviews of field studies show amphibian numbers are positively correlated with dead wood, and retaining this material can reduce effects of forest harvest; however, results are also species- and system-specific and specific to the size of dead wood (Riffell et al. 2011a, 2011b; Otto, Kroll, and McKenny 2013). Conventional, partial-cut harvests affect amphibians less than clearcuts that open the canopy and increase desiccation risk, especially for young amphibians (Semlitsch et al. 2009). Lungless salamanders (Family Plethodontidae) may be more sensitive to woody-biomass harvests as many species are closely associated with forests that provide a moist environment with a large supply of invertebrate prey and dead wood that provides cover from predators and nesting substrate. Removing dead wood may increase risks of predation and desiccation, especially for those species with small home ranges and poor dispersal capabilities (see Petranka 2010).

Collectively, this taxonomic group contributes substantially to biodiversity at local to continental scales. For example, more than 40 species are found in the S, and 11 of these species are listed on the International Union for the Conservation of Nature (IUCN) red list, and all regions have these salamanders. The presence of lungless salamanders across the United States varies from locally common populations with restricted geographic distributions to patchy or continuous populations with broad geographic distributions (see Petranka 2010).

Concern for these species may be most relevant in areas of the nation expected to have greater intensities of clearcutting activities due to whole-tree biomass harvesting, such as the NE and IW, where extensive open areas with decreased residues may restrict movement enough to further isolate metapopulations. Another potential effect of harvesting residues concerns forest types that harbor high proportions of these species. Reduced retention of larger-diameter residues may lower availability of defendable nesting sites and foraging opportunities, causing a decline in local populations. However, few studies have separated the effects of residue removal from the effects of conventional harvest (Otto, Kroll, and McKenny 2013). Best management practices (e.g., buffers, minimum residue retention guides) for some of the more endemic species in the group (e.g., those found in Appalachia) may minimize any additive effects, but many common species with broad distributions, such as the red-backed salamander (*Plethodon cinereus*) common in northern forests, are not usually addressed specifically.

respectively, and 66% under HH 2040 scenario; fig. 11.2b and 11.2c). For the S region, 100% full-tree harvest type is defined under ForSEAM as felled trees taken to the landing to be processed and where either the whole tree or remaining waste could be chipped (i.e., no cut-to-length harvest type occurred in this region).

The greatest concentration of counties with >5,000 acres of potential total harvest occurred more often in the Gulf region than in the Atlantic region, with lower harvested acres in 2040 under both ML and HH scenarios (fig. 11.4a, b, and c). In the Gulf region, counties with >10,000 acres of potential harvest were in east Texas, Louisiana, Alabama, and the Florida panhandle, but few counties exceeded 10,000 acres of total biomass harvest in 2040 under ML and HH scenarios. In the Atlantic region, counties with >10,000 acres of potential harvest were primarily in South Carolina, coastal North Carolina, and south Georgia, but there were no such counties in both 2040 scenarios (fig. 11.4). These spatial patterns were mostly attributed to distribution of logging residue harvests (fig. 11.4b).

The majority of the land base that had greatest acreage of woody biomass harvesting was within the Southeastern Mixed Forest Province (231) (see text box 11.3) and the Outer Coastal Plain Mixed Forest Province (232) (see text box 11.4) under all scenarios, with approximately equal harvested acres in both (fig. 11.1). Within each province ecoregion, almost all counties had some woody biomass-harvest activity under all scenarios. Therefore, we limited our examination of forest change to these two ecoregions. However, it is worth mentioning that northern Arkansas had several counties with > 5,000 acres of potential woody biomass harvesting. This pattern is primarily a result of clearcutting, which created young forests in the 2040 baseline scenario (fig. 11.5b).

11.3.2.1 Province 231

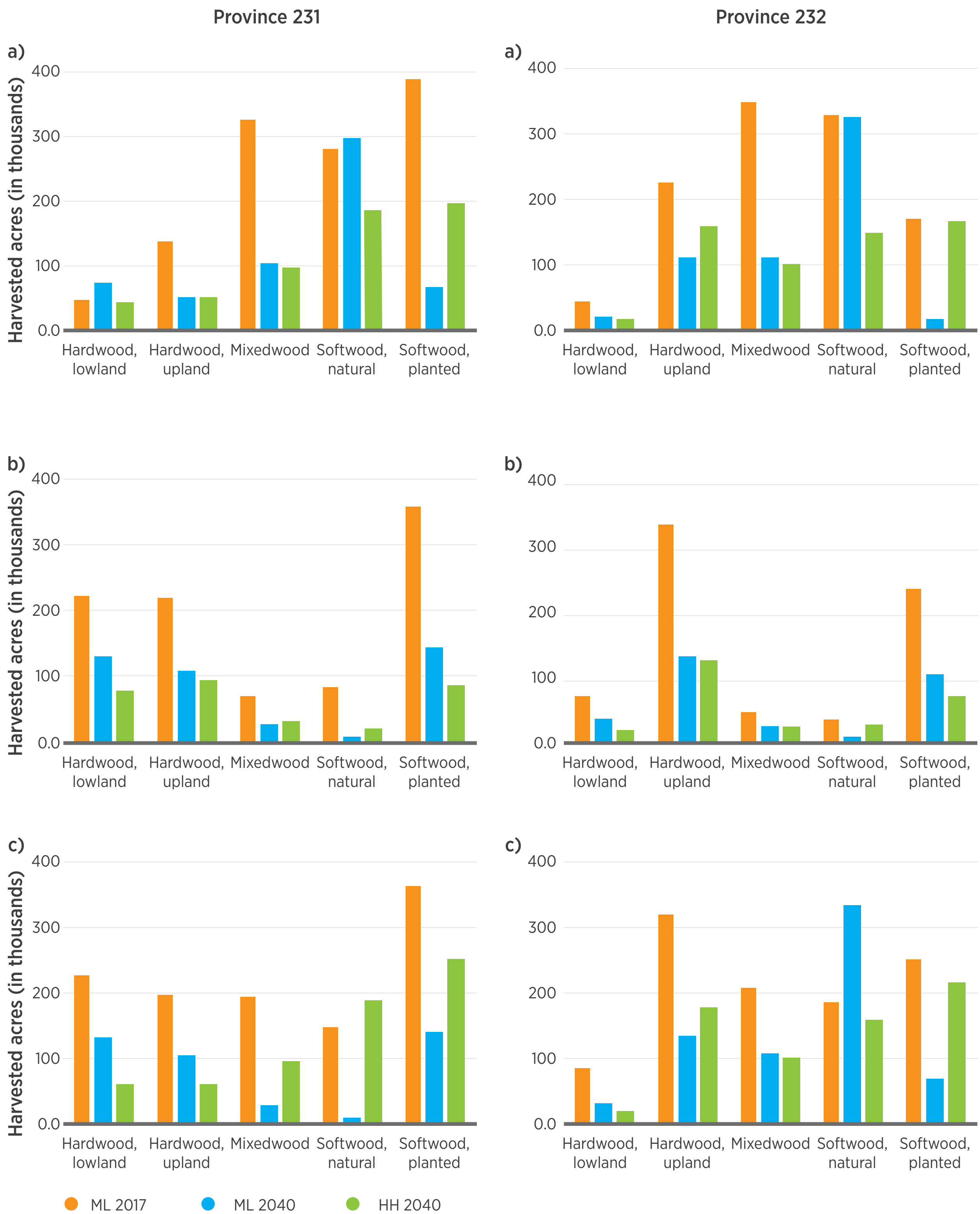
Province 231 covers 116.2 million acres (about 24.8% of the S). Under ML 2017, approximately 2 million acres were harvested (logging residues and whole-tree harvest), representing about 2% of the province. Harvested acres were reduced by half under both 2040 scenarios. Combined, planted, and natural softwoods produced approximately half of the feedstock under all scenarios (fig. 11.6). Planted softwood predominated in ML 2017 and HH 2040 scenarios (35.4% and 32.7%, respectively), but represented only 21.0% under ML 2040. Instead, natural softwood predominated under this scenario (30.9%).

All counties had some potential woody-biomass harvests under ML 2017, with the greatest concentration of counties having >5,000 acres harvested located in northeast Texas, northern Louisiana, southern Arkansas, eastward into northern Mississippi and Alabama, and northwest South Carolina and Virginia (fig. 11.1). This spatial pattern was driven by removal of logging residues for all scenarios (fig. 11.4b and 11.4c). Only three counties had >10,000 acres of potential whole-tree biomass harvest under ML and HH 2040 scenarios. Counties with greater areas of young forest as a result of whole-tree biomass harvests occurred in northern Arkansas primarily, followed by western South Carolina and West Virginia, but relatively few counties had harvests under the HH 2040 scenario (fig. 11.5).

Text Box 11.3 | Province 231: Southeastern Mixed Forest

- Maritime climate with mild winters and hot, humid summers; precipitation evenly distributed, but mid- to late-summer droughts may occur
- Hilly landscape with increasing relief farther inland
- Vegetation mixture of deciduous hardwoods and conifers
- Lowland hardwoods—primarily sweetgum/nuttall oak/willow oak and sugarberry/hackberry/elm/green ash
- Upland hardwoods—primarily sweetgum/yellow-poplar and white oak/red oak/hickory
- Mixedwood—primarily loblolly and shortleaf pine with southern red oak
- Natural softwoods—primarily loblolly and shortleaf pine, as well as Virginia pine
- Planted softwoods—primarily loblolly and Virginia pine.

Figure 11.6 | Potential acres harvested by forest cover for the Southeastern Mixed Forest Province (231; left) and the Outer Coastal Plain Mixed Forest (232; right) within the southern region by (a) logging residue feedstock, (b) whole-tree biomass feedstock, and (c) open forest canopy condition (i.e., young forests).



For logging residues under both 2040 scenarios, primary forest change was a 51% reduction in acres producing this feedstock from the 1.16 million acres harvested under ML 2017. Planted and natural softwoods produced most logging residues under all scenarios, from 57% under ML 2014 to approximately two-thirds in both 2040 scenarios (fig. 11.6). Within these acres, planted softwood predominated under ML 2017 (58.2%) and HH 2040 (35.0%), but represented only 11.0% in ML 2040; natural softwoods predominated in ML 2040 (52.0%). Thinning all forest cover classes (diameter class 2) produced 78.0% of this feedstock under ML 2017, while clearcutting (diameter class 1) produced all logging residues under both 2040 scenarios. Specifically, by forest cover in ML 2017, most logging residues were harvested from planted softwood (33%) and mixedwood (28%) forests, followed by natural softwood forests (24%) through thinning and clearcutting. The lowest quantities of logging residues were projected to be harvested from lowland hardwoods (3%). Under ML 2040, natural softwood harvests produced 52% of logging residues while mixedwood yielded 17%, and upland hardwood generated the least (8%). Under HH 2040, planted and natural softwoods each produced approximately a third of logging residues, and mixedwood produced 16%. Lowland hardwoods remained lowest at 7%. The majority of counties with >5,000 acres producing logging residues were located in northern Alabama.

For whole-tree biomass, harvests occurred on fewer acres than logging residues (930,000 acres under ML 2017) and declined 57% under ML 2040 and 69% under HH 2040. Harvest of planted softwoods produced 38.2% of feedstock under ML 2017, 34.8% under ML 2040, and 28.2% under HH 2040. However, acres harvested declined 60.5% under ML 2040 and 76.7% under HH 2040 from ML 2017 levels (fig. 11.6b). Combined, upland and lowland hardwood harvests represented 46.5% of feedstock in ML 2017, 57.7% in ML 2040, and 56.8% in HH 2040. Natural softwoods produced the lowest fraction of feedstock in all 2040 scenarios. Under ML 2017, 92% of harvested acres for whole-tree biomass were produced by clearcutting diameter classes 2 and 3, but in 2040, acres harvested by clearcutting (class 2) under ML and HH scenarios were only 26.7% and 28.0%, respectively. Under the ML 2040 scenario, no counties had <5,000 acres harvested for whole-tree biomass.

For young forests, approximately 1.1 million acres were produced through clearcutting under ML 2017, primarily from planted softwood (32%), followed by lowland hardwoods (32%). Under ML 2040 scenario, acres in young forest were primarily produced from clearcutting of natural softwoods (42%). Natural and planted softwoods produced two-thirds of young forests under ML 2040 and HH 2040 scenarios, followed by mixedwood (14%) for both scenarios. This trend was similar to logging residues because clearcutting was the primary method of generating feedstocks. the few counties with <5,000 acres harvested were primarily in northern Alabama.

11.3.2.2 Province 232

This province covers 137.8 million acres (about 29.4% of the S). Under ML 2017, approximately 1.8 million acres were harvested (logging residuals and whole-tree harvest), representing nearly 2% of the province. Total acres harvested were slightly less than those in Province 231, but the same reduction by approximately half was modeled in ML and HH 2040 scenarios (fig. 11.6). In addition, planted and natural softwoods produced approximately half of total acres harvested for woody biomass under both 2040 scenarios but represented only 41.9% under the ML 2017 model. Upland hardwoods produced the greatest quantity of woody biomass in ML 2017 and HH 2040 scenarios (30.5% and 32.9%, respectively). Natural softwoods produced the most biomass in the ML 2040 line (37.2%), followed by upland hardwood (27.3%). Lowland hardwoods produced the least woody biomass under all scenarios.

Text Box 11.4 | Province 232: Outer Coastal Plain Mixed

This province is characterized by the following:

- Gentle topography and very low (<90 m) elevation
- Humid, maritime climate with mild winters, and warm summers with rare periods of summer drought
- Vegetation dominated by conifers with deciduous hardwoods along major floodplains
- Lowland hardwoods—primarily bald cypress, black gum, and overcup oak
- Upland hardwoods—primarily oak, hickory, cherry/white ash/yellow-poplar, sweetgum, and magnolia
- Mixedwood—primarily loblolly and longleaf pine mixed with oak and hickory
- Natural softwood—primarily loblolly and longleaf pine
- Planted softwood—primarily loblolly and longleaf pine.

All counties within this province had some potential woody-biomass harvests. The densest concentration of harvesting occurred in the Gulf region within Louisiana, Alabama, the Florida panhandle, and southeastern Texas. In the Atlantic region, biomass harvests occurred mostly in eastern South Carolina and Virginia (fig. 11.1, 11.3a, b, and c;). This spatial pattern was driven by counties with >5,000 acres removal of logging residues compared to whole-tree harvests for all scenarios (fig. 11.4). Only three counties had >5,000 acres of whole-tree biomass harvest in ML 2040 and HH 2040 scenarios (fig. 11.4b and 11.4c). The greatest density of counties with young forests occurred mostly in the Gulf region in eastern Texas, southern Alabama, and eastern South Carolina, but few counties had >5,000 acres harvested under HH 2040 (fig. 11.5).

For logging residues, the primary forest change was a reduction under both 2040 scenarios from 1.12 million acres harvested under ML 2017. However, the predominant forest cover harvested changed under each scenario. Under ML 2017, logging residues were a byproduct primarily from mixedwood (31.0%), followed by natural softwood (29.5%) and upland hardwood (20.6%). However, under ML 2040, natural softwood produced 56.1% of logging residues, followed by upland hardwood (19.1%) and mixedwood (18.8%). Under HH 2040, planted softwood predominated (28.2%) in the percentage of logging residues, followed by upland hardwood (26.7%) and natural softwood (25.5%). Nearly half of all logging residues, however, were a byproduct of softwoods under all scenarios. Logging residues from diameter class 1 comprised 27.8% of harvested acres under ML 2017 but were the only source for logging residues under both 2040 scenarios. The remaining source for logging residues was thinning of diameter class 2 (70.2%)—mostly upland hardwoods. The greatest concentration of counties with >5,000 acres of harvested logging residues occurred primarily in southern Alabama and Florida and in eastern South Carolina (fig. 11.4). Fewer than five counties had >5,000 acres of whole-tree biomass harvest under HH 2040 (fig. 11.5c and 11.5d).

For whole-tree biomass, harvests occurred on approximately 740,000 acres under ML 2017, and declined by 56.1% and 61.2% under ML and HH 2040 scenarios, respectively (fig. 11.6). Under ML 2017, upland hardwoods (45.7%), followed by planted softwoods (32.5%) and lowland hardwoods (10.1%), produced this feedstock. This pattern was consistent under both 2040 scenarios, except mixedwood was third under HH 2040 (10.8%). Relatively few whole-tree biomass harvests occurred in natural softwood forests—only 3%–10% of harvested acres across scenarios. Whole-tree biomass was a byproduct of thinning diameter class 2 (4.9%) under ML 2017, but this feedstock increased to approximately 70% under both 2040 scenarios, mostly from thinning upland hardwoods. Clearcutting produced the most feedstock under ML 2017, again primarily from upland hardwoods. Spatially, the greatest density of whole-tree harvest was in southern Alabama, the Florida panhandle, and southern South Carolina (fig. 11.4), but under HH 2040, no counties had >5,000 acres harvested for whole-tree biomass.

Young forests were generated after clearcutting approximately 1 million potential acres under ML 2017, which also generating logging residues and whole-tree biomass. Harvested acres declined to approximately 670,000–680,000 acres under both 2040 scenarios. In ML 2017, potential acres to be clearcut were greatest in upland hardwood forested systems, and similar acreage was projected for natural and planted softwoods and mixedwood. As a result of clearcutting, acres in open-canopy cover increased in natural softwoods under ML 2040; acres in planted softwoods also increased under HH 2040, while declining approximately by half in the upland hardwoods. Spatial distribution of counties with >5,000 acres was similar to patterns for logging residues produced by clearcutting—mostly in Gulf areas in southern Louisiana, Mississippi, and Alabama, and on the Atlantic coast, mostly in southeastern South Carolina and West Virginia.

11.3.2.3 Biodiversity Effects—Provinces 231 and 232

Before discussing potential effects of biomass harvest on biodiversity, it is important to put results of the harvest scenarios in context of the southern landscape. For both Provinces 231 and 232 (fig. 11.1), less than 2% of the province's land area is potentially harvested by either whole-tree harvest or through the removal of forest residuals. Also in both provinces, logging residues were removed from approximately 1 million acres, and the whole-tree harvest was less than 1 million acres under ML 2017 that had the greatest potential harvested acres. Further, since acres of logging residuals are assumed to be a product of conventional harvest, it is not clear if potential harvests will have widespread, long-term effects on biodiversity in the S region. Given that harvested acres may be clumped in distribution, however, localized effects are possible. Additionally, as the most prominent result of biomass harvest will be changes in forest structure, it is important to understand how this may affect biodiversity in the S.

When comparing provinces, it appears that in general: (1) lowland hardwoods are projected to be the least affected by harvest in Province 232 but constitute a significant component of small-diameter whole-tree harvest in Province 231 (approximately 215,000 acres); (2) planted softwoods, natural softwoods, and upland hardwoods are projected to be the most affected by biomass harvest; (3) approximately 2.1 million acres of young, open forests are projected to be on the landscape under the ML 2017 scenario; (4) harvest of residuals may decline 51% between ML 2017 and both 2040 scenarios in Province 231; and (5) for Province 232, clearcutting is projected to be the most common harvest activity under ML 2017, with thinning as the dominant activity in both ML and HH 2040 scenarios. This summary is useful when considering potential biodiversity effects, given the initial caveat (biomass harvest in the context of total forest harvest in the S), which leads us to consider species and communities associated with upland forests more so than lowland forests.

One of the primary concerns associated with biomass harvest is removal of FWD and CWD due to the number of species dependent on, or associated with, these components of forest structure (see 11.1 Introduction). Removal of these materials may be prominent in both provinces because the primary sourcing feedstock is logging residues. However, it is not clear that the concern about forest structure loss extends as much to Province 232. The hot, humid conditions in the coastal plain of the southeastern United States lead to quick decomposition of downed wood. In fact, many species in the southeastern United States have shown minimal response to CWD (Mengak and Guynn 2003; McCay and Komoroski 2004; Davis, Castleberry, and Kilgo 2010b). Results from recent studies of operational biomass-production practices suggest minimal or short-term vertebrate species responses, potentially due to abundance of CWD retained on-site even after biomass harvests (Fritts 2014; Fritts, Moorman, et al. 2015; Fritts, Grodsky, et al. 2015; Fritts et al. 2016), which reflects recommendations commonly found in some biomass-harvesting guidelines (Perschel, Evans, and DeBonis 2012). Therefore, removal of CWD and FWD as part of biomass harvests may not be a strong driver of biodiversity response in these provinces.

Logging residues were a primary byproduct of planted and natural softwoods, but upland hardwoods and planted softwoods produced mostly whole-tree biomass. Lowland hardwoods are of conservation concern in the S, partly because of a perception that these forests are being extensively harvested for the wood-pellet market. As noted above, under all scenarios for Province 232 (Outer Coastal Plains), it appears that lowland hardwoods are minimally affected by biomass harvest using the parameters of this assessment, but this cover type is a primary source of whole-tree biomass harvests in Province 231 under all scenarios, meaning smaller-diameter classes are being harvested. However, the area that would be affected is, at most, approximately 215,000 hectares, which comprises less than 2% of a projected area of 12.15 million hectares of lowland hardwoods in the southeastern United States in 2010 (Wear and Greis 2012). Additionally, areas in bottomland hardwood forests remained relatively stable from 1970 to 1992, but slight declines in total acreage are expected between 1995 and 2040 (Wear and Greis 2002). Based on these scenarios, it does not appear that lowland hardwoods will be strongly or negatively affected by potential biomass harvest, although localized effects could be observed (see text box 11.5).

Under all scenarios, the amount of young forest created via biomass harvest is projected to decline dramatically between assessment periods, except for planted softwood in Province 232. Some of this change is due to potential conventional harvests of mature trees. However, the increasing amount of young forests generated from biomass harvests is a result of full-tree harvests. A suite of species requires early successional forests, and some of these species are in rapid decline in the eastern United States (see, e.g., King and Schlossberg 2014). There may be an influx of young forest conditions under

Text Box 11.5 | Case Study: Rafinesque's Big-Eared Bat—Rare Native

Rafinesque's big-eared bats (*Corynorhinus rafinesquii*) are a species of conservation concern across the southeastern United States. This species relies primarily on bottomland hardwood forests, roosting in tree cavities in larger hardwood trees, under bridges, and in buildings. Miller et al. (2011) estimated the potential roosting habitat for Rafinesque's big-eared bats by quantifying acres containing water tupelo (*Nyssa aquatic*) with greater than 50 cm diameter at breast height (dbh) based on U.S. Forest Service Forest Inventory and Analysis data. They found that there are approximately 308,000 hectares of bottomland hardwood forests with such trees in the southeastern United States (Miller et al. 2011). Given the relatively small area containing such potential roost trees, increased harvest of lowland hardwoods for biomass could have a localized, negative effect on this species if larger roost trees are removed or occupied habitat is harvested. However, given that most of the potential hardwood harvest for biomass is expected to be in smaller-diameter classes, it is not clear if such harvests will affect these more-mature lowland hardwood stands. Research is needed to further examine effects of potential biomass harvest in lowland hardwoods on the known distribution of this species and county-level (or more precise) potential harvest of lowland hardwoods for biomass.

the near-term ML 2017 scenario, but due to ForSEAM model assumptions, this pattern was not evident under both ML and HH 2040 scenarios. A decline in acres harvested by clearcutting, particularly in hardwood systems, may contribute to a strong trend of reduced oak regeneration, thereby changing forest composition and function in much of the eastern United States (McShea et al. 2007). Although the area affected by woody-biomass harvest appears relatively small compared to the total area of forested acres, even incremental changes in forest structure may have long-lasting effects on future forest composition.

Recently, an area of interest within the southern United States is creation and maintenance of open pine-canopy conditions (Greene et al. 2016). Historically, open-pine conditions on some site condition types were maintained by frequent fire, which suppressed hardwood encroachment, allowing herbaceous plant growth under a relatively open-pine canopy. Forest harvest can create these conditions in regenerating forests (see above) and also in older forest stands through thinning (Riffell et al. 2012). If there is a reduction in clearcut acres and thinning, as represented by changes from ML 2017 to both 2040 scenarios, biomass harvesting alone will likely not be able to help maintain open-pine conditions on the landscape (see text box 11.6). Therefore, planners need to consider the cumulative effects of conventional and biomass harvest when considering future distribution and amount of open-pine conditions in the southern landscape.

Text Box 11.6 | Case Study: Gopher Tortoise—Keystone Species

Based on the area of most change in Province 232, the gopher tortoise—a keystone species that is federally protected in the western portion of its range—is a species that could potentially be affected by biomass harvesting in this province. Gopher tortoises are associated with upland, sandy soils and require open-canopy pine forests with abundant herbaceous vegetation. Such open conditions can be created with clearcut harvests or thinning and can be maintained by prescribed fire and/or herbicide applications. The forest types associated with gopher tortoises in the assessment are natural and planted pine forests. The reduction in early successional forest stands and the relatively low level of potential thinning of natural and planted softwood stands under both ML and HH scenarios may result in less suitable habitat for gopher tortoises, assuming that other management activities do not ameliorate potential reductions. However, it must be recognized that the change would represent only a small portion of the projected occupied range for this species, which extends west from southern South Carolina and Florida through Georgia, Alabama, Mississippi, and eastern Louisiana. Also, a more precise spatial analysis is needed to understand location of potential harvest relative to appropriate soils (upland, deep sands) for gopher tortoises, which would help identify potential, localized effects of biomass harvest on gopher tortoises.

11.3.3 North Central Region

Overall, 19.4% of the total acres harvested for woody biomass occurred in the NC region under ML 2017, compared to 12% under both 2040 scenarios (fig. 11.2a). A total of 1.65 million acres were projected to be harvested under ML 2017, and acres declined 67.8% and 76.5% under ML and HH 2040 scenarios, respectively. Harvested acres were 27.2% lower under the HH 2040 scenario compared to the ML 2040 scenario. Logging residues and whole-tree biomass were produced from approximately the same amount of harvested acres under ML 2017 (approximately 800,000 acres), but logging residues were the primary feedstock under both ML 2040 and HH 2040, at 75.8% and 83.6%, respectively (fig. 11.2b and 11.2c). In this region, the assumption of ForSEAM was that the harvest method was 50% full-tree and 50% cut-to-length harvesting. Under the cut-to-length harvest method, trees are felled, delimbed, and bucked to length at the stump; then, logs are transported to landing (DOE 2016, p. 50). Residues stayed on the land, which also produced piles of residues (i.e., tops and limbs).

The densest concentration of counties with >5,000 acres of harvest would occur in northern Minnesota, Wisconsin, and the upper peninsula of Michigan (fig. 11.3). This area is encompassed by Province 212, Laurentian Mixed Forests (see text box 11.7; fig. 11.1). More than two-thirds of the counties within the NC region, however, had some forest woody-biomass harvest activity. Under HH 2040, only eight counties had >5,000 acres harvested (fig. 11.3c). It is worth noting that southern Missouri had eight counties under ML 2017 that experienced >5,000 acres harvested.

11.3.3.1 Province 212

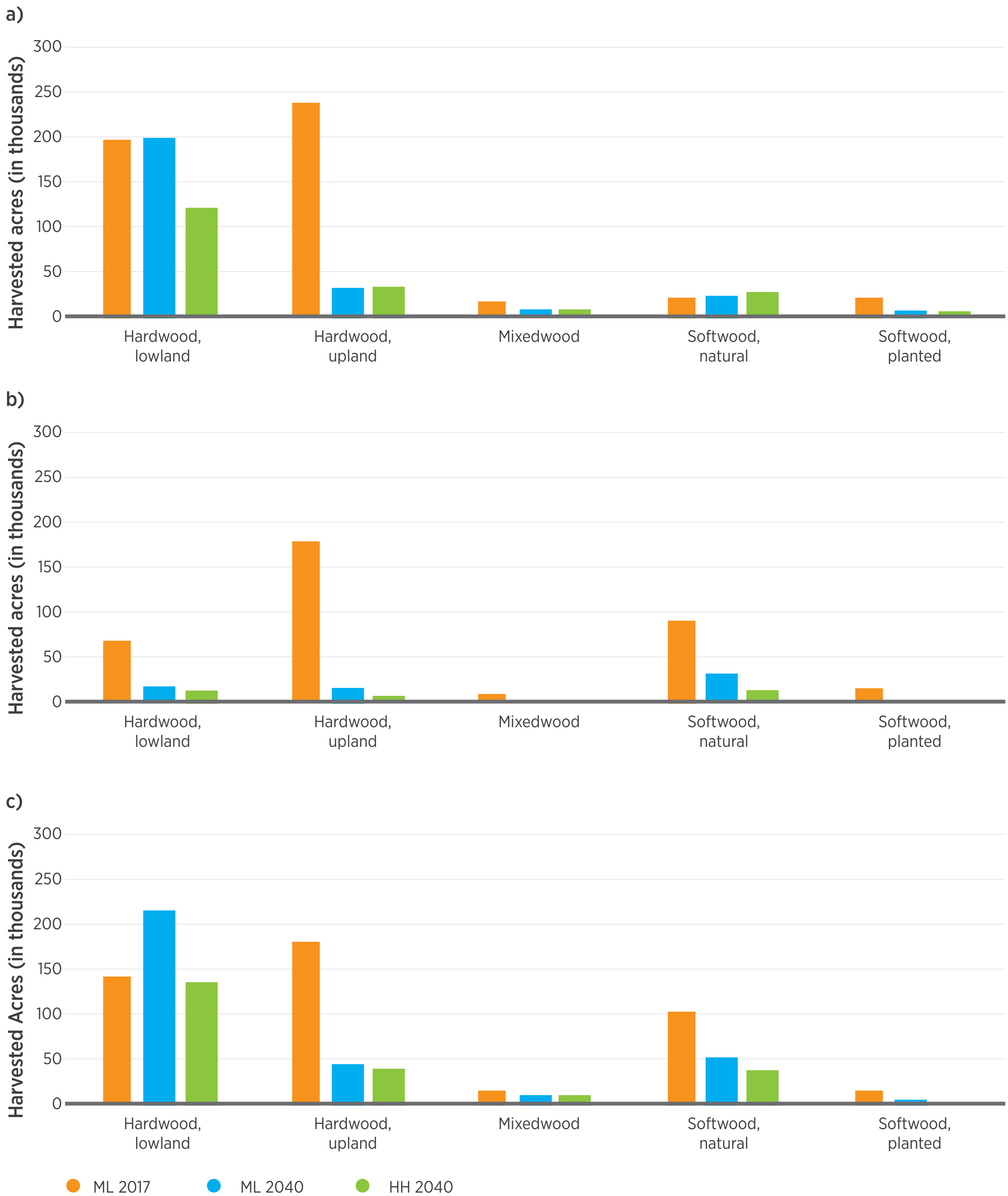
This province is 64.6 million acres, covering nearly 11% of the NC region. Under ML 2017, 838,080 acres were projected to be harvested, representing <1% of Province 212. The harvested land base for woody biomass declined by 61.0% and 73.7% under ML 2040 and HH 2040 scenarios, respectively; the HH 2040 scenario differed by 31.9% from the ML 2040 scenario. Under the ML 2017 scenario, counties with the greatest quantity of acres harvested were in northeast Minnesota and north-central Wisconsin, but under the HH 2040 scenario, only nine counties had >5,000 acres, and no counties had >10,000 acres (fig. 11.3).

Text Box 11.7 | Province 212: Laurentian Mixed Forest

This province is characterized by the following:

- Continental climatic regime with maritime influence along the Great Lakes; hilly landscapes with low relief and lakes, morainic hills, drumlins, eskers, outwash plains
- Ground continually snow-covered during the winter, with most precipitation occurring during summer
- Vegetation consisting of forests that are a transition between boreal and broadleaf deciduous zones
- Planted softwood—primarily red, jack, and white pine
- Natural softwood—primarily northern white cedar, balsam fir, tamarack, and black and white spruce
- Upland hardwoods—typically sugar maple-basswood mesic forests with red oak, American elm, red elm, green ash, and aspen-paper birch forests
- Lowland hardwoods—typically black ash with associated yellow birch, red maple, and beech
- Mixedwood—typically eastern white pine, northern red oak, and white ash mixed forests.

Figure 11.7 | Potential acres harvested by forest cover for the Laurentian Mixed Forest Province (212) by (a) logging residues, (b) whole-tree biomass feedstock, and (c) open forest canopy condition (i.e., young forest).



Logging residues were generated from 57.6% of potential harvested acres under ML 2017, and increased to 80.3% and 85.0% under ML and HH 2040 scenarios, respectively (fig. 11.7). Upland and lowland hardwood harvests produced 48.8% and 40.3% of this feedstock, respectively (89.1% combined), under ML 2017. However, under both ML and HH 2040 scenarios, lowland hardwoods were the predominant source of logging residues: 75.7% and 64.2%, respectively. Thinning of diameter class 2 produced 80.9% of logging residues under ML 2017, but only 1.2% and 1.4% under ML and HH 2040 scenarios, respectively. Instead, clearcutting of diameter class 1 produced 98.4% of logging residues under both 2040 scenarios. Harvest operation was full-tree conventional harvests for pulpwood under all scenarios. Only one county in northeastern Minnesota had >10,000 acres harvested under ML 2040; no counties under HH 2040 had >10,000 harvested acres. There were only four counties with >5,000 acres under the HH 2040 scenario—three in Wisconsin and one in Minnesota.

Whole-tree biomass was produced from harvest of upland hardwoods and natural softwoods at 49.9% and 25.0% of this feedstock, respectively, under ML 2017, but under the ML 2040 scenario, natural softwood produced 49.3% of feedstock, followed by lowland hardwood (25.7%) and upland hardwood (22.9%) (fig. 11.7). Under HH 2040, lowland hardwood comprised 39.2% of whole-tree biomass, followed by natural softwood (38.6%) and upland hardwood (20.3%). Under all scenarios, this feedstock was a byproduct of clearcutting diameter classes 2 and 3. The concentration of counties with >5,000-acre harvests was mainly in northern Minnesota under ML 2017; however, no counties had >5,000 acre harvests under HH 2040, and there was only one county in northern Minnesota that had >5,000 acres under the ML 2040 scenario (fig. 11.4).

Young forests were created through clearcutting on approximately half of the harvested acres under ML 2017, but accounted for >97% of total potential harvested acres under ML and HH 2040 scenarios. However, the potential harvested acres clearcut were 29.5% and 51.0% lower under ML and HH 2040 scenarios, respectively, than under ML 2017. Acres in young forests were 31.0% lower in the HH 2040 scenario than the ML 2040 scenario (fig. 11.7). Under ML 2017, clearcutting of upland and lowland hardwoods—and to some degree, natural softwoods—generated young forests, but under both 2040 scenarios, clearcutting of lowland hardwoods generated nearly two-thirds of young forest acreage. Whole-tree biomass harvesting accounted for 79.4% of clearcutting activities under ML 2017, but only 20.2% and 15.1% under ML and HH 2040 scenarios, respectively. Clearcutting associated with conventional harvests of lowland hardwood sawlogs produced most of the young forests under the 2040 scenarios. Only seven counties had >5,000 acres of potential harvest under the HH 2040 scenario (four in Minnesota and three in Wisconsin); there were no counties with >10,000 acres harvested under all scenarios (fig. 11.5).

11.3.3.2 Biodiversity Effects—Province 212

The primary forest change influencing biodiversity in this ecoregion is removal of logging residues from the forest floor under expanded biomass demand, as >80% of potential acres harvested produced residues under the 2040 scenarios. This forest change may have greater effects on biodiversity of species that rely on this material (see text box 11.8). Most of this feedstock was produced from harvests in lowland and upland hardwoods, especially lowland hardwoods under the 2040 scenarios. In the near term (ML 2017), logging residues were generated mostly through thinning harvests, but they were generated mostly through clearcut harvests under both 2040 scenarios. Upland and lowland hardwoods and natural softwoods generated most of the potential whole-tree harvests through clearcuts under ML 2017, but clearcuts of lowland hardwood sawlogs created most of the young forests under the 2040 scenarios. The total acres of lowland hardwoods clearcut was approximately the same between ML 2017 and HH 2040, but increased about half under ML 2040, meaning an influx of young forests in lowland hardwoods across all scenarios. However, lowland hardwoods on public lands in this region are not typically clearcut, so the effect on biodiversity is unclear as much of the land in Province 212 consists of public lands (see text box 11.9). The densest concentration of counties with >5,000 acres of total potential harvest occurred in northern areas of the province with a high proportion of public lands. With the reduction of potential harvest acres in 2040, the concentration of higher-intensity harvests was limited to northeast Minnesota and Wisconsin.

Text Box 11.8 | Case Study: American Marten—Species of Cultural Importance

Forest structural complexity is a critical habitat component for the American marten (*Martes Americana*). Dead wood, such as large snags, fallen trees, stumps and root mounts, and residual piles, provides den and resting sites; cover from fishers, lynx, and bobcat predators while traveling; forage areas for preferred small rodents, squirrels, and hares that also use residual piles; and access points to get to ground surface for foraging during snow cover (Corn and Raphael 1992). Marten have been found to prefer mature northern forest communities and avoid aspen-dominated systems, swamp conifer, and nonforested areas (Wright 1999). Within mature forests, tree-species composition is less important than the volume of downed woody debris and canopy closure (Buskirk and Ruggiero 1994; Buskirk 1994; Chapen, Harrison, and Phillips 1997). These two factors are often listed as major threats to marten viability in an area. Marten avoid recent clearcuts, and extensive clearcutting may lower local abundance (Hargis and McCullough 1984; Potvin and Breton 1997). Marten densities were found to be positively correlated with prey abundance in Maine (Soutiere 1979). Potential effects to marten in this province may be greater under the ML 2017 scenario, with more acres harvested producing logging residues in combination with increased whole-tree harvests by clearcutting in upland hardwoods. The loss of forest structure in combination with opening the canopy in preferred habitats may negatively affect this species. Furthermore, whole-tree harvesting in smaller-diameter trees rather than in the preferred mature forests may negatively affect this species in the long term if management practices result in significant reduction of mature forest. The southernmost distributional range of the American marten extends into the northern areas of the Northeast (NE) and Pacific Northwest (PNW) Forest Sustainable and Economic Analysis Model (ForSEAM) regions, so forest woody-biomass harvesting may affect this species in these regions as well and should be evaluated.

Text Box 11.9 | Case Study: Golden-Winged Warbler—Species of Concern

Young forests are an important habitat for the golden-winged warbler (*Vermivora chrysoptera*), a migratory bird found throughout the north-central and eastern United States. The golden-winged warbler population has declined range-wide, and the warbler is currently being considered for listing under the Endangered Species Act (Pruss et al. 2014). This decline has been attributed to loss of preferred breeding habitat caused by maturing forests. Regenerating upland and lowland habitat is used for breeding as dense foliage and shrubs provide cover for ground nests. Scattered trees or edges of forests provide singing perches. Dense foliage also lowers negative interactions with blue-winged warblers (*Vermivora cyanoptera*) and cowbirds (Molothrus spp.) (Pruss et al. 2014). Given the influx of young forests expected from clearcuts of mature lowland hardwoods under both 2040 scenarios, and from the same relative acreage in 2017 from whole-tree biomass harvesting, there may be opportunities in this ecoregion to contribute to the conservation of this warbler and other species that rely on young forests. Other birds associated with young forests showing range-wide declines are the chestnut-sided warbler (*Setophaga pensylvanica*), Bell's vireo (*Vireo belli*), alder flycatcher (*Empidonax alnorum*), American redstart (*Setophaga ruticilla*), and blue-winged warbler (*Vermivora cyanoptera*).

11.3.4 Northeast

Overall, 14.6% of potential total acres harvested for forest woody-biomass occurred in the NE under ML 2017 compared to 8.6% and 11.5% under ML 2040 and HH 2040 scenarios, respectively (fig. 11.2a). The total area harvested was 1.24 million acres under ML 2017. The total declined approximately 70% under both 2040 scenarios; projections for HH 2040 harvested acres were 7.3% greater than the projections for the ML 2040 scenario. Whole-tree biomass was harvested from 759,000 acres, while logging residues were harvested from 483,000 acres under ML 2017. However, logging residues dominated feedstock under both 2040 scenarios: 75.9% under ML 2040 and 86.1% under HH 2040 (fig. 11.2b and 11.2c). In this region, the assumption of ForSEAM was that the harvest method consisted of 100% full-tree harvest type, meaning felled trees were taken to the landing to be processed, and the full trees or remaining waste after processing could be chipped.

The densest concentration of counties with >5,000 acres of total potential harvest occurred in Maine and several counties in northern New York under all scenarios (fig. 11.3). However, almost all counties

Text Box 11.10 | Province 211: Northeastern Mixed Forest Province

This province is characterized by the following:

- Modified continental climatic regime with maritime influence along the Atlantic Ocean
- Summer peaks in annual precipitation, which is otherwise equally distributed throughout the year; winters with continual ground snow cover
- Vegetation transitions between boreal spruce-fir in the north and broadleaf deciduous forests to the south
- Planted softwood—primarily Eastern white and red pine
- Natural softwood—primarily red spruce/balsam fir, balsam fir, and black spruce
- Mixedwood—primarily Eastern white pine/northern red oak/white ash
- Upland hardwood—primarily aspen and paper birch
- Lowland hardwood—primarily sugar maple/beech/yellow birch and hard maple/basswood.

within the region would have some woody-biomass harvests. Under ML 2040, few counties had >5,000 acres projected to be harvested, mostly located in southern Maine; however, under HH 2040, three counties had >10,000 acres and four counties had >5,000 acres projected to be harvested (fig. 11.3c). Province 211 and M211 encompassed greatest concentration of counties with >5,000 acres of harvesting potential (see text boxes 11.10 and 11.11; fig. 11.1). Because the forest-change trends were similar between provinces, we reported combined total acres, but separated the provinces graphically (fig. 11.8).

11.3.4.1 Province 211 and M211

Province M211 is approximately 24.1 million acres, covering 10.6% of the NE Region, and Province 211 is approximately 33.7 million acres, covering 14.8% of the NE. Under the ML 2017 scenario, 216,290 acres were harvested in M212, and 277,720 acres were harvested in Province 212, representing about 2% and <1%, respectively. The harvested land base for woody biomass declined by 75.6% and 66.0% under ML 2040 and HH 2040 scenarios. The harvested land base is 39.6% higher under HH 2040 than under ML 2040. Under the 2040 scenarios, few counties had >5,000 acres harvested, mostly located in western New York and the southeastern corner of Maine.

Logging residues were the major feedstock only under ML and HH 2040 scenarios, comprising 72.1% and 87.7% of the total harvest, respectively. Logging residues were 46.0% lower under ML 2040 than under the ML 2017 scenario but were greater under HH 2040 than ML 2040. Much of this difference was due to greater logging residues produced after harvesting lowland hardwoods, which comprised 59.7% and 73.3% of harvested acres under ML 2040 and HH 2040, respectively (fig. 11.8a). Lowland hardwoods comprised 41.9% of harvested acres under ML 2017. Natural softwoods were the second largest forest type producing residues: 25.1% of acres harvested under ML 2017, 20.5% under ML 2040, and 16.2% under HH 2040 (fig. 11.8a). Thinning of diameter class 2 produced 66.6% of logging residues under ML 2017, but no logging residues were produced from thinning under either 2040 scenario. Instead, clearcutting of diameter class 1 produced 100% of logging residues. The harvest method was full-tree for pulpwood under ML 2017. The greatest concentration of counties with >5,000 acres was located in southeastern Maine and a few counties in upper New York.

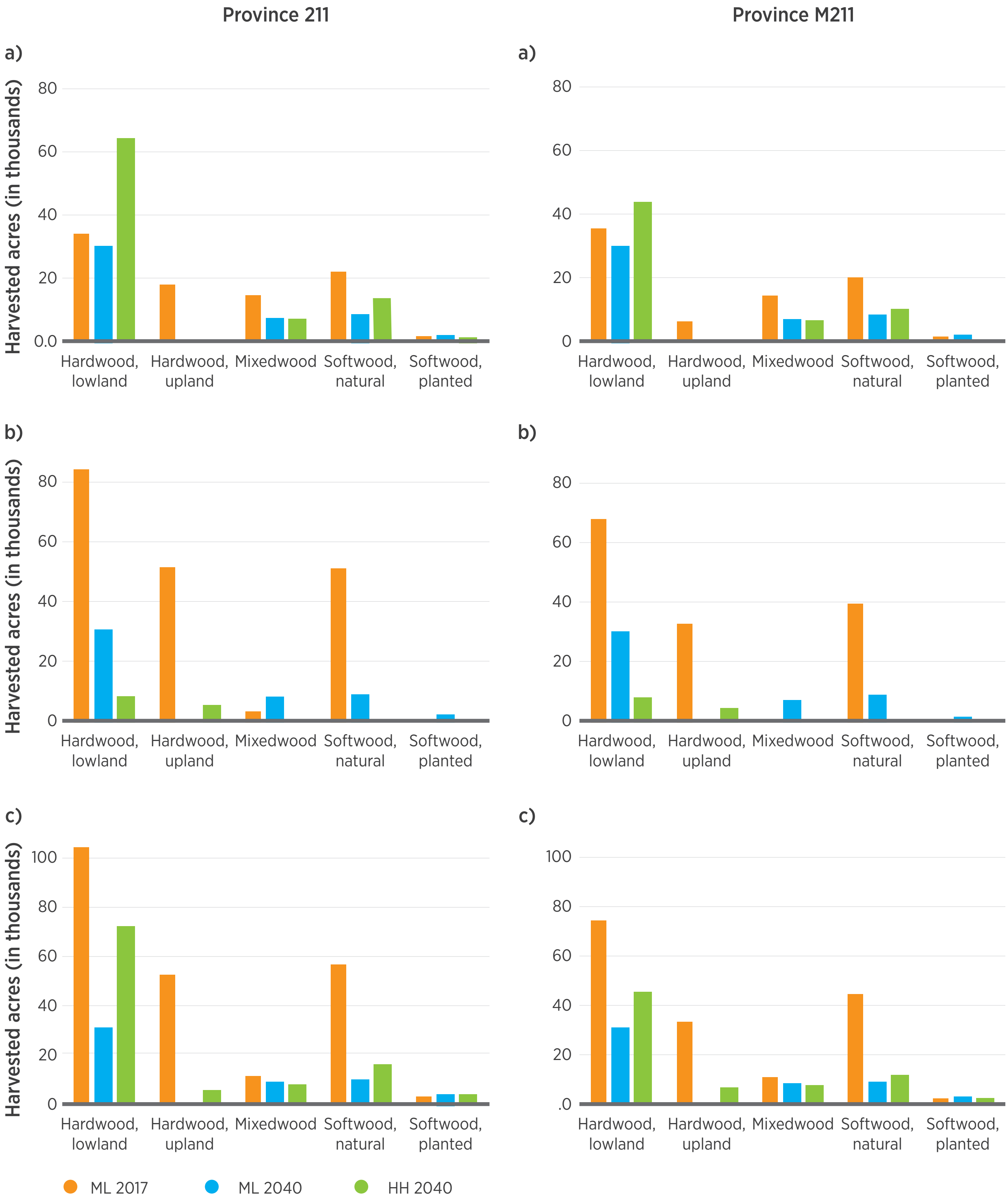
Whole-tree biomass was the primary feedstock in these provinces under ML 2017, comprising 66.5% of potential acres harvested, almost twice as much as logging residues (fig. 11.8b). However, this feedstock declined >90% under both 2040 scenarios from ML 2017. Lowland hardwoods produced 46.2% of the feedstock under ML 2017, followed by natural softwood (27.4%) and upland hardwoods (25.4%).

Text Box 11.11 | Province M211: Adirondack–New England Mixed Forest–Conifer Forest–Alpine Meadow Province

This province is characterized by the following:

- Continental climate regime with long winters and warm summers and annual precipitation evenly distributed across the year, distinguishing this climate from Province 211
- Mountainous landscape with dissected plateaus
- Vegetation transitions between boreal spruce-fir in the north and broadleaf deciduous forests in the south
- Planted softwood—primarily Eastern white and red pine
- Natural softwood—primarily red spruce, balsam fir, and black spruce
- Upland hardwoods—primarily aspen and paper birch
- Lowland hardwoods—primarily sugar maple/beech/yellow birch and red maple.

Figure 11.8 | Potential acres harvested by forest cover for Northeastern Mixed Forest Province (211; *left*) and Adirondack–New England Mixed Forest–Conifer Forest–Alpine Meadow Province (M211; *right*) by (a) logging residues, (b) whole-tree biomass feedstock, and (c) open forest canopy condition (i.e., young forest); note the difference in scale for young forests.



Upland and lowland hardwoods produced >90% of whole-tree biomass under both 2040 scenarios. Under ML 2017, whole-tree biomass was a byproduct of clearcutting diameter classes 2 and 3, but in the 2040 scenarios, only clearcutting diameter class 2 provided this feedstock. Much of this feedstock was from harvests in counties in western Maine, which is Province M212; these counties had >30,000 potential acres harvested under ML 2017. No counties had this level of harvest under the 2040 scenarios.

Text Box 11.12 | Case Study: American Woodcock—Recreational Species

American woodcock (*Scolopax minor*; hereafter, woodcock) breeds in northern states and provinces across eastern North America and winters from the Mid-Atlantic states south to the Gulf Coast, and west as far as eastern Texas. They use young hardwood forests as display areas and dense deciduous or mixed forests with closed canopy as diurnal feeding cover, and they nest in young open-canopy deciduous forests with well-drained soils (Keppie and Whiting 1994; Straw et al. 1994). Because of reduced availability of young forests in much of the eastern United States (King and Schlossberg 2014), woodcock populations have experienced significant declines since surveys were first implemented in the mid-1960s and thus is of conservation interest (Kelley et al. 2008). A conservation plan (Kelley, Williamson, and Cooper 2008) has suggested creating 20.8 million acres of new woodcock habitat if woodcock densities are to return to those observed during the early 1970s. Thus, increased harvest for woody biomass in the NE region is likely to enhance suitable habitat conditions for this species. Suitable habitat for woodcock is likely to be greater under ML 2017 than either ML or HH 2040 scenarios due to lower potential acres harvested in under these scenarios.

Text Box 11.13 | Case Study: Canada Lynx—Rare Native

Canada lynx (*Lynx Canadensis*; hereafter, lynx) is a federally threatened species; Maine is the only state in the northeastern United States known to support a resident population (Vashon et al. 2008a). It is a specialist predator of snowshoe hare (*Lepus americanus*) but also seeks alternative prey, such as red squirrels (*Tamiasciurus hudsonicus*) or Tetraonids (grouse) (Hoving et al. 2004). At the stand scale, prey abundance is a driving factor in lynx habitat selection. Male and female lynx in Maine strongly choose conifer-dominated sapling forests that contain high winter-hare densities and intermediate cover for hares (Fuller et al. 2007; Vashon et al. 2008a). Lynx selected tall (4.4–7.3 m) regenerating clearcuts (11–26-year post-harvest) and established partially harvested stands (11–21-year post-harvest) and selected against short (3.4–4.3 m) regenerating clearcuts, recent partially harvested stands (1–10-year), mature second-growth stands (>40-year), and roads and their edges (30 m on either side of roads) (Fuller, Harrison, and Vashon 2007). Vashon et al. (2008b), therefore, suggested that a mosaic of different-aged conifer stands would facilitate maintaining a component of regenerating conifer-dominated forest on the landscape. Lynx den sites in Maine were found primarily within conifer-dominated sapling and seedling stands, although lynx also did use dens in mature stands and in deciduous stands (Organ et al. 2008). However, coarse woody debris was not a useful predictor of lynx den-site selection despite its abundance. Rather, the combination of tip-up mounds of blown-down trees and visual obscurity from dense vegetation represented the within-stand characteristic predictive of lynx den sites (Organ et al. 2008). The authors recommended that managers in the northeast United States not focus on den habitat at the stand level. Similar to woodcock, potential suitable habitat for lynx is likely to be greater in the near term (ML 2017) rather than in the two scenarios for 2040 due to the lower potential acres harvested in the later time period.

Young forests were created through clearcutting from 77.7% of harvested acres under ML 2017; all acres under 2040 scenarios were clear-cut producing young forests. The harvested land base declined by 68.6% and 56.2% under ML 2040 and HH 2040, respectively. Young forests increased 39.6% under HH 2040 from ML 2040 levels (fig. 11.8c). Clearcutting of upland and lowland hardwoods accounted for >68% of the acres to be harvest under all scenarios. For HH2040, lowland hardwoods accounted for 69.6% of acres harvested. Given almost all acres were clearcut in these provinces, the location of large amounts of young forests tracked whole-tree biomass trends.

11.3.4.2 Biodiversity Effects—Provinces 211 and M211

The major forest change in the near term (ML 2017) was a major influx of young forests in the near term (ML 2017) from an increase in whole-tree biomass harvests through clearcutting smaller-diameter trees (see text boxes 11.12 and 11.13). The forest types contributing most to this feedstock were lowland and upland hardwoods, and natural softwoods of balsam fir and black and red spruce. Upland hardwoods were aspen and paper birch, and lowland hardwoods were primarily sugar maple/beech/yellow birch. Under the 2040 scenarios, the major feedstock switched to logging residues primarily from clearcutting mature, lowland hardwoods (diameter class 1). However, it is important to note that the land base with potential harvests declined three-quarters from ML 2017 to both 2040 scenarios, but the concentration of higher-intensity harvests remained in southern Maine and northwest New York. From a biodiversity perspective, Province M211 has some unique specialist species compared to Province 212 due to the alpine tundra such as long-tailed shrew, boreal (southern) redback vole (*Clethrionomys gapperi*), gray-cheeked thrush, and spruce grouse. Other species worth mentioning due to importance of structure or early successional forests are northern bog lemming (*Synaptomys borealis*) and New England cottontail (*Sylvilagus transitionalis*).

11.3.5 Pacific Northwest Region

Overall, 8.5% of potential total acres harvested for woody biomass occurred in the PNW region under ML 2017, compared to 12.6% and 11.6% under ML and HH 2040 scenarios, respectively (fig. 11.2a). Although proportion of total harvested acres increased in the PNW relative to other regions under both 2040 scenarios, total harvested acres in the PNW was lower by approximately 27.4% and 48.0% under ML and HH 2040 scenarios, respectively, from 720,253 acres under the ML 2017 scenario. The difference in potential harvested acres between ML 2040 and HH 2040 was 28.5%. Whole-tree biomass was the predominate feedstock harvested under ML 2017 and ML 2040: 457,676 acres compared to 262,577 acres producing logging residues; however, logging residues dominated feedstock under HH 2040, at 69.5% (fig. 11.2). In this region, the assumption of ForSEAM was that harvest method consisted of 100% full-tree harvest type, meaning no residues remained on the land except for any breakage that occurred during transfer to the landing.

The greatest concentration of counties with >5,000 acres of total potential harvest occurred in northern California, southwest Oregon, and western Washington (fig. 11.3). Many counties in southern California, Washington, and eastern Oregon had no potential harvests. The concentration of counties with >5,000 acres producing logging residues remained relatively consistent across scenarios, but the concentration of counties with >5,000 acres of whole-tree biomass harvests declined to four counties under HH 2040. Counties with >10,000 acres of young forests created through clearcutting were limited to six counties along California-Oregon state lines and several counties in northern Washington under ML 2017. This concentration of counties remained fairly consistent across scenarios (fig. 11.5). Provinces M261, M242 and 242 encompassed the greatest concentration of counties with total woody-biomass harvesting (see text boxes 11.14 and 11.15; fig. 11.1). Because Province 242 is narrower than many county boundaries, and we used county center points to designate the province in which each county was located, it is difficult to determine whether forest change is indicative for this province or an artifact of scale and methodology used. We therefore combined Province 242 with M242 into a Cascade province in the results. In addition, the counties in eastern Washington are encompassed under M333 (see the IW region).

Text Box 11.14 | Province M261: Sierran Steppe–Mixed Forest–Coniferous Forest–Alpine Meadow Province

This province is characterized by the following:

- Mountainous landscape with steep slopes crossed by many valleys with steep gradients
- Precipitation strongly influenced by altitude and direction of mountain ranges; hot and dry summers with most precipitation occurring in winter as snow
- Elevation-delineated vegetation with conifer and shrub associations at low elevations
- Higher elevations dominated by digger pine and blue oak; on western slopes, ponderosa pine, Jeffrey pine, Douglas-fir, sugar pine, white fir, and red fir predominate; on eastern slopes, Jeffrey pine replaces ponderosa pine and sagebrush-pinyon forest replace pine forests
- Lowland hardwoods—primarily red alder and Pacific madrone
- Upland hardwoods—primarily California black oak, Canyon live oak, and Oregon white oak
- Natural softwoods—primarily Ponderosa pine, white fir, lodgepole pine, and western juniper
- Planted softwoods—primarily Douglas-fir, Ponderosa pine, Jeffrey pine, and incense-cedar.

Text Box 11.15 | Provinces 242 (Pacific Lowland Mixed Forest Province) and M242 (Cascade Mixed Forest–Coniferous Forest–Alpine Meadow Province)

Both provinces are characterized by the following:

- Mild, modified marine climate with M242 having areas of cold-dry climate
- Province 242 occupying a north-south depression between the coastal and interior Cascade Mountains, characterized by level plains to low mountains with much of the natural forests replaced by agriculture
- Forests of western red cedar, western hemlock, and Douglas-fir; in the valleys, hardwoods of big-leaf maple, Oregon ash, and black cottonwood; prairies supporting Oregon white oak and Pacific madrone
- In Province M242, steep, rugged mountains along coast and several high-elevation peaks of volcanic origin with strong relief to foothills and plateaus
- Primarily montane vegetation, but at lowest elevations Douglas-fir predominates, but also western red cedar, western hemlock, grand fir, silver fir, Sitka spruce, and Alaska-cedar
- Ponderosa pine found along dry eastern slopes of the Cascades
- Lowland hardwoods—typically red alder and bigleaf maple
- Upland hardwoods—primarily Oregon white oak and paper birch
- Natural softwoods—primarily Douglas-fir, ponderosa pine, western hemlock, and white fir
- Planted softwoods—primarily Douglas-fir and ponderosa pine.

11.3.5.1 Province M261: Sierran Steppe-Mixed Forest-Coniferous Forest-Alpine Meadow

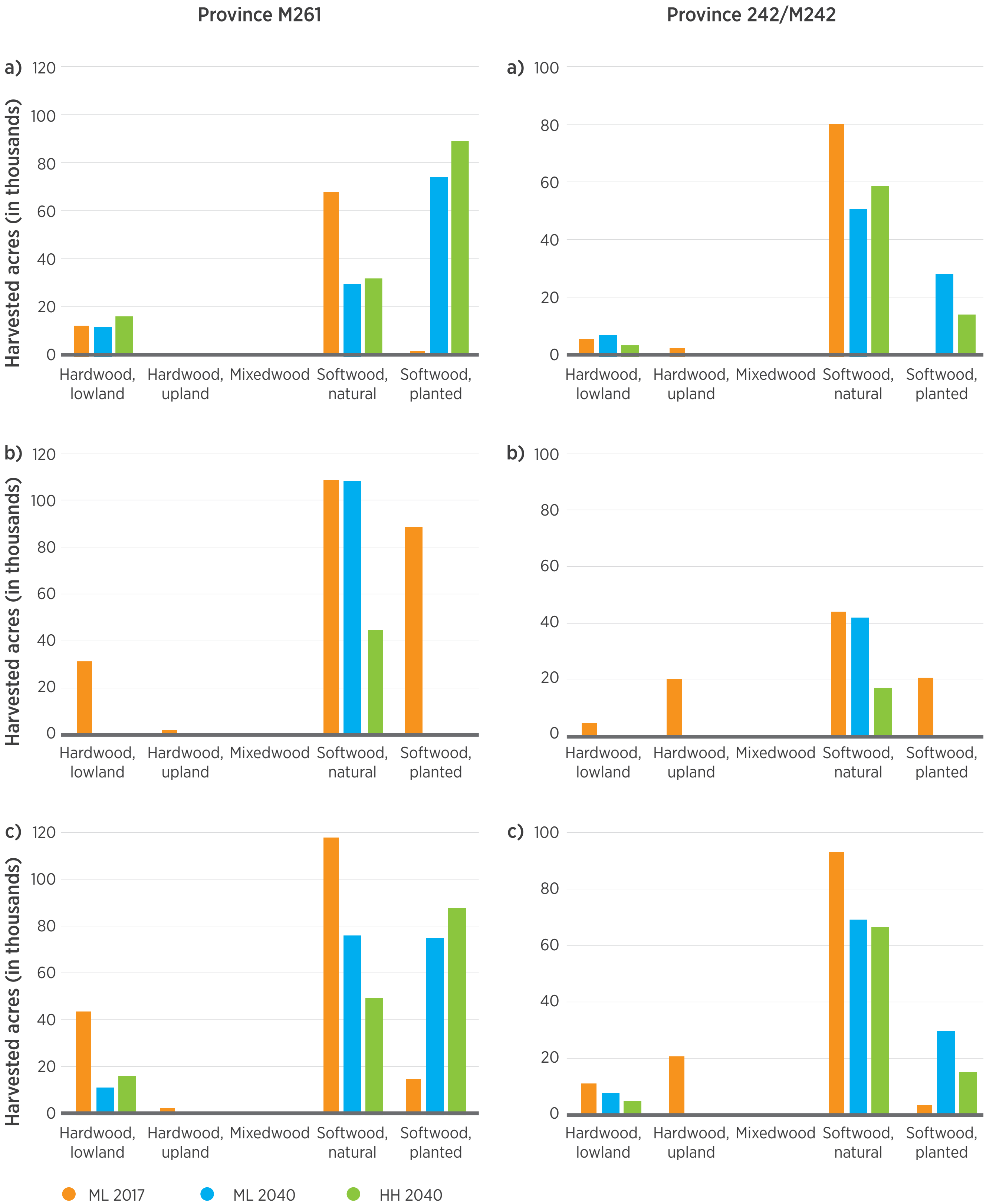
This province covers 43.0 million ac (21.1% of the PNW Region). Under ML 2017, 176,895 ac were harvested, representing less than 1% of Province M261. The projected harvested land base for wood biomass declined by 27.7% and 46.9% under 2040 baseline and high-yield scenarios; HH 2040 scenario decreased 26.5% from ML 2040 levels. Harvests with logging residues were approximately half of the feedstock under ML 2017, and increased to 67.7% and 81.6% under ML 2040 and HH 2040 with acres harvested ranging from 886,548 to 766,769, respectively. Under ML 2017, counties with the greatest potential acres harvested were in northern California and in southern Oregon, but under HH 2040, only three counties had >5,000 acres (fig. 11.3).

Logging residues were primarily a byproduct of natural softwood harvests, 90.1% under ML 2017, and 58.5% and 76.8% under ML and HH 2040 scenarios, respectively (fig. 11.9). Planted softwood harvests became the more prominent source of logging residues under ML and HH 2040 scenarios—33.1% and 18.3%, respectively, compared to only 0.4% under ML 2017. Nearly all logging residues were produced from clearcutting diameter class 1, natural softwoods under ML 2017 (86.8%). Logging residues from thinning operations (diameter class 2) contributed 6.6% under ML 2017 but provided no feedstock under ML and HH 2040 scenarios. Counties with >5,000 acres remained the same across all scenarios (fig. 11.4).

Whole-tree biomass was produced from harvest of natural softwoods (43.9%), planted softwoods (20.3%), upland hardwoods (19.6%), and lowland hardwoods (4.4%) under ML 2017, but under both 2040 scenarios, whole-tree biomass was only produced from harvests of natural softwoods. In addition, acres harvested declined by approximately half under ML 2040 and by 80.4% under HH 2040. Under all scenarios, approximately half of this feedstock was a byproduct of clearcutting diameter class 2, while the remaining half was a byproduct of thinning diameter class 2 operations. Counties with >5,000-acre potential harvests were mainly concentrated in southern Oregon in all scenarios, but only two counties under HH 2040 had greater harvesting commensurate with reduced total acres harvested (fig. 11.4).

Young forests were created by clearcutting 70.4% of the harvested acres under ML 2017, and this land base declined by 16.4% and 32.6% under ML 2040 and HH 2040, respectively. Between the 2040 scenarios, acres in young forests declined by 31.0% under HH 2040. Under ML 2017, clearcutting of natural softwoods was the primary source of young forests, but young forests were also created through clearcutting of upland and lowland hardwoods and, to some degree, planted softwoods (fig. 11.9c). But under ML and HH 2040 scenarios, young forests were created almost entirely through clearcutting of natural softwoods: 65.4% and 78.5%, respectively. Planted softwoods and lowland hardwoods also contributed to a much lesser degree. Spatially, counties with >5,000 potential acres harvested were located along California's and Oregon's borders under all scenarios.

Figure 11.9 | Potential acres harvested by forest cover for Province M261 (*left*) and Provinces 242/M242 (*right*) within the Pacific Northwest region by (a) logging residue feedstock, (b) whole-tree biomass feedstock, and (c) open forest canopy condition (i.e., young forest); note the different scales for each province.



11.3.5.2 Provinces 242 and M242: Cascade Provinces

These provinces cover 42.8 million acres, which is 21.0% of the PNW region. Under ML 2017, 311,083 potential acres were harvested, representing <1% of the province. Acres harvested for woody biomass were 28.3% and 42.3% lower under ML and HH 2040 than under ML 2017; HH 2040 was 19.6% lower than ML 2040. Whole-tree biomass harvests were the predominate feedstock under ML 2017 (74.2%), but harvests producing logging residues comprised greatest percentages under ML and HH 2040 scenarios (48.7% and 75.2% from 223,100 and 179,370 acres, respectively). Most counties in these provinces had >5,000 potential acres harvested, mostly concentrated in southern Oregon and northern Washington across all scenarios (fig. 11.3).

Similar to Province M261, in Provinces 242 and M242, logging residues were primarily a byproduct of natural softwood harvests (83.3% under ML 2017), but only represented 25.5% and 22.3% under ML and HH 2040 scenarios (fig. 11.9). Under ML and HH 2040 scenarios, two-thirds of logging residues were produced from planted softwood—64.6% and 65.5%, respectively—compared to only 2.0% under ML 2017. Eighty-three percent of clearcut acres generated logging residues from natural softwoods of diameter class 1 under ML 2017, but natural softwoods generated only 25.5% and 22.9% of logging residues under ML and HH 2040 scenarios, respectively. Instead, logging residues from planted softwood diameter class 1 generated approximately two-thirds of residues under both 2040 scenarios. Logging residues from thinning operations (diameter class 2) contributed 2.9% of total logging residues under ML 2017, but provided no feedstock under ML and HH 2040 scenarios. Counties with >5,000 potential acres harvested were concentrated in central Washington under ML 2017, but shifted to western Washington under both 2040 scenarios (fig. 11.4).

Whole-tree biomass was produced from potential harvests of natural softwoods (47.1%), planted softwoods (38.3%), upland hardwoods (13.6%), and lowland hardwoods (1.0%) under ML 2017; however, under both 2040 scenarios, >99% of whole-tree biomass was produced from harvests of natural softwoods. Similar to Province M261, in Provinces 242 and M242, acres harvested declined by approximately half under ML 2040 and by 80.7% under HH 2040. Under all scenarios, nearly half of whole-tree biomass feedstock was a byproduct of clearcutting diameter class 2, while the remaining half was a byproduct of thinning diameter class 2 operations. Counties with >5,000-acre potential harvests were concentrated throughout western Washington and Oregon under ML 2017, but these areas of high potential harvest were limited to a few counties in southern Oregon under both 2040 scenarios (fig. 11.4).

Young forests were generated after clearcutting 57.0% of harvested acres under ML 2017, and clearcutting 72.0% and 85.8% under ML and HH 2040 scenarios, respectively. Although the percentage of acres was greater than ML 2017 under both 2040 scenarios, total harvested acres were lower under ML 2040 and HH 2040 by 9.3% and 13.2%, respectively. Total acres of young forests differed by only 4.2% between ML and HH 2040 scenarios. Under ML 2017, clearcuts of natural softwoods generated the majority acres of young forest acres (66.8%), followed by lowland hardwoods (24.2%; fig. 11.9c). However, under ML 2040 and HH 2040 scenarios, young forests were created almost entirely after clearcutting natural and planted softwoods—92.8% and 89.7%, respectively. Few acres of lowland hardwoods were clearcut under both 2040 scenarios. Counties with >5,000 acres were concentrated in western Washington and southwest Oregon under all scenarios (fig. 11.5).

11.3.5.3 Biodiversity Effects—Provinces M242/242 and 261

When comparing provinces in the near term (ML 2017), logging residues were the major feedstock in northern California (M261), while whole-tree harvests were the major feedstock in western Washington and Oregon. However, under both 2040 scenarios, logging residues were the primary feedstock in both provinces. Natural softwoods produced the majority of each feedstock, mostly through clearcutting mature forests (diameter class 1), but in California, nearly half of residues were generated after clearcutting smaller-diameter trees. Natural softwoods were primarily Douglas-fir and ponderosa pine. To a much lesser degree, clearcutting smaller-diameter trees of upland and lowland hardwoods contributed to forest change under the ML 2017 scenario. Planted softwoods were the primary source of whole-tree biomass in the near term (ML 2017). Under all scenarios, potential acres for harvesting woody biomass comprise a small percentage of forests and decline under both 2040 scenarios, so it is unclear the effect these added harvests, especially whole-tree harvests, will have on biodiversity (see text box 11.16).

Text Box 11.16 | Case Study: Northern Flying Squirrel—Keystone Species

The Northern flying squirrel (*Glaucomys sabrinus*) is a forest-dwelling, arboreal rodent that inhabits boreal conifer and mixed forests with old-growth elements, such as substantial ground cover (Smith et al. 2005). This rodent travels by gliding and spends a lot of time on the forest floor foraging on fungus, lichens, and moss, which depend on an abundance of dead and downed wood, especially in moist, organic soils typical in older forests of western Washington and Oregon (Carey 1995; Weigl 2007). Conservation for this species focuses around its obligate symbiotic association with forest fungi (truffles) in which it feeds upon fruiting bodies and spreads mycorrhizal fungus through excreting pores. These fungi contribute to nutrient and water uptake of forests. Early successional stands have lower numbers of fungi, so large-scale clearcutting can be a threat to this species, especially in southern margins of its range, such as the Sierra Nevada, Rocky, and Appalachian mountains (Weigl 2007). The potential whole-tree harvests of conifers, especially in the near term, through clearcutting may affect the conservation of this species, but the effect is uncertain, given the degree of other aspects influencing their conservation, such as competition from the Southern flying squirrel (Weigl 2007). This squirrel is also an important prey species for the federally endangered spotted owl. Given the small contribution to national, potential woody-biomass harvests and the small potential area of lands with whole-tree harvests in the scenarios, the direct effects of woody-biomass removal on this squirrel are uncertain. However, their significance to forest-system productivity through their link with fungi and other trophic levels specific to the Pacific Northwest (PNW) should be considered. The importance of old growth versus successional forests to rare species in the PNW is often debated (Lehmkuhl et al. 2006). Whole-tree woody-biomass harvests may influence the canopy that this species requires, but removing residues may also lower the quality of habitat due to less dead and downed material that harbor fungus and lichen. Fungus and lichen are some of the most diverse communities associated with dead and downed wood. The southern part of this squirrel's distributional range covers the Inland West, North Central, and Northeast, which also have increases in whole-tree harvests for biomass, which increases stressors in the southern part of this species range.

Forest structure created by dead and downed wood is viewed as a positive characteristic in terms of wildlife and biodiversity (Bull 2002). However, in this region, retaining recent downed wood must be weighed against the risk of insect infestations as well. Storing this material in the forest before transport may attract saproxylic insects, some of which may be deleterious. Given that the majority of feedstock was generated from natural softwoods, fresh pine slash piles may increase the risk of spruce fir beetle, pin engraver, and California five-spined ips outbreaks. This interaction is beyond the scope of this chapter, but these risks should be weighed against the benefits of retaining forest residues for forest structure.

11.3.6 Inland West Region

Overall, the IW region had the lowest potential total acres harvested for woody biomass compared to the other regions under ML 2017—6.0%—but harvested total acres increased to 13.5% and 9.0% under ML and HH 2040 scenarios, respectively (fig. 11.2a). Within the IW region, there were a total of 512,134 potential acres harvested under ML 2017; under ML 2040, this increased 9.6%, but under HH 2040, total harvested acres declined to 301,013 (fig. 11.2). Whole-tree biomass was the predominate feedstock under all scenarios, comprising 65.1% of feedstock under ML 2017 and 59.6% and 51.3% under ML and HH 2040 scenarios, respectively (fig. 11.2). In this region, the harvest method assumption of ForSEAM was that harvests were 50% full-tree method and 50% cut-to-length; under cut-to-length, residues remained on the land.

The greatest concentration of counties with >5,000 acres of total potential harvest occurred in northern Idaho and western Montana, with several counties along the Rocky Mountains in Wyoming, Colorado, and New Mexico (fig. 11.3). Many counties in the southern IW region had few or no acres harvested. This pattern mostly contributed to predominate whole-tree biomass harvests across the scenarios, except for the HH 2040 scenario, where the counties with >5,000 acres were located in New Mexico and one county in Arizona that had predominately logging-residue feedstock (fig. 11.4). Counties with >5,000 acres of young forests created after clearcutting were concentrated in the same locations and also contributed to the large acreage in an Arizona county (i.e., logging residues were produced through clearcutting harvests) (fig. 11.5). Provinces M332 and M333 (fig. 11.1) encompassed the concentration of counties with >5,000 acres total woody-biomass harvesting. Province M333 actually covers counties in the PNW region, and we have included these counties in our results. Because forest-change trends were similar across provinces, total acres were combined and reported below, but were separated graphically (fig. 11.10).

11.3.6.1 Province M332 and M333

Province M332 is 48.8 million acres, and Province M333 is 24.0 million acres. Under ML 2017, 362,363 potential acres were harvested, representing about 0.5% of the land base. Acres harvested for woody biomass were 11.0% and 54.2% lower under ML and HH 2040 scenarios, respectively; the HH 2040 scenario had 48.6% fewer acres harvested compared to ML 2040. Whole-tree biomass harvests were the predominate feedstock under all scenarios: 67.1% under ML 2017 and 69.8% and 59.4% under ML and HH 2040 scenarios, respectively. All counties had >5,000 acres harvested for woody biomass in Province M333. In Province M332, nearly half of the counties that had >5,000 potential acres harvested were located in southwest Wyoming and along the Idaho state border (fig. 11.3).

Logging residues were a byproduct that was almost entirely generated from natural softwood harvests, 99.8% under all scenarios (fig. 11.10). Harvests of lowland hardwood produced remaining logging residues. Nearly all logging residues were generated from clearcut harvests of diameter class 1 under all scenarios. Logging residues from thinning operations (diameter class 2) contributed <0.3% under ML 2017, but no thinning operations occurred under both 2040 scenarios (fig. 11.10). Logging residues were primarily produced from potential harvests in the northeast corner of Washington and western Wyoming.

Text Box 11.17 | Province M332: Middle Rocky Mountain Steppe–Coniferous Forest–Alpine Meadow Province

This province is characterized by the following:

- Temperate desert with warm, dry summers and cool to cold, moist winters
- Precipitation mainly occurs in fall, winter, spring
- Mountainous landscape of moderate elevation or a basin-and-range area consisting of Blue and Salmon River Mountains with high altitudes, and floodplains draining valleys
- Lowland hardwoods—primarily cottonwoods
- Upland hardwoods—primarily aspen
- Natural softwoods—primarily Douglas-fir, lodgepole and ponderosa pine, and subalpine fir
- Planted softwoods—primarily ponderosa pine and Douglas-fir.

Whole-tree biomass was generated from harvest of natural softwoods (96.2%), planted softwoods (1.2%), upland hardwoods (2.5%), and lowland hardwoods (0.20%) under ML 2017, but under ML and HH 2040 scenarios, >99% was generated from harvests of natural softwoods. Acres harvested were 7.4% and 59.5% lower under ML 2040 and HH 2040 compared to ML 2017, respectively. Under ML 2017, 57.1% of feedstock was generated from thinning diameter class 2 natural softwoods; the remaining feedstock was produced from clearcut harvests of diameter classes 2 and 3 natural softwoods. Under ML and HH 2040 scenarios, approximately 58% of feedstock was produced by thinning natural softwood, similar to ML 2017; however, remaining feedstock was produced from clearcut harvests of diameter class 2 only. Only under the HH 2040 scenario did the counties with >5,000 potential acres harvested change significantly, and these counties were only found in northern Wyoming (fig. 11.4b).

Nearly all young forests were created by clearcut harvests of natural softwoods under all scenarios. Harvested acres were 14.4% and 48.4% lower than ML 2017 under ML 2040 and HH 2040, respectively. Total acres of young forest were 39.7% lower under HH 2040 compared to ML 2040. Clearcutting of lowland hardwoods generated remaining young forests under all scenarios (fig. 11.10). In Province M333, the distribution of young forests was in northeast Washington, northern Idaho, and northwestern Wyoming, and in Province M332, distribution of young forests was in southern Wyoming and northeast Idaho borders (fig. 11.5). Under HH 2040, new, young forests shifted to north-central Idaho (fig. 11.5).

11.3.6.2 Biodiversity Effects—Provinces M332 and M333

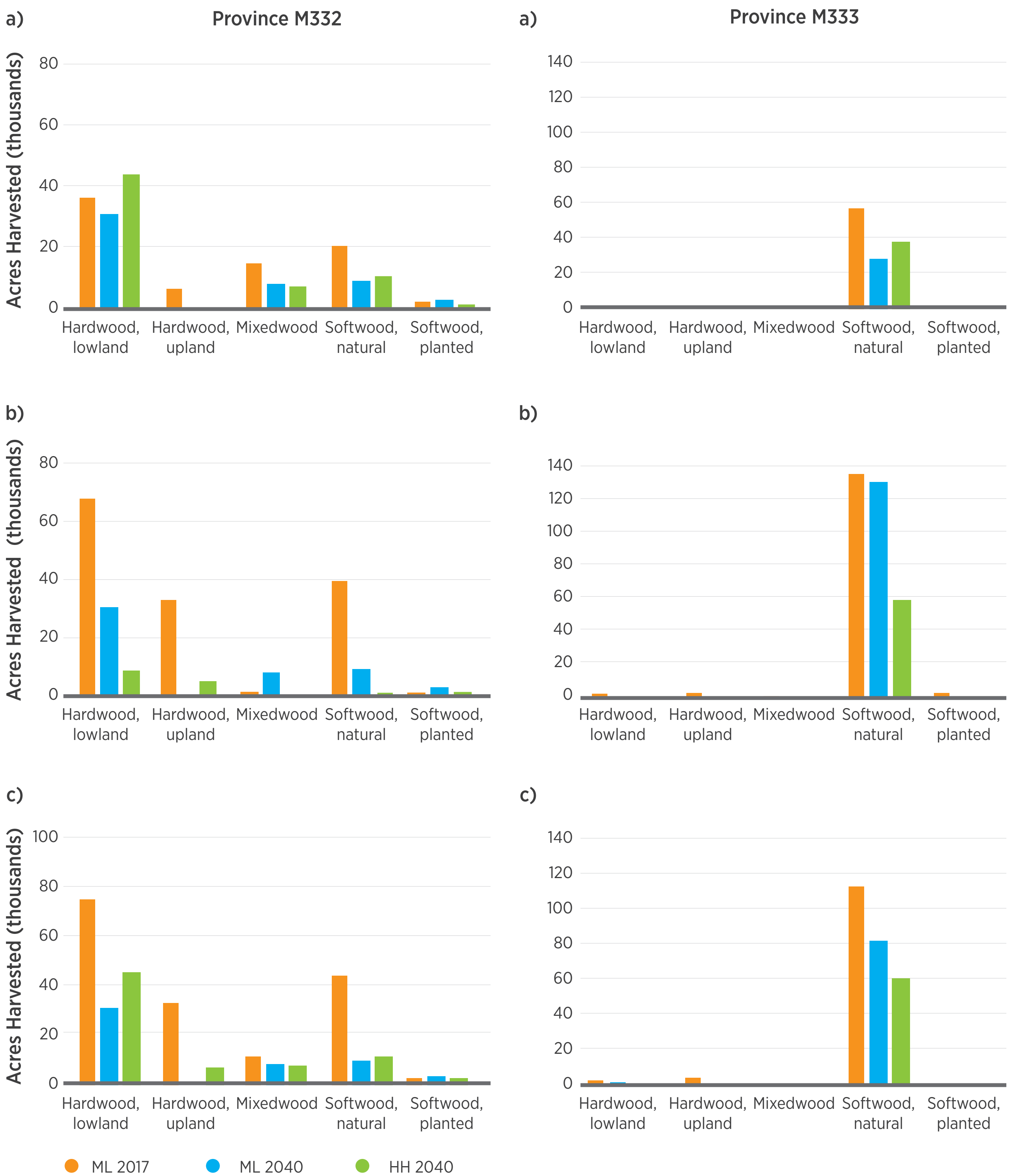
The IW region contributed the lowest quantity of feedstock to national woody-biomass harvests. Most of this contribution was from whole-tree harvests, primarily in natural softwoods, Douglas-fir, and ponderosa pine systems. The IW was the only region in which whole-tree biomass was the major source of feedstock compared to logging residues. About half of whole-tree harvests were generated through clearcutting in the northern Rocky Mountains that would result in young forests. The counties in the central and southern Rockies were predominately harvests with logging residues removed, presenting opportunities to examine effects on biodiversity at a smaller scale. Because relatively small woody-biomass harvests were simulated in this region, we did not present a case study. However, an important aspect of logging residues or effects of whole-tree harvest associated with woody-biomass harvesting in the dry coniferous forest types in this region (e.g., Douglas-fir or ponderosa pine) should be weighed within the context of fire risk in this region and the western United States. Many of the issues surrounding woody-biomass removal are similar to fuel-reduction treatments and biodiversity in these systems (Pilliod et al. 2006); whole-tree biomass harvests could be fuel-reduction harvests under the assumptions of the model.

Text Box 11.18 | Provinces M333: Northern Rocky Mountain Forest–Steppe–Coniferous Forest–Alpine Meadow Province

This high-elevation area is characterized by the following:

- Temperate climate with warm, dry summers and cold, moist winters with heavy snowfall; small glaciers in northern areas
- Mountainous landscape of high-relief; mixed conifer-deciduous forests predominant with major forest types being Douglas-fir and cedar-hemlock-Douglas-fir forests
- Subalpine dominated by Engelmann spruce and subalpine fir
- Montane belt dominated by Western red cedar and Western hemlock, and other common species include western white pine, western larch, grand fir, and western ponderosa pine
- Lowland hardwoods—primarily cottonwoods and red alder
- Upland hardwoods—primarily aspen and paper birch
- Natural softwoods—primarily Douglas-fir, lodgepole and ponderosa pines, western larch, grand fir, and western red cedar
- Planted softwoods—primarily Douglas-fir and ponderosa pine.

Figure 11.10 | Potential acres harvested by forest cover for Province M332 (*left*) and M333 (*right*) within the IW region by (a) logging residue feedstock, (b) whole-tree biomass feedstock, and (c) open forest canopy condition (i.e., young forest); note the different scales for each province.



11.4 Discussion

Overall, it appears that forest woody-biomass harvest, as modeled under the examined scenarios in *BT16* volume 1, will primarily affect biodiversity through changes in forest structure, both at the stand scale (e.g., CWD, FWD, canopy closure, etc.) and the landscape scale (e.g., distribution of stand ages). For all ForSEAM regions and scenarios we examined, effects of biomass removal on habitat conditions may not be a driver for biodiversity responses at broad spatial scales due to the small proportion of forested area harvested (generally <2% for most regions) and other potential broad-scale processes. However, the spatial distribution of potential harvests under all scenarios indicate that harvesting activities are concentrated in the same relative locations across the United States. Species could be negatively or positively affected at the province ecoregion unit scale based on species distributions, specific habitat requirements, and proportion of forest types affected by biomass harvest at the local scale. For example, potential biomass-harvesting activities were more intense in some forest systems that may be of concern in a given region, such as lowland hardwoods in the S region.

A primary concern with biomass harvest relative to biodiversity is the removal of dead and downed wood, and an increase in young forests from clearcutting smaller-diameter trees. However, as outlined in the introduction, it cannot be assumed that removal of this material due to biomass-only harvest will be a direct cause of local extirpations, especially as logging residues could be a product of conventional harvests under the integrated harvesting system of the ForSEAM model. In some cases, removal may lower habitat quality to such an extent that it reduces local numbers, thereby increasing vulnerability to other factors affecting the population, such as competition or fragmentation effects. In other cases, species associated with FWD and CWD may not actually be dependent on long-term presence of this material, or the creation of young forests may benefit other species. Economics of biomass harvest dictate that some material will be left on-site (i.e., material that is not economical to remove), meaning the amount retained after a biomass harvest may in fact be greater than retention rates recommended in existing biomass-harvest BMP guidelines for some forest systems (e.g., the S region). Recent studies in pine forests in the S indicate minimal response by vertebrate species to removal of FWD and CWD under current operational practices, even without application of biomass-harvest guidelines (see citations in the introduction). However, there is a general lack of studies that have examined potential causality between thresholds of woody debris amounts and biodiversity in forest systems and ecoregions across the United States, especially for relationships between biodiversity and FWD.

11.4.1 Implications of Results

Our results show that effects of woody-biomass potential varied regionally based on the forest systems sourcing feedstock. ForSEAM is an economic demand model that met analysis region demands first through logging residues associated with conventional harvests. Whole-tree biomass harvests increased use of smaller-diameter trees in those regions where demand was not met by logging residues, such as in the NE region in the near term (ML 2017) and the IW (all scenarios). An increase in young forests through clearcutting may be beneficial for NE species given the forest types present, but it also may be negative for a suite of species in temperate rainforests of the PNW that depend on closed canopies and moist conditions. Although harvests of logging residues in the model included a 30% retention rate to address BMPs, the modeled biomass harvests were not constrained further based on any certification or regulatory requirements. For example, most biomass harvests will be carried out under the auspices of a forest-certification program, biomass-certification program, or the Sustainable Forestry Initiative Fiber Sourcing

Standard, all of which mandate protection of known occurrences of Threatened and Endangered species, rare communities, and forest types of conservation concern. Additional state and federal forest-management regulations, federal rules (e.g., Endangered Species Act), state regulations for imperiled species, and forestry and biomass BMPs also govern specifics of any forest harvest, including biomass harvest. This provides an overarching structure of protection for imperiled species and communities that was not considered in the examined scenarios. Potential effects of biomass harvests, particularly on protected species or rare communities, should be assessed within the ecological context of these regulations as well as other driving factors influencing populations, such as competition.

As mentioned under the PNW and IW regions, the tradeoffs of retaining dead and downed material must be weighed within the broader context of other processes affecting forests regionally. Lowering habitat quality for some species by removing forest structure or smaller-diameter trees must be assessed against removing material to lower the risk of insect infestations and fire, decreasing old-growth characteristics in the western United States, and negatively impacting local economics. For example, in the eastern United States, urbanization is the greatest threat to forest cover, especially in the southeastern United States, as more than 80% of forested land in the region is privately owned (Wear and Greis 2012). As such, it is critically important that private landowners realize an economic return on their land so that it remains forested (Lubowski, Plantinga, and Stavins 2008). Biomass markets provide a potential revenue source for private landowners that may help provide these economic incentives (Abt et al. 2014). Therefore, when examining potential implications of biomass harvest on biodiversity, it is important to not only put effects in their ecological context, but also in the broader context of maintaining forest cover across the landscape.

11.4.2 Uncertainties and Limitations

The influence of model assumptions on results must be considered when interpreting reported patterns. The assumption that a stand could only be harvested once during the modeling time period contributed to the general decline of total potential acres harvested under 2040 scenarios. Given the two-decade time period between 2017 and 2040 model scenarios, some forest-type stands would at the very least be available for a thinning harvest after initial clearcuts, and stands could have been thinned one to two additional times during the scenario period. Therefore, the potential reductions in some habitat classifications (e.g., early successional conditions) may not be realistic. In addition, the order of entrance by cheapest forest type (hardwoods) into the ForSEAM model to meet supply demands may have shifted impact to forest systems not usually harvested through clearcuts, such as lowland hardwood. The potential expanded role of lowland hardwoods in providing feedstock in certain regions may not be realistic given regional and local management practices. In addition, harvested logging residues from other forest systems could be greater in some regions than what is reported here. Biomass harvesting intensities at smaller spatial scales should be assessed. Although logging residues were considered part of conventional harvests, a reality not captured by the model is that sawtimber harvest largely drives timber markets in the S region. As a result, biomass is, at best, a "come along" activity, and not a primary driver of forest harvest in a region that could provide half of woody-biomass feedstock. Therefore, potential effects described in this assessment could be viewed as not the primary causative factor for biodiversity response to forest management, especially when considering the much larger issue of forest conversion due to urbanization. The ForSEAM assumption of no forest conversion (loss), especially in the eastern United States, simplifies to some degree the effect of the biomass market;

however, as stated earlier, if private landowners are not able to make their land economically viable, the greater impact to biodiversity may be habitat loss rather than habitat quality issues in urbanizing areas of the United States.

Several other assumptions of the ForSEAM model or our approach limited our ability to assess effects of forest woody-biomass harvesting on biodiversity. Because we only compared harvest intensities between two points in time and under explicit assumptions, we were not able to assess cumulative effects of annual removal. In addition, the model constrained potential biomass harvests to within a small distance from roads. This limitation may provide an unrealistic estimation of potential biomass-harvest activities and restrict the modeling of potential landscape-scale changes to a smaller area than is likely truly available for harvest. Because data are presented at the county-level, we could not assess road density or widening of road effects (e.g., no cover) on biodiversity. This county-level resolution also compromised spatial interpretations of potential outcomes. Landscape pattern was not integrated, and we were unable to determine site-level impacts as harvested sites will be located in various landscape contexts. Managers can also implement harvests in various ways to influence residual stand structure to address occurrences of species of concern. By focusing our assessment on province ecoregions encompassing counties with greater potential harvests (i.e., >5,000 acres), we did not assess the effects of removing woody biomass or increasing whole-tree harvests (i.e., clearcuts) from landscapes that are predominantly agricultural or urban, rather than forest. Removing logging residues or increasing whole-tree harvests in these counties may have a proportionally greater impact on species assemblages (e.g., minimum patch sizes, increased isolation effects) than in the more continuously forested landscapes that we assessed.

11.5 Summary and Future Research

In *BT16* volume 1, the potential harvest intensity of woody-biomass harvests varied across the United States, but nearly half of potential harvests occurred in the southern ForSEAM region under all model scenarios. The NC and NE provided the next greatest quantities of biomass under the scenarios. The total potential acres harvested declined under both ML and HH 2040 scenarios, but the regional location of greatest harvest intensities remained primarily along the Atlantic, Gulf, and Pacific coasts, upper Midwest, northern Rocky Mountains, and upper Northeast regions of the country. Logging residues were the dominant potential source feedstock, except in the northern Rocky Mountains where whole-tree biomass harvests were the dominant source feedstock.

Feedstock and forest types producing this potential feedstock varied across the nation, contributing to the variability of biodiversity responses. For example, areas where increasing whole-tree biomass clearcuts were modeled may positively influence some species with the influx of early succession forest stands, but negatively influence other species that rely on moist forest floors. In other words, removing logging residues from some forest systems, especially dry forest types, may not be as negative as removing this structure from lowland hardwoods or forest systems in temperate rainforests of the PNW. This variability, coupled with broader processes, such as economics, urbanization, and insect and fire risk, make it difficult to generalize effects of woody-biomass harvesting.

Given the county-scale data generated by ForSEAM, we used a coarse-filter approach to characterize broad patterns in harvesting intensities. Ecoregion and county-level patterns can be coupled with biodiversity assessments completed at finer resolutions, such as the state level, that track large numbers of species (e.g., state wildlife actions plans) (Mawdsley, Humpert, and Pfaffko 2016).

As noted above, the exact relationships between woody-biomass harvest and biodiversity are not well understood in many regions and forest types due to a lack of empirical research; one exception may be the southeastern Coastal Plain (see 11.1 Introduction). Although general trends in biodiversity response and potential causal relationships can be addressed, the relationships discussed herein should be viewed as the basis for establishing testable hypotheses regarding biodiversity response to biomass harvest.

There is a need to conduct more manipulative studies that vary amounts of CWD and FWD retained across gradients in forest cover and forest types. By measuring the response of multiple species across trophic levels, results can improve understanding of these interactions and how they may influence local and landscape diversity. Manipulative studies can also help determine whether responses are due to the forest-harvest treatment itself or the additive effect of removing dead and downed wood.

Also, there is a need to continue established studies over longer time periods to better understand the effects of removing CWD and FWD during second- and third-rotation harvests. Despite many studies investigating the correlation between biodiversity and the amount of dead and downed material, outstanding questions remain on critical threshold amounts across a variety of forest types and regions to help determine resilience of forest systems to potential harvest intensification. For example, not much is known on the historical range of variability of CWD and FWD prior to fire suppression and other large-scale processes. Are U.S. forests within this historical range of variability in CWD and FWD amounts? Or, functionally, is CWD sufficient to provide the needed structure for many species, given more rapid decomposition of FWD?

Conservation of species amidst an increasing national demand for woody biomass will require taking a multi-scale approach and continued monitoring of species functionally dependent on the material to fulfill their life history requirements.

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12 | Qualitative Analysis of Environmental Effects of Algae Production

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12.1 Introduction

This chapter addresses the environmental effects of potential algal biomass production for biofuels and bioproducts, as described in volume 1 of the *2016 Billion-Ton Report* (*BT16*) (DOE 2016). The chapter emphasizes greenhouse gas (GHG) emissions and water consumption, and considers effects of potential algal biomass production on other environmental indicators. The scenarios include algae production that is co-located with waste CO_2 sources in the conterminous United States.

Microalgae and cyanobacteria are widespread and highly efficient photosynthetic organisms that can use sunlight and nutrients (carbon dioxide [CO_2], nitrogen, phosphorus, and trace metals) to create biomass. Algal biomass contains lipids, proteins, and carbohydrates that can be converted and upgraded to a variety of biogas and biofuel end products, including but not limited to hydrogen, methane, renewable diesel, biodiesel, aviation kerosene, gasoline, butanol, and ethanol. (In pathways not considered in *BT16* volume 1, ethanol can be produced directly by organisms that serve as biological catalysts.) Bioproducts derived from algae include livestock feed, nutritional supplements, and plastics.

Unlike the terrestrial biomass described in earlier chapters, algal biomass for biofuels is not yet economically viable, despite the potential benefits of high biomass yields per unit area and significantly higher energy content per unit mass compared to other terrestrial bioenergy feedstocks (Singh et al. 2011). The smaller-scale production of algae for high-value bioproducts, such as nutritional supplements, fertilizers, and cosmetics, is already economically viable. Technological advances are needed to make algae for biofuel cost-competitive. Because the energy-scale production of algae, especially for fuels, has not yet been demonstrated (White and Ryan 2015), environmental effects of commercial-scale cultivation systems have also rarely been investigated in the field.

The objective of this chapter is to provide a qualitative analysis of environmental effects of the algal biomass potential estimated in *BT16* volume 1. In contrast to the other analyses in this report that focus on three specific price scenarios in 2017 and 2040, this chapter considers aspects of many algal biomass supply and price scenarios from volume 1.

12.2 Scenarios

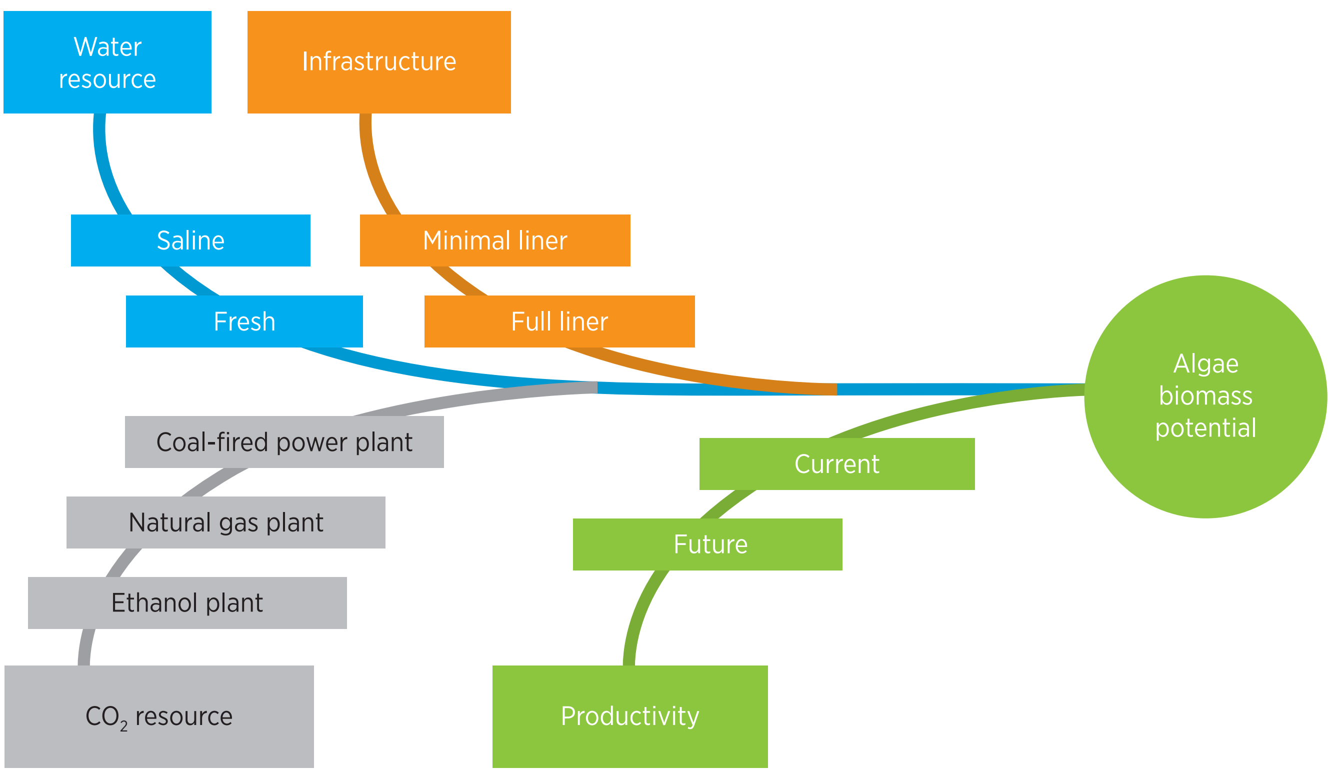
The scenarios from *BT16* volume 1 comprise a subset of the algae production potential that could be co-located with CO_2 sources, i.e., ethanol-production plants, coal-fired power plants, or natural gas–fired power plants. The potential algal biomass represents cultivation at distances from CO_2 sources that would represent cost savings compared to the commercial purchase of CO_2. CO_2 co-location is a strategy used in *BT16* volume 1 to quantify the most likely locations and quantities of algal biomass production in lieu of the strategy used to identify economically available agricultural biomass, i.e., modeling the economics of land management alternatives and selecting the most profitable option for each county. Costs for algae cultivation were taken from an established techno-economic model and recent DOE production design case study (Davis et al. 2016).

The variables that were combined to define the scenarios in the algae analysis in *BT16* volume 1 are depicted in figure 12.1, and rather than providing abbreviations for the scenarios, this chapter and *BT16* volume 1 refer to scenarios as combinations of variables. Potential algal biomass production was estimated for algae grown in open pond-raceway systems that included 405 hectares (ha)—1,000 acres—of pond area. Ponds were ten acres in size and 30 cm in depth, with 100 ponds comprising a "unit farm." The cultivation systems used freshwater or saline water sources and associated algal strains, and minimal (only covering corners prone to erosion) or full, high-density polyethylene liners (the latter in the saline case only). Site-specific current and future productivity scenarios were considered. There was little certainty regarding when particular algae productivities might be achieved, so future productivities were not linked to particular years in algae scenarios as they were in the other scenarios in this report. Algae were assumed to be dewatered to a 20 weight percent (wt %) solids content.

In *BT16* volume 1, national biomass potential and minimum selling prices for the biomass were estimated for *Chlorella sorokiniana* (a freshwater strain) and *Nannochloropsis salina* (a saline strain). Current productivity-rate and future high-productivity scenarios were presented for both strains in *BT16* volume 1. In the current productivity scenarios, while the modeling was done on a site-specific basis at an hourly time step for 30 years, the mean annual biomass growth was 12.8 g/m^2/day for *Chlorella sorokiniana* and 13.8 g/m^2/day for *Nannochloropsis salina*. For the future productivity scenarios, a factor of 1.8 was used to scale up productivities on all freshwater algae cultivation sites, and a factor of 1.95 was used to scale up productivities on all saline cultivation sites, resulting in mean annual productivities of 25 g/m^2/day for both species. In *BT16* algae scenarios, biomass potentials for *Chlorella sorokiniana* in freshwater media under current productivities were estimated to be 12 million tons, 19 million tons, and 15 million tons annually for co-location scenarios with CO_2 from ethanol production plants, coal-fired electric-generating units (EGUs), and natural gas EGUs, respectively.

BT16 volume 1 included algal biomass production scenarios that used fully lined and minimally lined ponds. Ponds were lined with high-density polyethylene liners. The minimally lined ponds used liners to cover small areas at pond turns to prevent erosion. Cultivation systems that were the source of data used

Figure 12.1. | Key variables in the algae analyses in *BT16* volume 1. Full liners were not considered for the freshwater cases. The freshwater algae strain was *Chlorella sorokiniana,* and the saline algae strain was *Nannochloropsis salina*.



to develop the base case in Davis et al. (2016) had liners that covered 2% to 25% of the pond area, and the base case assumptions were used for costing capital and operating expenses in *BT16* volume 1. Only minimally lined ponds were considered for freshwater scenarios.

12.2.1 Environmental Indicators for Algae

Chapter 1 describes a set of environmental indicators that were proposed for sustainability of bioenergy systems in general (McBride et al. 2011). Most of the chapters in this report model these indicators. However, McBride et al. (2011) and a subsequent article (Efroymson et al. 2013) acknowledged that the indicators are generic and would need to be modified for particular contexts, such as algae applications.

Environmental indicators for sustainable bioenergy systems were evaluated for applicability to algal biofuels, including production processes and technologies (Efroymson and Dale 2015). Special emphasis was placed on the indicators proposed by McBride et al. (2011), which represent a focused, scientifically based, and practical set of metrics selected from a broad range of sources. Large sets of indicators recommended by the Global Bioenergy Partnership (2011) and the Roundtable on Sustainable Biomaterials (2010), as well as metrics of potential environmental impacts and resource requirements for sustainable development of algal biofuels addressed by the National Research Council (NRC) (2012), were examined. Environmental indicators for algal biomass and biofuels were selected to be practical, widely applicable, predictable in response, anticipatory of future changes, independent of scale, and responsive to management. Major differences between algae and terrestrial bioenergy feedstocks, as well as their supply chains for biofuel, were considered. Table 12.1 presents a list of 16 proposed environmental indicators for the sustainable production of algae for biofuels (Efroymson and Dale 2015); these are applicable to the estimated algal biomass potential in scenarios from *BT16* volume 1. The proposed indicators are also listed in a section on sustainability considerations for algae cultivation in the Algae Biomass Organization's *Industrial Algae Measurements* (ABO 2015).

The major categories of indicators (i.e., soil quality, water quantity and quality, GHG emissions, biodiversity, air quality, and productivity) are identical to those described in chapter 1 of this report and in McBride et al. (2011). The use of water instead of soil as the growth medium for algae means that

Table 12.1. | A Set of 16 Proposed, Generic Environmental Indicators for Modeling or Measuring the Sustainable Production of Algal Biomass and Biofuels, as Derived from Many National and International Recommendations for Sustainability Indicators, Criteria, and Standards for Bioenergy

Category	Indicator	Units
Soil quality	Bulk density	g/cm^3
Water quantity	Peak storm flow	m^3/s
	Minimum base flow	m^3/s
	Consumptive water use (incorporates base flow)	$m^3/ha/day$; m^3/ton; m^3/GJ (gigajoule)
Water quality	Nitrate concentration in streams (and export)	Concentration: mg/L; export: kg/ha/yr
	Total phosphorus (P) concentration in streams (and export)	Concentration: mg/L; export: kg/ha/yr
	Salinity	Practical salinity unit (PSU)

Category	Indicator	Units
Greenhouse gases	CO_2 equivalent (CO_2e) emissions (CO_2, CH_4, and N_2O)	kg CO_2e/GJ
Biodiversity	Presence of taxa of special concern	Presence
	Habitat of taxa of special concern	ha
	Abundance of released algae	Number/L
Air quality	Tropospheric ozone	Parts per billion (ppb)
	Carbon monoxide	Parts per million (ppm)
	Particulate matter less than 2.5 micrometers (µm) diameter (PM2.5)	Micrograms per m^3 (µg/m^3)
	Particulate matter less than 10µm diameter (PM10)	µg/m^3
Productivity	Primary productivity or yield	g/m^2/day or based on chlorophyll a

Modified from Efroymson and Dale (2015).

water-related indicators could be more important than soil quality indicators, such as soil organic carbon, soil nitrate, and soil phosphorus (Efroymson and Dale 2015). In contrast to the indicators proposed for terrestrial biomass, salinity is included as an environmental indicator for algal biomass production because salinity could be a concern for groundwater and surface waters if saline waters are extracted from the ground or pumped inland from the sea.

Some indicators represent a scientific consensus, whereas other indicators do not. CO_2 equivalent (CO_2e) emissions are an indicator with national and international support, and without competing proposals. Therefore, we do not discuss the advantages or disadvantages of this indicator. While consumptive water use is generally agreed to be an important water quantity indicator, many indicators and indices that incorporate regional context for water have been proposed, and some of these are discussed below and in appendix 12-A and appendix 12-B.

The context in which indicators are measured or modeled may necessitate the use of different functional units from those described above (Efroymson et al. 2013). Indicators may be expressed per biomass, per fuel gallon or gallon gasoline equivalent, per British thermal unit (Btu) impact, or per unit area, for example. Some typical functional units include fuel gallon per consumed gallon of water, fuel gallon per ton CO_2e, and consumed gallons of water per Btu. Water consumption may be expressed with respect to regional water supply or needs. Indicators are typically measured with respect to a baseline.

12.2.2 Indicators and Indices for Water Quantity—The Importance of Regional Context

The distinction between water consumption or consumptive water use (table 12.1) and water withdrawals is important to state upfront. Water withdrawn from a hydrologic system can be used for a purpose, and, depending on the use, a fraction of that water is returned into the system, where it can potentially be used for another purpose (subject to changes in water-quality attributes such as temperature and chemistry) within a short time cycle. Consumptive water use represents the water that is used and removed from the immediate hydrologic system and is not available for other uses. Consumptive water use or water consumption can be driven by evaporation or transpiration or may result from "virtual water," i.e., water that is taken up into a product, such as fruits, vegetables, beverages, etc., and transported as a commodity, often taking the water out of its basin.

Water quantity indicators go beyond the simplicity of the water consumption indicator, and even the flow indicators, described above. Regional climate, competitive uses, and valued entities (e.g., human health, rare ecological populations) are all important factors for selecting water quantity indicators. Indices are generally combinations of measured variables (indicators).

Numerous methods are available to quantify vulnerabilities in available freshwater resources at various temporal and spatial scales. Three key terms related to water-resource vulnerabilities (water scarcity, water stress, and water risk) are defined as part of the United Nations Global Compact CEO Water Mandate[1] (i.e., for chief executive officers of businesses), and with regard to bioenergy development, these must be considered holistically with all aspects of water use. Water scarcity and water stress are discussed in this chapter. Water risk, the probability and severity of an entity experiencing a deleterious water-related event, is considered a socioeconomic indicator (described as risk of catastrophe in Efroymson et al. 2016) and is outside the scope of this chapter.

- **Water scarcity:** The volumetric availability of water supply and the total use of that supply. This indicator is most often calculated as a simple ratio of total consumptive water use to the available water supply within a geographic bound, such as an individual or collection of connected basins or sub-basins. Water scarcity can theoretically be measured as often as needed and at the scales required and, accordingly, is a measure that can be compared spatially and temporally (Schulte 2014).
- **Water stress:** The ability to meet human and ecological water demand in the context of volumetric availability, water quality, environmental flows, and accessibility. Compared to water scarcity, water stress incorporates more elements beyond water supply and water use. Many methods are available for estimating water stress, and the chosen method depends on the temporal and spatial scale, the availability of data, the level of detail required, and the elements of concern for a given location (i.e., a regional study will differ from a site-specific study) (Schulte 2014).

To operationalize these indicators, the total water supply for a given geographic domain and appropriate temporal period needs to be established. In addition, current water withdrawals, consumptive use, and competing uses (including environmental flow requirements) need to be quantified at a common geographic domain and temporal period.

Several key methods are appropriate for regional- and national-scale water planning. These include the Water Resources Vulnerability Index, the Water Supply Stress Index (WaSSI) and Water Supply Stress Index Ratio, the Water Scarcity Index (Wsci), and the Water Stress Ratio.

Water resource indices are described in detail in appendix 12-A. The Water Resources Vulnerability Index, often referred to as the "withdrawal to availability ratio" (WTA ratio), is a water scarcity index and is probably the most simple and most widely used of the water resources indices. All other indices described in this chapter are variants of the basic ratio of water supply to demand (Rijsberman 2006). The WaSSI, originally proposed by Sun et al. (2008a, 2008b) and used in chapter 7, provides a measure of the relative supply and demand of water at a monthly time step for eight-digit Hydrologic Unit Code (HUC) watersheds. Despite the water-stress-related name, by the definitions herein, WaSSI is also a water scarcity index. Asheesh (2007) established the Wsci

[1] See the United Nations Global Compact CEO Water Mandate website for more: http://ceowatermandate.org/.

as a method to measure change in water availability and identify gaps that would lead to unbalanced water supply and demand in the context of a complex relationship of variables, including ecological requirements and population growth rates. This complex relationship of variables is referred to as the Water Equality Accounting System. Under the definitions laid out herein, the Wsci would be considered a holistic water stress index. Smakhtin et al. (2005) provide a simple environmental water-scarcity method, the Water Stress Indicator (WSI), which considers the relationship of water withdrawals to ecosystem water requirements.

Four key points need to be considered when using water-resource indices to evaluate environmental effects of algal biomass or bioenergy production. First, many indices use total or sector-based water withdrawal as an input, such as data that are available from the U.S. Geological Survey (USGS) 5-year water use reports (see http://water.usgs.gov/watuse/). As discussed above, water withdrawal and consumptive water use can lead to different outcomes in the volume of water available for use. Water that is consumptively used is no longer available for use in the basin or hydrologic area of interest, whereas for water withdrawals, depending on the water use sector, some portion of the withdrawn water will be returned to the system.

Second, because microalgae have growth cycles that are largely dependent on meteorological variables (primarily light and temperature), the timing of when water resources are available is critical; thus, indices that use mean annual values do not consider critical seasonal cycles, whereas indices that can incorporate a monthly evaluation are well suited to provide an appropriate level of detail. In addition, because meteorologically-induced growth cycles are also highly location-dependent, indices need to have a reasonably high level of spatial granularity to show variability, where a recommended minimum would be an eight-digit HUC boundary (see http://water.usgs.gov/GIS/huc.html).

Third, for sector-based and competitive water use assessments, indices often do not reflect required environmental flows and ecosystem requirements. Therefore, indices need to incorporate a broader-use context with respect to available supply, even if it means part of that supply remains in the river (an additional competitive water use). Assessment methods for environmental flow requirements can vary significantly in their level of detail.

Lastly, the consideration of future, altered climate, and non-stationarity effects needs to be addressed. Therefore, the use of historic long-term averages may not provide the best approach when considering potential vulnerabilities and changes to future water-resource supply. To meet future needs, the indices may need to be applied differently, but to establish a baseline, the use of historic long-term averages is appropriate.

Environmental flow is an important component of the regional context of water quantity effects of any water-use sector. Peak flow and minimum base flow are described as basic indicators for water quantity in table 12.1, but more complex measures may be needed to incorporate the regional context, some of which are specified in the water resources indices described above and in appendix 12-A. Tharme (2003) identified >200 methods available to assess environmental flows, and generally, they can be classified as hydrological, hydraulic rating, habitat simulation, and holistic methodologies. This taxonomy of environmental flow methods, as well as some of the methods themselves, are described in appendix 12-B. Indicators of environmental flow can be considered indicators of aquatic biodiversity, where flow is an important variable controlling a population or community.

12.3 Methods

In this chapter, we provide a qualitative analysis of environmental effects. Unlike most other chapters, county-level estimates of environmental indicators are not estimated.

We highlight inputs and outputs of models that were used for volume 1 and examine methods that could be used to assess particular environmental effects. For example, GHG emissions estimates are discussed for the base case pond design in the *Design and Economics for the Production of Algal Biomass* "design case study" (Davis et al. 2016), which has similar features to those in the *BT16* algal biomass scenarios and which was used for cost estimates in volume 1.[2] We also examine the modeled water consumption from the Biomass Assessment Tool (BAT) for particular scenarios and describe how those results could be put in the context of regional water use.

The BAT is the Pacific Northwest National Laboratory's integrated model, analysis, and data management suite that couples advanced spatial and numerical models to assess resource requirements, multi-criteria land suitability, site-specific biophysically-based biomass and bioenergy potential, techno-economics, and trade-off analyses (Coleman et al. 2014). With respect to production, the pond temperature and subsequent net consumptive water use (evaporation – precipitation) was modeled using a mass and energy balance model for about 88,000 potential algal production sites across the country using 30 years of hourly stochastic meteorology data and averaged across each state (Wigmosta et al. 2011). Biomass growth was modeled at an hourly time-step over a 30-year period (Wigmosta et al. 2011) for the *Chlorella sorokiniana* and *Nannochloropsis salina* parameterized with monthly temperature data.[3] Additional assumptions used in BAT are described in *BT16* volume 1, chapter 7. We describe some of the water-supply constraints here, as those are pertinent to the water quantity analysis. Annual water consumption estimates for the scenarios are presented graphically in the results section below, although monthly values were also recorded.

Sites for algae cultivation were limited, in part, by water availability (Chiu and Wu 2013; Venteris et al. 2013). In the *BT16* volume 1, a consumptive freshwater-use constraint of no more than 5% of mean annual basin flow (cumulative for sites within a watershed) helped determine the number of sites allowed (ANL, NREL, and PNNL 2012). The map that shows this initial screening of suitable sites is figure 12.2. The 5% target was based on the U.S. Environmental Protection Agency's (EPA's) established water-use rule for new thermoelectric power plants (EPA 2001). Sites were prioritized and selected based on water-use rate within the six-digit HUC until the allocated "water-use reservoir" was depleted. Because saline water resources are more plentiful, they were not constrained by required volume but rather by (1) locations where salinity ranges from 2 to 70 practical salinity units (PSU)[4], which was considered suitable

[2] This design case is used as an illustration of potential assumptions. This design case would probably not be commercially scaled because it does not produce fuel that is cost-competitive with fossil fuel.

[3] Strain-specific biomass productivity is a function of water temperature (minimum, optimal minimum, optimal maximum, and maximum) and light utilization efficiency of photosynthetically active radiation (PAR), both which have site-specific hourly and seasonal signals. Additional parameters that do not vary hourly or seasonally include transmission efficiency of incident solar radiation to microalgae, biomass accumulation efficiency, and others, as defined in Wigmosta et al. (2011).

[4] Bartley et al. (2013) found that salinities of 22 PSU to 34 PSU provided the highest growth rates for *Nannochloropsis salina*; however, growth is possible between 8 PSU and 68 PSU. Abu-Rezq et al. (1999) found that ideal salinities for the same strain are between 20 PSU and 40 PSU. While the salinity range of 2 PSU to 70 PSU is broader than the ideal salinity target range for *Nannochloropsis salina*, it represents possible salinities that support growth of a wide range of other saline-based algae strains (Shen et al. 2015; Varshney et al. 2015; Kim, Lee, and Lee 2016). The wide salinity range also captures the uncertainties in the source data and geostatistical processing of saline water resources.

for *Nannochloropsis salina*, and (2) cultivation sites within 6.2 miles (10 km) of acceptable salinity-range groundwater or seawater sources. The constraints accounted partially for uncertainties in salinity ranges and provide economically viable water transport distances.

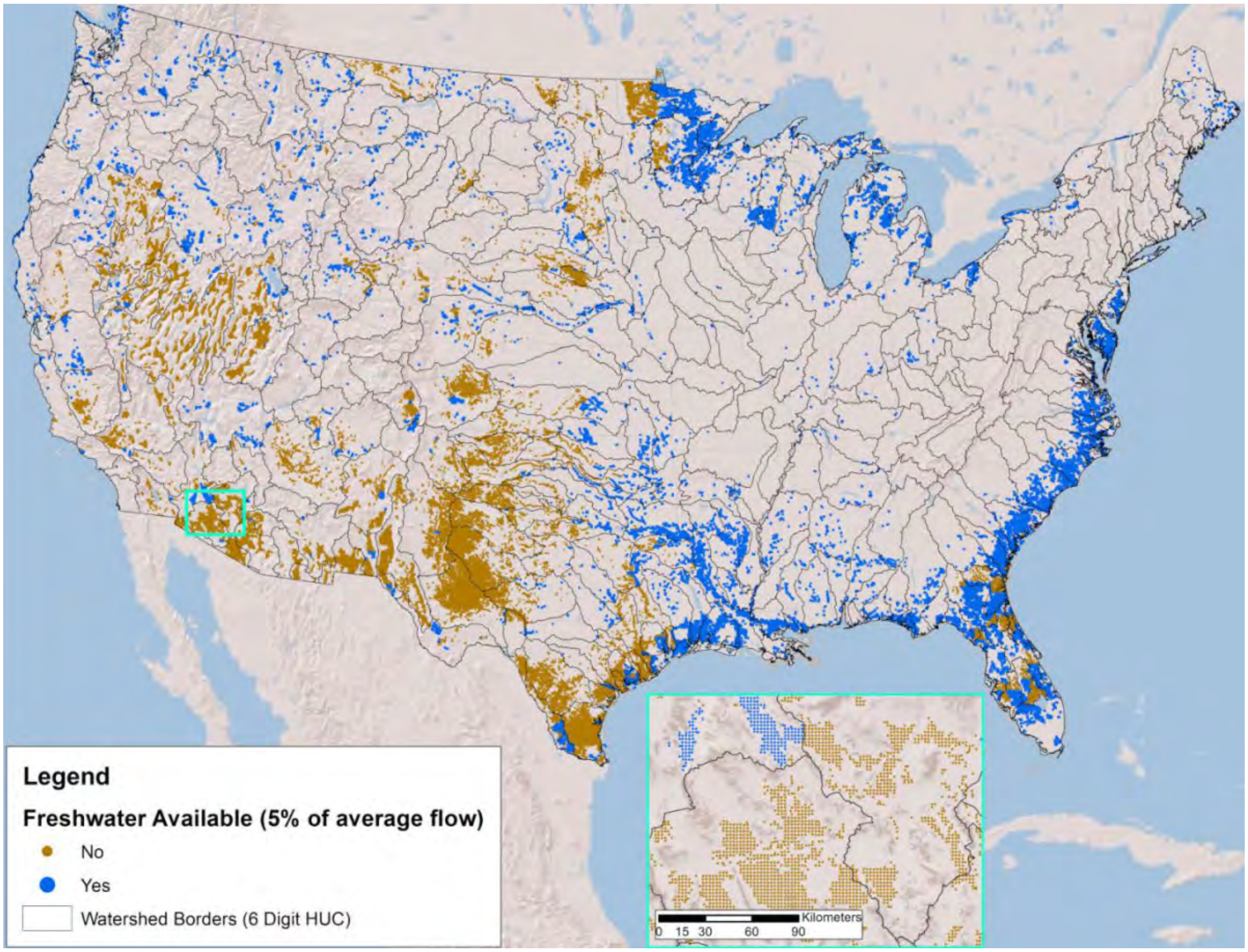
Additional siting considerations in *BT16* volume 1 related to topography and land use. For example, forest and cultivated cropland were not considered for potential algae cultivation facilities.

No simulation models were used above and beyond those applied in the resource assessment studies in *BT16* volume 1. For some indicators, we have provided estimates from relevant scientific literature and summarize variables that can affect those indicators.

12.3.1 Scope of Assessment

The variables that were considered are shown in figure 12.1 and included CO_2 co-location source, fresh or saline water, productivity, and pond liner area. All categories of environmental indicators from table 12.1 are discussed, with an emphasis on GHG emissions and water quantity indicators.

Figure 12.2. | Freshwater availability for potential algal-production sites in the conterminous United States. Water availability is determined using 5% of long-term mean annual flow at a six-digit HUC and budgeted against modeled open-pond consumptive water use at each site (Image credit: ANL, NREL, and PNNL 2012).



GHG and water quantity indicators are emphasized because (1) GHG emissions relate to many of the variables in the *BT16* volume 1 scenarios (co-location with CO_2, productivity, pond liner area), (2) water consumption was tracked in the biomass production modeling and comprised a supply constraint for algae production, and (3) most environmental analyses of algal biomass in the literature focus on GHG and water indicators. Water consumption associated with algae production in the scenarios is described quantitatively. Other environmental indicators are discussed more generally. Directional changes in environmental indicators that might result from changes in variables in the scenarios are also discussed. The methods and context of indicator measurements are described.

All scenarios from *BT16* volume 1 are considered, rather than selecting key scenarios for analysis, as for terrestrial biomass. As with other chapters, the only comparisons that can be made are among scenarios, but we cite scientific literature that makes other comparisons.

The use of photobioreactors (PBRs) is an alternative major production strategy that was not considered in the biomass estimates in *BT16* volume 1 and therefore is not addressed in detail. Pathways where algae serve as a "biocatalyst" (for example, whereby ethanol and/or hydrocarbons are secreted by cyanobacteria), were not considered in *BT16* volume 1 or in this analysis. The exclusion of these systems from the analysis does not reflect their presumed effects or lack of effects with respect to the environmental indicators described in this chapter.

12.4 Results and Discussion

This chapter is a qualitative discussion of potential environmental effects of algal biomass production with implications and uncertainties discussed in an integrated way. Formal quantitative analysis of indicators is not presented. The results and discussion sections are combined.

12.4.1 GHG Emissions

GHG emissions contribute to climate change (IPCC 2007) and related environmental and health effects, some of which are adverse (Church et al. 2013). Because the atmosphere is well mixed, effects do not depend on the counties where the gases were released or sequestered. However, climate-change effects are regional.

The primary environmental indicator of GHG emissions is CO_2e, which includes CO_2, methane [CH_4] and nitrogen dioxide [N_2O]) (table 12.1). Because the objective of this chapter is to describe cultivation-related processes that contribute to GHG emissions and other effects, we do not focus on conversion or treatment of waste products, nor do we quantify net emissions from the full supply chain. However, we discuss processes that affect GHG emissions in the full supply chain in section 12.4.1.7.

In this chapter, we focus primarily on CO_2 emissions, though we discuss other GHGs. GHG emissions from algae cultivation and dewatering are driven by the processes to capture and potentially purify and transport the CO_2, as well as additional process electricity and nitrogen and phosphorus inputs. As a general rule of thumb, to produce 1 ton of ash-free dry weight (AFDW), algal biomass requires about 1.8 tons of CO_2. The optimum CO_2 concentration for algae production depends on the strain, system design, meteorological conditions (temperature and light) and operating conditions (Zhu 2015), as well as pH.

CH_4 and N_2O are important components of CO_2e for algal biofuel systems (table 12.1) but not for the algal biomass production step of the supply chain (this analysis). The primary source of these gases is an anaerobic digestion process if used to generate power from lipid extracted algae or other process waste (Frank et al. 2012). N_2O emissions have also been measured under some algae cultivation systems, e.g., *Nannochloropsis salina* (Eustigmatophyceae) under a nitrogen headspace (Fagerstone et al. 2011), *Nannochloris* (Chlorophyta) in coastal open-pond systems emit N_2O during senescence (Florez-Leiva et al. 2010), and *Chlorella vulgaris* in high rate algal pond wastewater treatment systems supplied with nitrite in darkness (Alcántara et al. 2015). We do not quantify or discuss further potential emissions of N_2O from ponds, as these emissions would probably be small or negligible.

The consideration of the GHG impacts of algae supply chains should include the full life cycle of production through use. A number of end-use applications of algae offer the potential for substantial life-cycle GHG reductions relative to approaches that involve only cultivation. Some of these aspects of the life cycle are addressed in section 12.4.1.7.

The discussion of GHG emissions begins with the summary of a base case from the National Renewable Energy Laboratory (NREL) design case report, *Process Design and Economics for the Production of Algal Biomass* (Davis et al. 2016). Then, we discuss the benefits of co-location of algae with CO_2 sources, and in particular, the CO_2 used in the algal biomass projections in volume 1. Other variables in volume 1 scenarios (current and future productivity, freshwater versus saltwater, fully lined versus minimally lined ponds) and their potential implications for GHG emissions are also discussed. Finally, we summarize some of the important variables from the literature that have been shown to influence GHG emissions from algae cultivation or dewatering systems. These include variables that affect energy return on investment (EROI).

12.4.1.1 An Algae Base Case

Analogous to the algae culture-system design and costs in *BT16* volume 1, which modify costs in Davis et al. (2016), the GHG emissions discussion in this chapter begins with the GHG emissions estimate from *Process Design and Economics for the Production of Algal Biomass* (Davis et al. 2016). Departures from Davis et al.'s (2016) assumptions in *BT16* volume 1 regarding unit farm size, pond size, species, and resource assumptions are discussed in detail in volume 1 and summarized below.

Davis et al. (2016) assume a freshwater open-pond/raceway cultivation system with an assumed cost that represents the average cost of four 10-acre pond designs, and, unlike the strains assumed in this analysis—*Chlorella sorokiniana* and *Nannochloropsis salina*—they project productivities and GHG emissions for *Scenedesmus acutus (LRB-AP 0401)*. Davis et al.'s (2016) base case and most of the algae scenarios from *BT16* volume 1 assume a minimal liner that covers 2%–25% of total pond area in the four pond designs from which Davis et al.'s (2016) base case is derived. Davis et al. (2016) assume in-ground gravity settlers, followed by hollow fiber membranes and centrifugation to concentrate or dewater the harvested biomass; however, they note that the dewatering performance represents research and development advancement goals to meet cost targets.

We assume the same inoculum technology, water-circulation pipelines, average pond-circulation power demand, and product storage tanks as in Davis et al. (2016), and therefore, the same GHG emissions for these components. As in Davis et al. (2016), biomass is harvested and processed through three dewatering steps—gravity settling, hollow fiber membranes, and centrifugation—to concentrate the biomass from 0.5 g/L (0.05 wt % AFDW) to 200 g/L (20 wt %) in the product stream.

Davis et al. (2016) assume that purified CO_2 from flue gas carbon capture is captured (amine scrubbing, membrane purification, etc.) and delivered to the unit farm (cultivation system) via pipeline under high pressure, is stored in pressurized spherical storage tanks, and is distributed and sparged into individual ponds during daytime production. The *BT16* scenarios assume that cultivation is co-located with existing natural gas power plants, coal-fired power plants, or ethanol plants, and CO_2 is sourced via a low pressure, direct flue gas feed. While CO_2 concentrations vary depending on the source and thus will impact transport efficiencies, the movement of non-stripped flue gas can reduce capital equipment needs and lower the parasitic power load (energy used for internal purposes rather than exporting) from the power plants. For non–power plant CO_2, a reduction in imported energy could be expected.

As Davis et al. (2016) note, "Both the CO_2 input and 'emissions to the air' from the cultivation ponds (attributed to CO_2 retention efficiency losses) are treated as biogenic in nature, following accepted methodologies for CO_2 accounting in algal biofuels LCA [life-cycle-analyses] which dictate that although the CO_2 originates from fossil power plant flue gas, the power plant is operated to generate power and not to provide CO_2, which otherwise would be emitted to the atmosphere and then later could be utilized in dilute form as biogenic CO_2 for growing a different biomass resource" (Frank et al. 2011). Biogenic CO_2 does not add to GHG in the atmosphere (Karlsson and Byström 2010) and is not accounted for in the Intergovernmental Panel on Climate Change global warming methodology (Fisher et al. 2007). CO_2 sourced from an ethanol production plant would also be biogenic, since biomass is processed to produce it.

The GHG emissions for the base case (ending with the partial dewatering, as described above) in Davis et al. (2016) are estimated at 0.73 ton CO_2e/ton AFDW biomass, with 0.38 ton CO_2e/ton AFDW biomass representing emissions due to carbon capture from flue gas and 0.30 ton CO_2e/ton AFDW biomass representing process electricity. Given that the process of CO_2 transport assumed in *BT16* should not use as much compression energy (or related electricity) as monoethanolamine carbon capture, the associated CO_2e/ton algal biomass in the *BT16* system should be lower than emissions in Davis et al. (2016). One algal biofuel company that is located adjacent to a coal-fired power plant from which it obtains CO_2, captures, and delivers CO_2 without compression, with GHG emissions reported on the order of 0.03 ton CO_2e/ton AFDW algal biomass.

The life-cycle inventory on which GHG emissions in Davis et al. (2016) are based includes values for biomass; nutrient, water and electricity demands; water and biomass lost to blowdown (i.e., pumping water exchange); and water, CO_2, and O_2 emissions, with the energy to capture CO_2 estimated separately (Davis et al. 2016, table 20). The quantity of GHG emissions is driven primarily by the processes to capture, purify, and transport the CO_2, and also includes process electricity (U.S. average electricity mix from the grid, 0.65 kg CO_2e/kWh) and ammonia and diammonium phosphate nutrient inputs. The inputs and outputs are not presented here to avoid confusion with the site-specific analysis that was performed in *BT16* volume 1 and resulted in biomass, water, and CO_2 used that were driven by meteorological variables. So, while Davis et al. (2016) provide a good starting point to estimate GHG emissions, they do not provide regionally specific GHG emissions.

Some differences between the assumptions in this chapter (taken from *BT16* volume 1) and in Davis et al. (2016) affect GHG emissions from algal biomass production and logistics processes for the current or future productivity cases. Scenario differences in *BT16* are summarized in table 7.5 of volume 1. *BT16* scenarios include 100 10-acre ponds per facility, rather than Davis et al.'s 500 10-acre ponds; *Chlorella sorokiniana* and *Nannochloropsis salina,* rather than mid-harvest, high-carbohydrate *Scenedesmus acutus*; site-specific current and future productivities, rather than a cultivation productivity target; and saline media for some of the scenarios instead of just freshwater.

Some of the differences between scenarios in *BT16* and Davis et al. (2016)—for example, productivity estimates—relate to the different purpose of *BT16*, which is to estimate current and future national biomass potential, compared with that of the cultivation design case report, which is to describe "aspirational" targets for future facilities. For the current productivity scenarios in *BT16* volume 1, we assumed lower site productivities than the target in Davis et al. (2016).

The use of saline water affects estimates of GHG emissions. We consider scenarios that assume that ponds are fully lined if saline water is used. However, we recognize liners are not a requirement for every location, so we also consider scenarios wherein saline ponds are minimally lined, as with freshwater, with the objective of controlling pond erosion. The use of injection wells is assumed for media disposal under all saline water scenarios. Note, however, that other saline-disposal options exist, including evaporation ponds and landfill-style disposal of salt, discharge to a water-treatment facility, and, in the case of coastal sites, cleanup and discharge to the ocean (Mickley 2001). Additionally, there are beneficial uses for saline concentrate including oil-well field injection, solar ponds, aquaculture, wetland creation/restoration, and high-value salt and chemical products (GEO-Processors USA 2006; Jordahl 2006).

In addition, the *BT16* scenarios do not reflect the carbon capture and compression assumptions from Davis et al. (2016). Instead, the CO_2 in *BT16* is transported from sources to the algae production sites (unit farms) using pipelines and blowers, which have a smaller (but unquantified) energy footprint and GHG emissions. The specifics of co-location are described below.

12.4.1.2 Co-Location with CO_2

Table 12.2 shows the CO_2-related benefits for power plants from the scenarios in *BT16* volume 1, including the total CO_2 used in the algae production scenarios and the percentage of the total across the conterminous United States. These quantities of CO_2 are largely utilized by algae with a fraction released to the atmosphere through pond outgassing. This fraction is assumed to be 18% in *BT16* volume 2 and 10% in Davis et al. (2016). (Atmospheric release of CO_2 for PBR systems would be minimal).

Co-location of algae with CO_2 in these scenarios should delay CO_2 emissions to the air and provide a beneficial use of the CO_2 compared to scenarios in which the gas is emitted directly to air. In addition, the potential displacement of fossil-based fuels with algae-sourced fuels should be considered in the estimate of overall CO_2 reduction. Although CO_2 can be temporarily used by algae (Menetrez 2012), the decomposition rate of waste biomass and recycling is an important consideration for determining the quantity and rate of emissions (Fernandez et al. 2012) (See section 12.4.1.7). Moreover, waste CO_2 utilized by algae in the *BT16* scenarios might be released to the atmosphere more rapidly than if the CO_2 were captured and sequestered in an underground geological formation (carbon capture and sequestration [CCS] technology). The additional power (i.e., parasitic power load) required for CCS is substantial (20%–30%); thus, more power needs to be generated to maintain contracted electricity exports. The CCS approach then requires more energy resources, such as coal or natural gas, and ultimately generates more CO_2. This scenario needs to be considered in the context of the reference case.

The energy and infrastructure required to capture and transport impure CO_2, as in *BT16* volume 1, can be substantial. However, large capital and energy costs and related GHG emissions associated with purifying and compressing the CO_2 are avoided.

The GHG emissions estimate from Davis et al. (2016) above was presented with the caveat that an important factor determining the GHG emissions is the CO_2 carbon-capture technology occurring off-site, which is based on current technology, i.e., monoethanolamine carbon capture. This accounted for 55% of the emissions. This emissions output should improve with co-location with CO_2 sources considered in volume 1 or second-generation carbon-capture technology.

Table 12.2. | Summary Results for Potential Algal Biomass from CO_2 Co-Location with Ethanol Production Plants, Coal Power Plants, and Natural Gas Power Plants Using *Chlorella sorokiniana* (Freshwater) or *Nannochloropsis salina* (Saline) under Current and Anticipated Future Productivities

	Chlorella sorokiniana			*Nannochloropsis salina*		
	Ethanol production	Coal EGU	Natural gas EGU	Ethanol production	Coal EGU	Natural gas EGU
Current productivity						
Total annual biomass (million tons/year)	12	19	15	10	54	21
Total cultivation area (thousand acres)	905	1,257	790	793	3,349	1,096
Total CO_2 used (million tons/year)	29	46	37	25	134	52
Percentage of total CO_2 in conterminous United States used in co-located algae production	19.3%	1.7%	8.9%	16.8%	4.91%	12.6%
Average distance from CO_2 source to algae facility (miles)	15.2	6.2	4.8	16.0	8.9	6.7
Future productivity						
Total annual biomass (million tons/year)	13	10	--	11	12	--
Total cultivation area (thousand acres)	508	257	--	435	299	--
Total CO_2 used (million tons/year)	32	25	--	28	30	--
Percentage of total CO_2 in conterminous United States used in co-located algae production	21.3%	0.9%	--	18.5%	1.1%	--
Average distance from CO_2 source to algae facility (miles)	14.5	3.8	--	14.6	4.4	--

These analyses are limited to the conterminous United States.

EGU is electric generating unit

Low compression of CO_2 with blowers, as well as piping, varies with productivities, strains of algae, and the co-location scenarios, i.e., the concentration of CO_2 in flue gas and distances from source to facility. Both compression and piping will influence CO_2 emissions. CO_2 outgassing losses are usually higher for sparging low-pressure, nitrogen-containing flue gas into pond sumps (15%–25% losses), than sparging high-pressure, purified gas into sumps (10% losses), according to Bao et al. 2012, de Godos et al. 2014, and Davis et al. 2016.

This variability in CO_2-use efficiency was not captured in *BT16* volume 1, which assumed 82% utilization efficiency that was directly related to variability in biomass growth and subsequent CO_2 demand.

12.4.1.3 Productivity

Increases in productivity are associated with improved efficiencies and more energetically favorable cultivation systems than lower productivities (Sills et al. 2013). Energy requirements associated with plastic liner manufacture, mixing, sparging CO_2, water transport, and dewatering are lower per unit biomass as areal productivity increases (Sills et al. 2013). Moving the water to and from the dewatering step is energy and CO_2 intensive (Frank et al. 2011; Weschler et al. 2014), so moving less water per unit biomass is advantageous with respect to GHG emissions on a biomass or biofuel basis.

12.4.1.4 Saline versus Freshwater

We do not know of a study that has modeled or measured GHG emissions from saline and freshwater culture media under similar conditions. While freshwater is easier and less expensive to access, it has a higher rate of evaporation and, thus, a greater consumptive use of water. There is an opportunity for water recycling, which will ultimately help minimize pumping energetics, as opposed to continually drawing the full, required water volume from a clean source. Algae production using brackish, saline, or hypersaline waters (dependent upon the requirements of specific algal strains) requires water sources that can be more energy intensive to access (e.g., deep saline groundwater). Near-coastal sites may be an exception. These ponds must maintain specific salinity ranges for optimal biomass production, requiring new water to be pumped in and old water expelled (i.e., blowdown). In general, required blowdown water and associated pumping energetics will decrease with an increase in the strain-specific allowable pond salinity concentration.

For both freshwater and saline-based ponds, there is a need to dispose of the water. Freshwater can be recycled back to the production ponds where remnant nutrients can be made available for use, or water can be treated and discharged according permitted regulation (e.g., National Pollutant Discharge Elimination System permit). Saline water will require one of several options for disposal, as noted previously, some of which could be more energy- and cost-intensive (e.g., re-injection wells, ocean outfall) than freshwater disposal, and some less energy-intensive but not necessarily less cost-intensive (e.g., evaporation ponds).

12.4.1.5 Full Plastic Liner versus Minimal Plastic Liner

Energy is required to produce the plastic for a polyethylene pond liner. This energy translates to GHG emissions. While some energy could be required to make unlined ponds suitable for cultivation (e.g., compaction, movement of clay, or addition of carbon source to promote microbial clogging), this energy, and the associated GHG emissions, should be lower than that associated with lined ponds.

Canter et al. (2014) studied infrastructure-associated emissions for renewable diesel production from algae in ponds based on a design by Lundquist et al. (2010) and found that unlined ponds showed a 39% decrease in GHG emissions compared to the baseline high-density polyethylene (HDPE) pond liner design. Even if plastic liners were only used to cover berms to protect against erosion, this infrastructure accounted for a large fraction of infrastructure-related pond emissions. GHG emissions were highly sensitive to pond liner thickness and material lifetime. As Canter et al. (2014) note, "the first step to reducing infrastructure-cycle emissions would be to reduce or eliminate pond liners if soil conditions and environmental regulations permit." The material lifetime would be an important determinant of emissions.

Liners also affect hydrodynamics and the energy required for mixing. The hydrodynamics are related to roughness coefficients (Chow 1959). Energy differences translate into differences in GHG emissions.

Moreover, excavating and moving soil or covering soil with a liner would be expected to affect the carbon dynamics of soil and associated GHG emissions, compared to unlined ponds or relatively undisturbed soils. However, these potential effects have not been studied.

12.4.1.6 Highlights of the Literature on GHG Emissions, Energy Inputs, and Land-Use Change Related to Algae Cultivation

In this subsection, we highlight some of the literature on factors that influence GHG emissions from cultivation in ponds, the resources and infrastructure needed for cultivation, and dewatering processes. These are factors that are not considered in the comparisons above. Literature on net energy inputs (i.e., energy return on investment [EROI]) is also considered, because energy use—particularly fossil energy use—affects GHG emissions.

Losses of CO_2 from open ponds influence net emissions. Both Davis et al.'s (2016) and the *BT16* volume 2 estimates of CO_2-use efficiency and outgassing would change with productivity, pH, temperature, and water flow changes associated with the *BT16* scenarios, in addition to changing with the CO_2 concentration in flue gas, as discussed above. White and Ryan (2015) note that Sapphire Energy's placement of CO_2 diffusers within a sump for high CO_2-use efficiency would not be feasible at a scale of thousands of acres, because of the tendency of sumps to fill with sand and silt.

Factors affecting EROI of algal biofuel production were described in an evaluation of socioeconomic indicators of algal biofuels (Efroymson et al. 2016). The subset of factors that apply to cultivation of algae in ponds and subsequent dewatering are presented in table 12.3. A theme in the literature is that environmental and economic costs of biofuel production are more favorable when microalgae or cyanobacteria are produced using renewable energy sources, such as solar, wind, or on-site biomass-generated methane for electricity production (Beal et al. 2015). Passell et al. (2013) found that increasing productivity and scale decreased the net energy ratio (energy in/energy out).

GHG-emissions indicators also reflect land-use change (LUC) that would be attributable to algal biofuel systems. Land converted to algal biomass production systems is expected to include industrial brownfields, rangelands, deserts, abandoned or unproductive farmland, dredge spoil islands, or other coastal areas (NRC 2012). The production system could decrease soil carbon sequestration to an extent that would depend on the CO_2 storage associated with the baseline land condition and the surface soil that was excavated. Arita et al. (2016) found that including the contribution of the direct LUC associated with carbon stocks disturbed by algae facilities would mean that some of the suitable siting locations for algae facilities from the scientific literature (based on GHG emissions criteria) would no longer meet the net emissions benefits criterion for advanced biofuels under the U.S. Renewable Fuel Standard (RFS2).

Indirect LUC could result from algae production if land management is altered on distant land as a result of algae production, and that could have GHG implications. Variations in the definition of indirect LUC are addressed in chapter 3. Cropland is not used to cultivate algae in the *BT16* volume 1 scenarios, so the probability that algae production would lead to the transition of forest or other land to cropland is low. Pastureland is not excluded from the transitions in *BT16* volume 1, however. If protein coproducts were produced, algae production could allay potential concerns about food-related LUC, because land area required to produce protein could be reduced.

Table 12.3. | Factors Affecting EROI of Production and Dewatering of Algal Biomass

Infrastructure	Installation of ponds
	Geometry of pond/raceway (e.g., baffles)
	Pond liner[1]
	Mixing method (e.g., paddlewheel assumptions, airlift pond circulation)
	Number, type, and size of pumps or gravity-fed volume transfers
Resource Requirements	Fertilizer (embodied energy, recycling)
	Source and purity of CO_2 and distance to source (e.g., flue gas)[1]
	Technology for purifying CO_2
	Wastewater use
	Rate of sparging of CO_2
	Source of water and delivery (drilled wells or pipeline)
Cultivation	Areal growth rate, including improvement by species selection, genetic modification or enhanced growth conditions[1]
	Algal strain—lipid composition and properties, such as ability to settle[1]
	Temperature control system
	Use of artificial lighting at night (or not)
	Storage of flue gas (or not)
	Recirculation of water
Processing	Pre-harvesting with settling ponds
	Harvesting (e.g., filtration, flocculation, flocculant choice, centrifugation)
	Dewatering, drying (including source of heat)
Other Energy Credits	Quantity and type of coproducts, if included in system boundary
	Wastewater treatment credits (and aeration energy offsets)

Table is modified from table 4 in Efroymson et al. (2016), which includes the references.
[1] Variable in *BT16* algal biomass scenarios

12.4.1.7 GHG Emissions and the Supply Chain

The discussion above, as well as the scenarios in *BT16* volume 1, pertains to biomass potential from cultivation and dewatering. Including the whole supply chain would allow more complete consideration of environmental indicators such as CO_2e throughout the life cycle. Compared to terrestrial biomass, algae cultivation is more tightly integrated with downstream fuel production processes. The purpose of this chapter is not to review life-cycle analyses; however, some early GHG life-cycle analyses for algal biofuels were reviewed by the NRC (2012).

To include downstream processes involves making assumptions about later steps in the supply chain, such as (1) assuming a conversion process, fate of waste products, or target fuel quantity, as in the U.S. Department of Energy's (DOE's) "design cases," or (2) assuming particular technical and economic criteria for "state of technology" analyses.

For example, Argonne National Laboratory conducted supply-chain life-cycle analyses for a hydrothermal liquefaction and upgrading conversion pathway to estimate seasonal energy use and GHG emissions associated with renewable diesel production (Pegallapati et al. 2015). The material and energy intensity of the biomass-conversion step was taken from Jones et al. (2014). GHG emissions were from fuel combustion, fertilizer production (used for cultivation), energy for pumping biomass from the harvesting process to the biorefinery, and other processes. Conversion was assumed to be co-located with cultivation, which is the norm for algae, so transportation fuel to the conversion facility was not needed. These emissions are sensitive to how emissions are allocated to coproducts in life-cycle analyses (Wang et al. 2011). The model used was a version of the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model used for GHG analysis of terrestrial feedstocks in chapter 4 of this volume. The conversion facility was sized based on an average algae productivity of 30 g/m^2/day, which is close to that of the future-productivity scenarios in *BT16* volume 1. This study concluded that algae renewable diesel has lower GHG emissions, fossil energy use, and petroleum use than does petroleum diesel under the conditions in that report, which included the use of flue gas, rather than captured CO_2 (Pegallapati et al. 2015). Further analyses by Argonne National Laboratory (Frank et al. 2016) for lipid extraction and thermal processing pathways used in three scenarios showed that GHG emissions satisfied the RFS2 for advanced biofuels only when flue gas transported under low pressure was the source of CO_2, as in *BT16*.

In the Davis et al. (2016) base case for algae cultivation in ponds that was described earlier in this section, the authors describe recycle and consequent GHG benefits that could accrue if the downstream conversion process were integrated with the cultivation process. These include the following:

- "Recycle of a fraction of CO_2 anywhere between 10%–40% (depending on downstream conversion steps and yields), which would reduce incoming makeup CO_2 demands by a similar fraction."
- "Recycle of a substantial fraction of nutrients on the order of 50% P and 50%–90% N (dependent on similar factors as noted for CO_2)."
- "In some cases, the generation of a net electricity coproduct in the downstream conversion facility would partially offset the power demand . . . for the biomass production facility."

The fate of the algal biomass is important when estimating GHG emissions. Remaining biomass after oil extraction (in the lipid-extraction pathway) can be used for products such as fertilizer or animal feed. Biomass can be fermented to produce ethanol or pyrolyzed to create oil, gas, and char. Whether the whole biomass is directly converted to biocrude via hydrothermal liquefaction (Elliott et al. 2015) or lipid-extracted biomass is anaerobically digested to produce CH_4 for electricity production, the resulting digestate can be recycled to provide microalgae-required nutrients or applied as a soil treatment. Frank et al. (2012) found that GHG life-cycle analyses were highly dependent on biogas-production parameters, including "yields from digesters, yields from gasification, fugitive emissions, nutrient recovery rates, and electrical efficiency of the [combined heat and power] generator." CH_4 and N_2O emissions contributed substantially to total GHG emissions when the anaerobic digestate was applied to soil. Luo et al. (2010) assumed that annual disposal of cyanobacteria biomass would be via deep well injection, which could result in a slight net GHG reduction for the PBR system.

Two algal biofuel pathways that involve the use of cyanobacteria as catalysts that secrete ethanol in PBRs were approved by EPA for generation of advanced biofuel Renewable Identification Numbers under RFS2. These pathways are not pertinent to the open pond/raceway cultivation systems that were the focus of *BT16* volume 1, but they are relevant to this discussion of supply chain strategies to reduce GHG emissions. The pathways include the Helioculture Sunflow-E ethanol process of Joule Unlimited

Technologies, Inc., which is estimated to reduce life-cycle GHG emissions compared to the statutory petroleum baseline by 85% (Grundler 2016), and the Algenol Direct-to-Ethanol Process, which is estimated to reduce life-cycle GHG emissions compared to the statutory petroleum baseline by 69% (Grundler 2014).

12.4.2 Water Quantity

As is noted in chapters 7 and 8, freshwater availability is declining in some regions as a result of increased water demand for irrigation, power generation, and domestic water use, in part because of a growing population, and partly as a result of altered climate patterns. Tracking water resource use for algal biomass cultivation is the first step toward determining effects on water availability and water-related effects. This section discusses water consumption, as well as indicators and methods that place that consumption in the context of competitive uses of water and the regional environment.

Most commonly, microalgae feedstock production occurs in open raceway ponds that can be operated with either fresh or saline water sources, depending on the algal strain. These are relatively shallow ponds (30 cm deep in the scenarios in this chapter) with a large surface-to-depth ratio designed to maximize capture of sunlight and minimize "dark zones" that may result in loss of biomass through dark respiration. Cooling of the open ponds is generally achieved through evaporation (a significant source of consumptive water use), and warming occurs through solar-radiation inputs. The thermal mass of water and surrounding soil provide some buffering against rapid changes in pond temperature; thus, pond temperature fluxes are not as rapid as changes in air temperature. A complex relationship exists between the pond water temperature, hourly meteorological data, optimal operating conditions for maximal biomass productivity, and evaporative loss of water.

Environmental indicators include water quantity indicators and water quality indicators. Water quantity indicators are emphasized because (1) the scenarios in *BT16* volume 1 tracked water quantity, and (2) more research has focused on effects of algae production on water quantity than on potential impacts to water quality.

Consumptive water use from pond systems is affected by algae cultivation operations and varies geographically. Cultivation systems with summertime high temperatures and low humidity have higher rates of evaporation, greater pond cooling, and consequently, higher rates of water consumption. Alternatively, cultivation systems located in regions with high summertime temperatures and high humidity have lower evaporation rates, less water use, and limited capacity to cool. The selection of algal strains that can operate under site- or region-specific, seasonal environmental conditions can provide for more favorable ratios of biomass production to water consumed.

Consumptive water use is a system-specific indicator that alone does not capture local availability and competing uses (NRC 2012). Peak storm flow and minimum base flow (table 12.1) are ecosystem-related indicators of water quantity. Competing uses are discussed below, as well as more complex methods for assessing water-quantity-related effects that integrate local availability and environmental water requirements with system withdrawals and use.

This section on water quantity describes water consumption that was estimated in the biomass production modeling in *BT16* volume 1. Water consumption estimates are placed in the context of the variables in the scenarios, namely, productivity, freshwater vs. saltwater, and full liner vs. minimal liner. The source of co-located CO_2 is not thought to influence water withdrawals or consumption. Water consumption is discussed in the context of competitive use and regional availability.

Water is a regional resource. Therefore, the discussion of the environmental effects of algal biomass production with respect to water includes a discussion of the national context of water use and water availability. This emphasis contrasts with that of GHG emissions. While the regional environment can affect net GHG emissions (e.g., via land management–related changes in soil organic carbon and temperature effects on CO_2 use), the implications of GHG emissions from a region are global. The effects of algae cultivation systems with respect to existing water sources, competing use, future demand, and water quality are regionally variable.

12.4.2.1 The Context of Water Use

The increased interdependencies of energy, socioeconomic variables, and environment related to available water resources are magnified by higher variability in inter-annual climate, extreme events, non-stationarity, and spatiotemporal migrations of climate (Skaggs et al. 2012). Within the energy-production domain alone, demand for water resources is rapidly growing, as various types of energy production (thermoelectric, hydroelectric, hydraulic fracturing for natural gas, bioenergy, coal, etc.) continue to increase (DOE 2006; Bauer et al. 2014; McMahon and Price 2011). Water use for algae production needs to be evaluated in this larger context of the food-energy-water nexus, particularly since freshwater withdrawals for agriculture represent about 32% of all freshwater withdrawals, and thermoelectric power generation represents about 45%, totaling about 78% for these two sectors (Maupin et al. 2014). Between conveyance and consumptive use by the crops, irrigated agriculture (fresh surface water and groundwater) has the highest rates of national water use, where ~80% of water withdrawn is consumed, and 20% is provided as return flows (Solley et al. 1998). Wigmosta et al. (2011) review the water intensity of transportation fuels in volume of water per distance driven, based largely on King and Webber (2008). Biologically based transportation fuels typically consume much more water than petroleum-based gasoline.

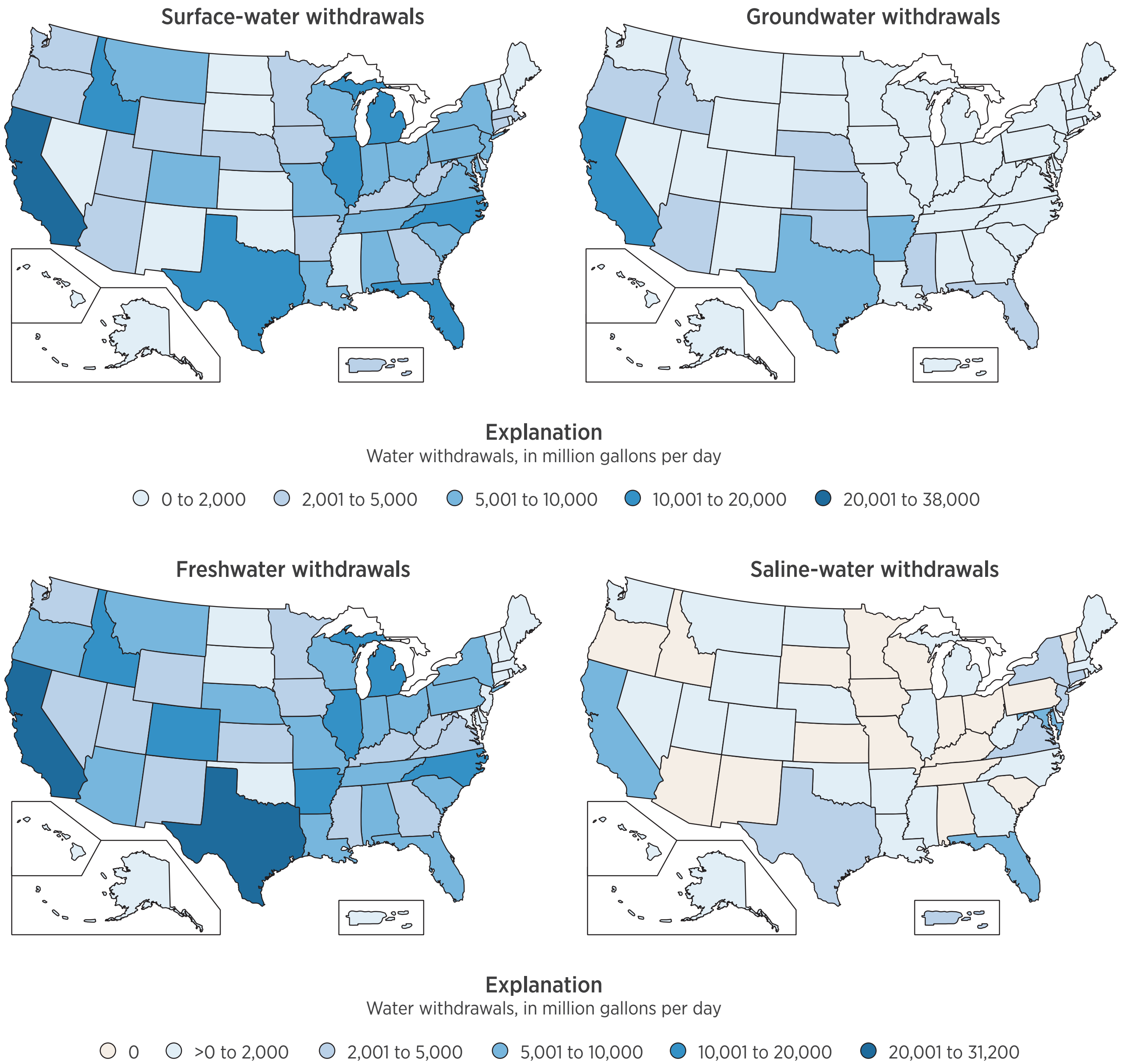
In the context of water-resource competition, Moore et al. (2015) note that "…available water resources [are] understood by evaluating the quantity, timing, and spatial distribution of water availability and use. The location and timing at which water is available and consumed dominantly affects the extent to which not only energy and water influence one another, but also the greater cross-sector dependencies that, for example, influence agriculture, industry, environment, economics, and social well-being."

To help describe the water resource landscape, the USGS produces a county-scale U.S. water-use report at 5-year intervals and provides sector-specific water-use information, including irrigation/agriculture, domestic, industrial, thermoelectric, livestock, and mining sectors (Kenny 2009; Solley et al. 1998; Maupin et al. 2014). It is important to note that after 1995, these reports only include water withdrawals and not consumptive water use. The distinction is critical, as described above. To provide context on surface and groundwater use for both fresh and saline water, Maupin et al. (2014) provide a state-level look of average daily withdrawals from 2005–2010 (fig. 12.3).

Maupin et al. (2014) estimated that for average water withdrawals over 2005–2010 (surface and groundwater), 86% were freshwater sources, and the remaining 14% were saline. Of the total withdrawals, 78% of the water resources (freshwater or saline) came from surface water resources (84% freshwater, 16% saline). Groundwater sources represented 22% of total withdrawals with 96% being freshwater and 4% saline. States with the nation's majority of withdrawals include California, Texas, Idaho, Florida, Illinois, North Carolina, Arkansas, Colorado, Michigan, New York, Alabama, and Ohio.

Appendix 12-C shows the fractional contributions of sectors to total consumptive water use. Most competitive uses are for freshwater; competitive use of the saline water supply is primarily related to thermoelectric power plants.

Figure 12.3. | The USGS provides a state-level look at withdrawals of fresh and saline water for both surface and groundwater sources averaged during 2005–2010 (Image credit: Maupin et al. 2014).



12.4.2.2 Water Consumption in Scenarios from BT16 Volume 1

Saline versus Freshwater—Qualitative Discussion

Both saline and freshwater scenarios were included in *BT16* volume 1. Differences in the use and operations of freshwater and saline water result in tradeoffs between the two types of sources. Saline and freshwater sources differ with respect to availability, access and transport cost, competitive use, maintenance, and disposal needs and costs.

Sources of freshwater are generally easier and less expensive to access than saline water sources. Saline resources are not as well characterized as freshwater resources. The last nationwide saline water assessment was conducted by the U.S. Geological Survey (USGS) in 1965 (Feth et al. 1965). More recent characterization of these water sources has been supported by carbon-sequestration efforts, though the emphasis has been on very deep saline groundwater reservoirs (>800 m depth) that would not be economically viable as a source for algae production (Venteris et al. 2013).

Freshwater has high competitive use and is in limited supply in many parts of the country; however, saline water sources (brackish, saline, and hypersaline) from groundwater or seawater are abundant resources with lower competitive use than freshwater but typically require more energy to transport from source to pond. In addition, the ion chemistry in saline groundwater is highly variable, and sources need to be screened for toxicants and composition compatible with specific algal strains (Venteris et al. 2013). For example, a total dissolved solids (i.e., salinity) characterization of existing, produced water (i.e. oil and gas) wells in the conterminous U.S. is shown in figure 12.4, where variability in salinity ranges from 1 to 400 practical salinity units (PSU) (1,000–400,000 mg/L). Seawater ion chemistry is more consistent with salinity between 33 and 37 g/kg.

For freshwater open-pond cultivation systems, the water systems must be maintained to compensate for net losses (evaporation minus precipitation); whereas, for saline open-pond systems, water is maintained not only for the volume of water, but more importantly, to maintain the salinity required by the cultivated algal strain (see fig. 12.5). Freshwater open ponds have a higher rate of evaporation than do saline sources; however, freshwater can be treated and recycled for further use, reducing pumping costs associated with bringing clean water to the site. In order to maintain salinity targets, saline pond systems can require 2-3 times the amount of withdrawn water that freshwater ponds require, depending upon the salinity target, source water salinity, and local meteorological conditions (Venteris et al. 2013). Note that this water use is not required to replace evaporative loss, but rather, for pond blowdown, where a fraction of the pond water is discharged and replaced with new water to keep an ideal pond operating environment for the strain of interest.

Saline water concentrate must be disposed after blowdown. The most commonly considered option for saline groundwater is through re-injection wells and for seawater is a marine outfall that may or may not be the same pipe construction that draws in source water. Other alternative saline-disposal options exist, including evaporation ponds and landfill-style disposal of residual salt, discharge to a water treatment facility, oil well-field injection for secondary oil and gas recovery, solar ponds, aquaculture, wetland creation or restoration, and high-value salt and chemical products (GEO-Processors USA 2006; Jordahl 2006; Mickley 2001).

In general, the use of saline water resources comes with an increased capital and operational expense, as compared to freshwater. An example is the \$32 minimum selling price per ton of biomass added for blowdown waste disposal for saline systems in *BT16* volume 1 and Davis et al. (2016). Other increased capital and operating expenses for the use of saline resources (e.g., deeper wells to access water, corrosion-tolerant construction materials) were not considered in *BT16* volume 1. From the perspective of production facility siting, managing for salinity concentrations can increase operational expenses.

From the perspective of operations and operational expenses for saline water sources, the relationship of salinity concentration to water use (evaporative loss and blowdown) affects the quantity of water that needs to be moved to the site (fig. 12.5). Higher pond target salinities require less blowdown and lower inputs of new saline water. The salinity of the water source is also important; a low salinity water source requires less blowdown and lower inputs of new saline water to maintain pond target concentrations.

Figure 12.4. | Oil and gas wells with associated total dissolved solids measurements from produced water. These well locations do not include recent hydraulic fracturing wells. (Data source: Breit 2002).

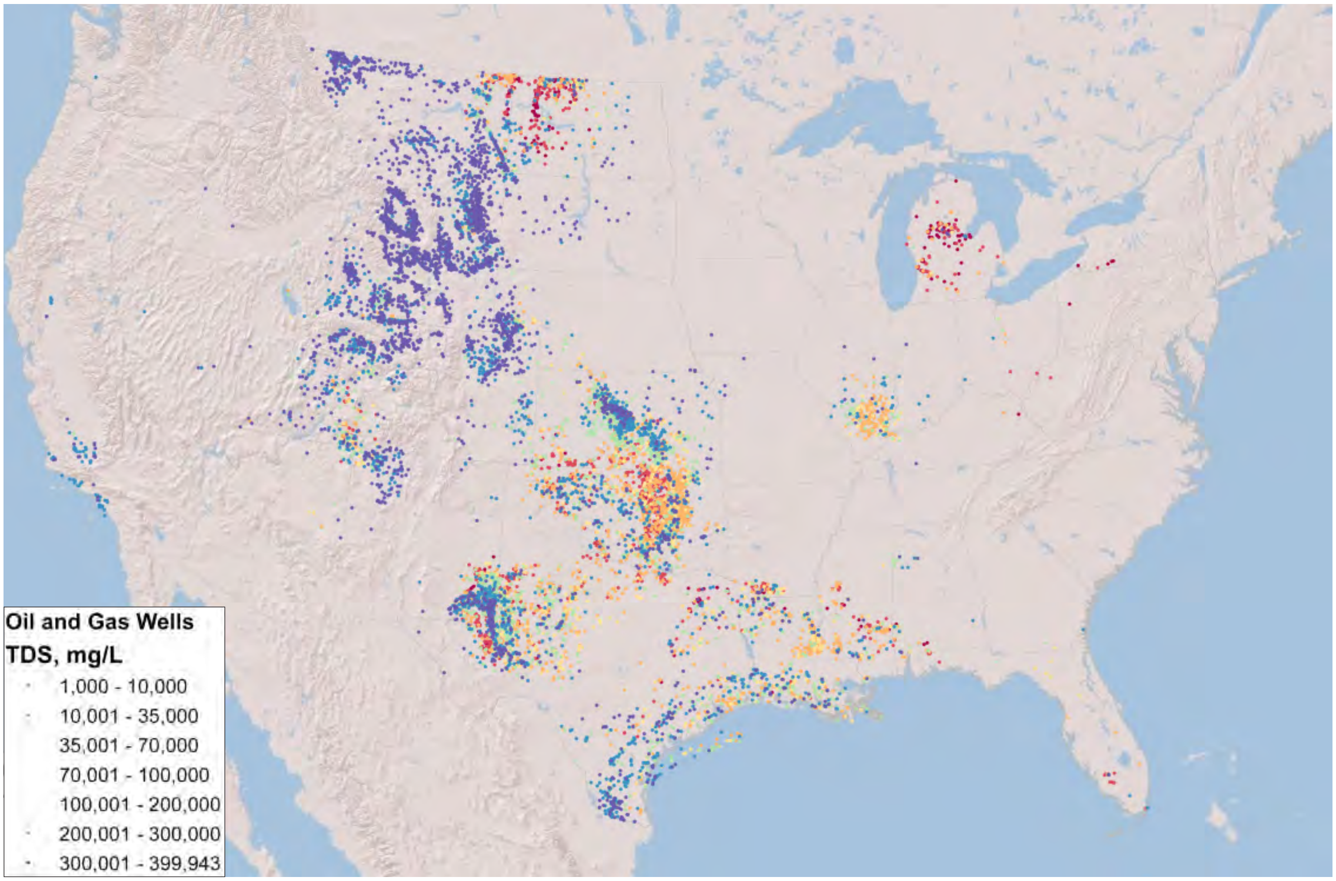
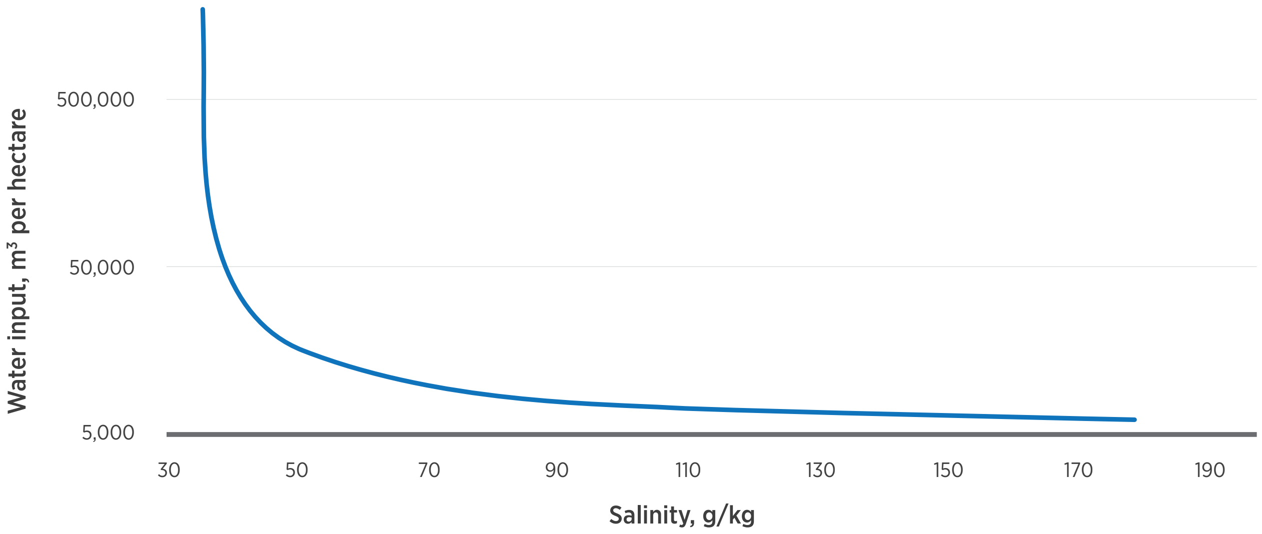


Figure 12.5. | Modeled relationship of water input to maintain a target open-pond salinity, considering salinity of the water source, blowdown and evaporative loss in Tucson, Arizona. Site-specific curves such as this are required for operational planning in order to maintain a target salinity for the cultivated algal strain.



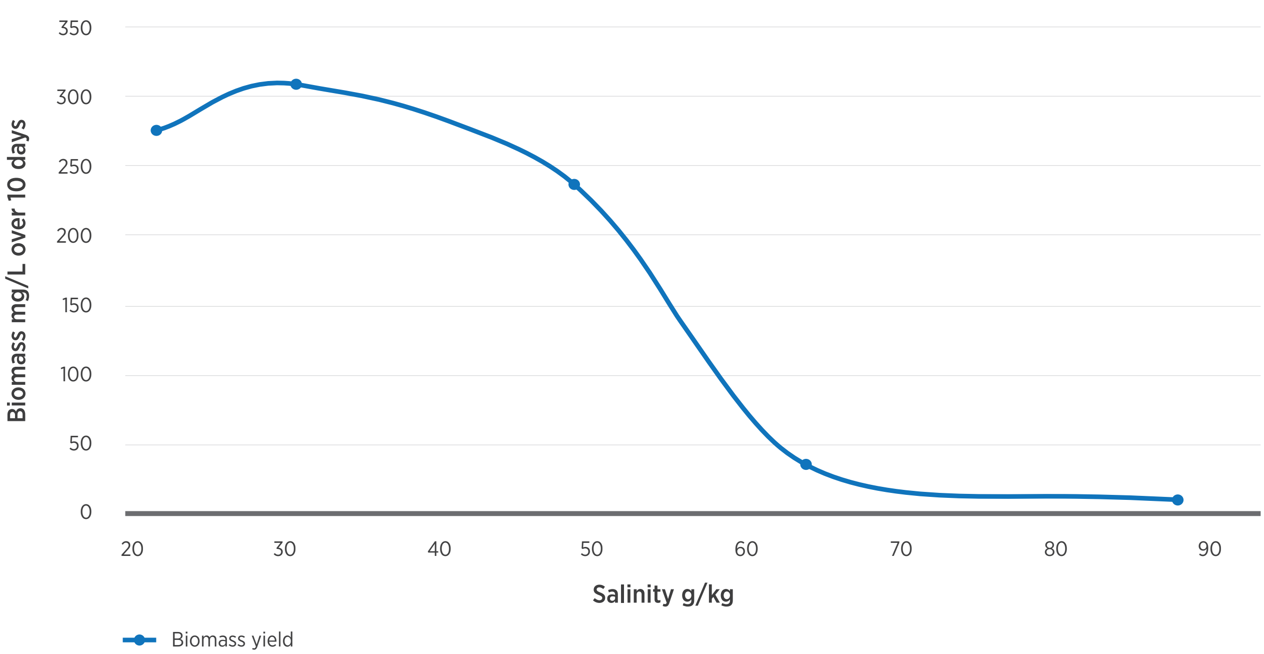
It should be noted that the exact form of the quantitative relationship shown in figure 12.5 is site-specific and varies depending on location and local climate conditions. Also, different strains of microalgae have varying tolerances for salinity concentrations and optimal ranges that provide opportunity for the highest growth rates. Figure 12.6 illustrates how careful management of saline algal strains is imperative; an optimal salinity of ~30 g/kg provides the highest productivity rate for *Nannochloropsis salina,* which then decreases with an increase in salinity concentration.

In Venteris et al. (2013), a water trade-off analysis is conducted that considers the ratio of algae-produced bio-oil to total consumptive water use required in cultivating the algal biomass; water availability of seawater, saline groundwater and freshwater; and the costs associated with delivering each type of water to potentially suitable open-pond sites identified in Wigmosta et al. (2011). In Venteris et al. (2013) potential sites were screened for cost-effectiveness targets for water delivery. Delivery could not exceed 20%, 10%, or 5% of a $2.90/gallon wholesale renewable diesel cost, targets which would represent the most water-efficient and cost-efficient sites within the conterminous United States. In the majority of cases, freshwater was the least-expensive source of water, but this resource is often subject to a high degree of competitive use. Saline groundwater was often the next most-economical water source, due to shorter transport distances than from seawater-based sources.

Saline versus Freshwater—Water Consumption

The CO_2 co-location scenarios developed in *BT16* volume 1 included consumptive water use simulations of freshwater and blowdown requirements for saline water as part of the physics-based mass and energy balance models that predict biomass growth (Wigmosta et al. 2011). For the freshwater scenarios, the consumptive water use was tracked as evaporative water loss from the open ponds and total loss from a unit farm. For the saline water scenarios, following the procedure in Venteris et al. (2013), the pond salinity was set to a concentration of 60 g/kg; as water evaporated, pond salinity increased, and thus, a given quantity of pond water was expelled (as blow-

Figure 12.6. | Relationship between biomass productivity and salinity concentration for *Nannochloropsis salina*



down), and new water was brought in to maintain the required salinity. The blowdown rates are variable by site, as the source water salinity and meteorology are site-specific. Water use for freshwater is focused on consumptive use (evaporative loss), and water use for saline water is more focused on the movement of and use of water for blowdown requirements.

In this chapter, consumptive water use for both saline water and freshwater sites co-located with existing waste CO_2 sources (ethanol production plants, coal power plants, natural gas power plants) are reported through the use of national maps, rather than by reporting quantitative totals for each site and scenario. The intent is to provide the reader with a qualitative understanding of consumptive use patterns under different scenarios and locations throughout the country. Each figure set is organized to illustrate peak consumptive water use (summer months, top figure) and annual average rates (bottom figure), which can differ significantly. The following scenarios are presented (table 12.4).

The most notable pattern across all scenarios is in the difference in water use between the average hourly summertime use and average hourly use across the year. This is particularly evident in the western states and, to a lesser extent, in the Midwest, whereas areas along the Gulf Coast, southeastern states, eastern seaboard, and Great Lakes regions have a smaller difference of water use seasonally. The regional and seasonal differences in consumptive use are driven by notable differences in the climate, where relative humidity plays a significant role in evaporative water loss.

The constraint on freshwater use had differing outcomes, depending on the co-location scenario. Overall, instituting the 5% mean annual flow rule for freshwater sites did not significantly impact the number of potential algal production sites available under the ethanol production and natural gas power plant co-location scenarios. In these cases, the physical co-location with an economically viable waste CO_2 source was the most significant limiting factor. However, because of the large number of coal-fired power plants available for co-location, under the freshwater scenario, potential sites were in fact limited as compared to sites that were sourced with saline water, which did not have explicit constraints on total use. This was most notable in the Gulf Coast region, southeastern states, and eastern seaboard.

For saline water, the source salinity constraints were broad in this analysis and as such, potential algal production sites were rarely excluded based on saline

Table 12.4. | List of Illustrative Algae Production Scenarios (combinations of variables) for Which National Consumptive Water Use Is Presented

Figure	CO_2 Co-Location Source	Water Source	Algal Strain
Figure 12.7	Ethanol	Saline	*Nannochloropsis salina*
Figure 12.8	Coal power plant	Saline	*Nannochloropsis salina*
Figure 12.9	Natural gas power plant	Saline	*Nannochloropsis salina*
Figure 12.10	Ethanol	Freshwater	*Chlorella sorokiniana*
Figure 12.11	Coal power plant	Freshwater	*Chlorella sorokiniana*
Figure 12.12	Natural gas power plant	Freshwater	*Chlorella sorokiniana*

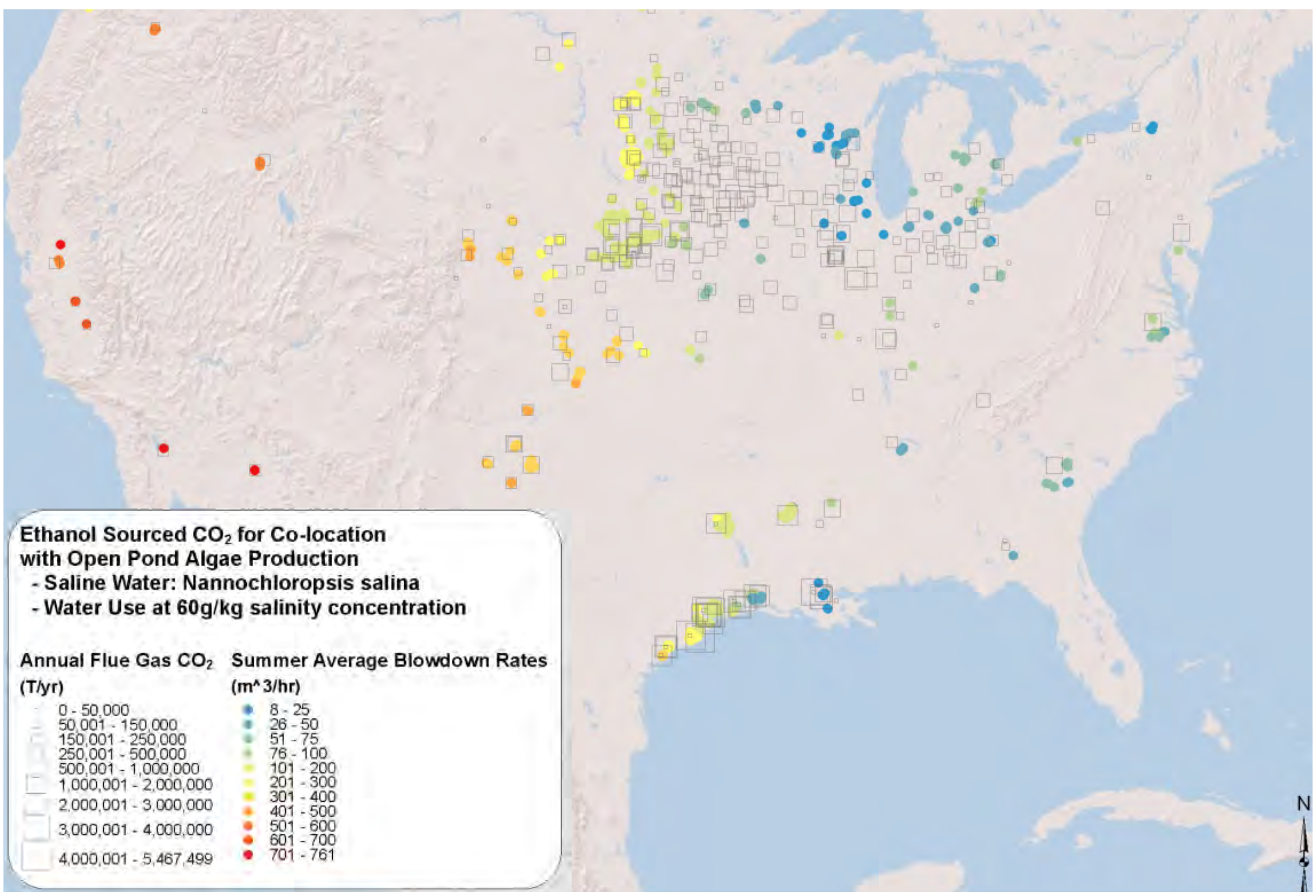
For each scenario above, both summer average hourly blowdown rates and annual average hourly blowdown rates are shown.

water availability, but as with freshwater, co-location to a waste CO_2 source was the primary siting constraint. Of importance with respect to freshwater are the seasonal water scarcity estimates illustrated in figure 12.7 (see also discussion in section 12.5.2.3, Water Consumption and Timing of Supply). Despite the use of the 5% mean annual water use rule, a number of sites across all co-location scenarios—but especially natural gas and coal power plant co-location scenarios—could potentially be impacted by high seasonal water scarcity, particularly under an altered climate. Note, however, that the water scarcity analysis is based on local water scarcity; thus, a major upstream water source (e.g., Missouri River, Mississippi River) could potentially dampen the water scarcity risk. Careful regional evaluation, long-range planning, competitive use, and climate-based risk evaluation, with respect to sustainable water use, is required. With regards to saline water, while best available public data and geostatistical analysis of these data were put forth, there is still a significant degree of uncertainty in the saline water estimates, partly due to uncertainties of geologic formations and high variability in ion chemistry (Venteris et al. 2013). In addition, site-specific assessments to sample saline waters for toxicants are necessary.

Productivity

Under the future productivity scenarios from *BT16* volume 1, a decrease in consumptive water use per unit biomass would be expected during the algae cultivation phase, as more biomass is produced in the same amount of time as under the current productivity scenarios. Increased productivity, however, also means more harvesting, dewatering, and pro-

Figure 12.7 | Blowdown rates for 405-ha saline open ponds co-located with ethanol production plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)



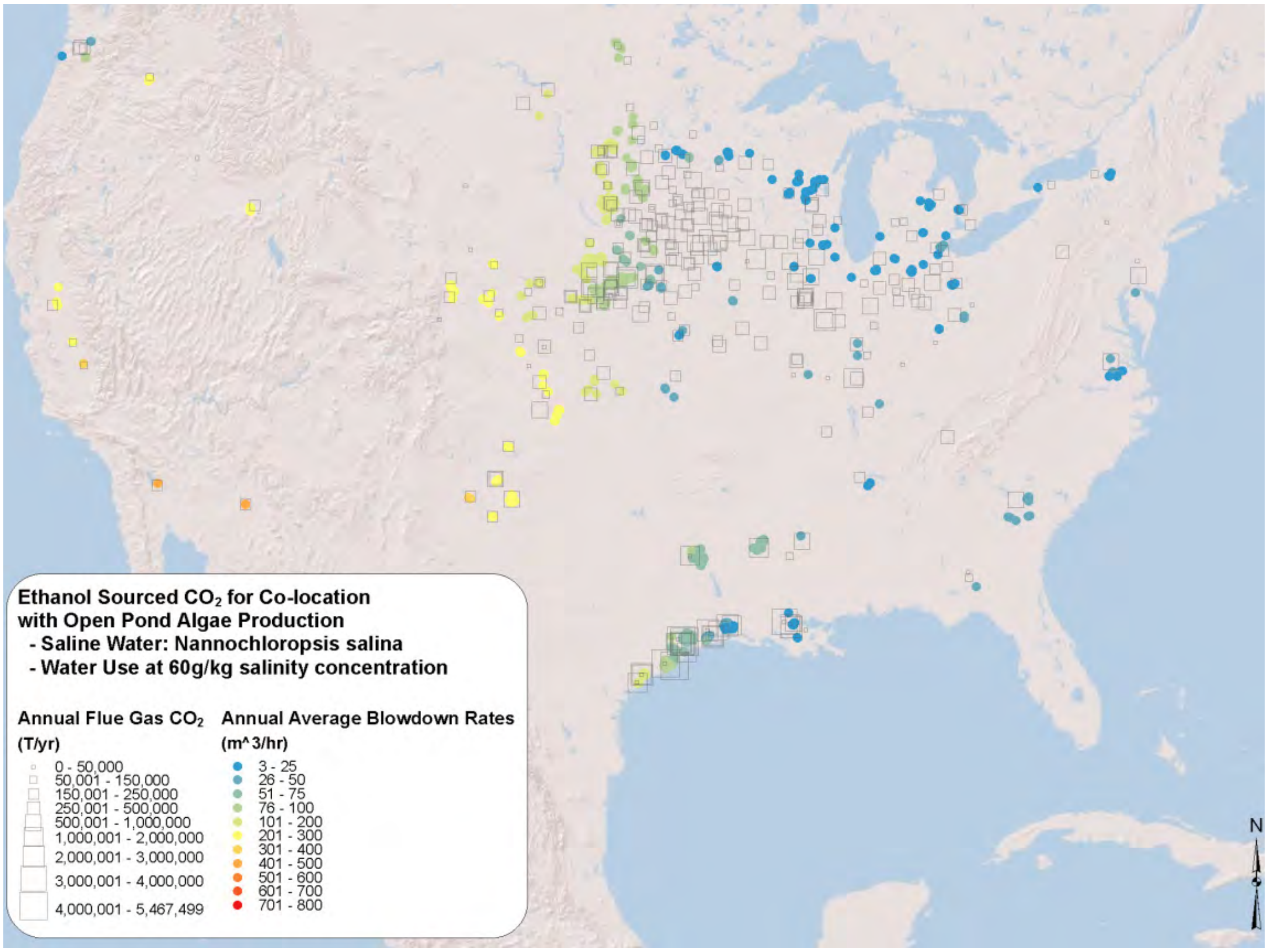

Ethanol Sourced CO_2 for Co-location
with Open Pond Algae Production
- Saline Water: Nannochloropsis salina
- Water Use at 60g/kg salinity concentration
Annual Flue Gas CO_2
(T/yr)
0 - 50,000
50,001 - 150,000
150,001 - 250,000
250,001 - 500,000
500,001 - 1,000,000
1,000,001 - 2,000,000
2,000,001 - 3,000,000
3,000,001 - 4,000,000
4,000,001 - 5,467,499
Annual Average Blowdown Rates
(m^3/hr)
3 - 25
26 - 50
51 - 75
76 - 100
101 - 200
201 - 300
301 - 400
401 - 500
501 - 600
601 - 700
701 - 800
N

Figure 12.8 | Blowdown rates for 405-ha saline open ponds co-located with coal-based power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

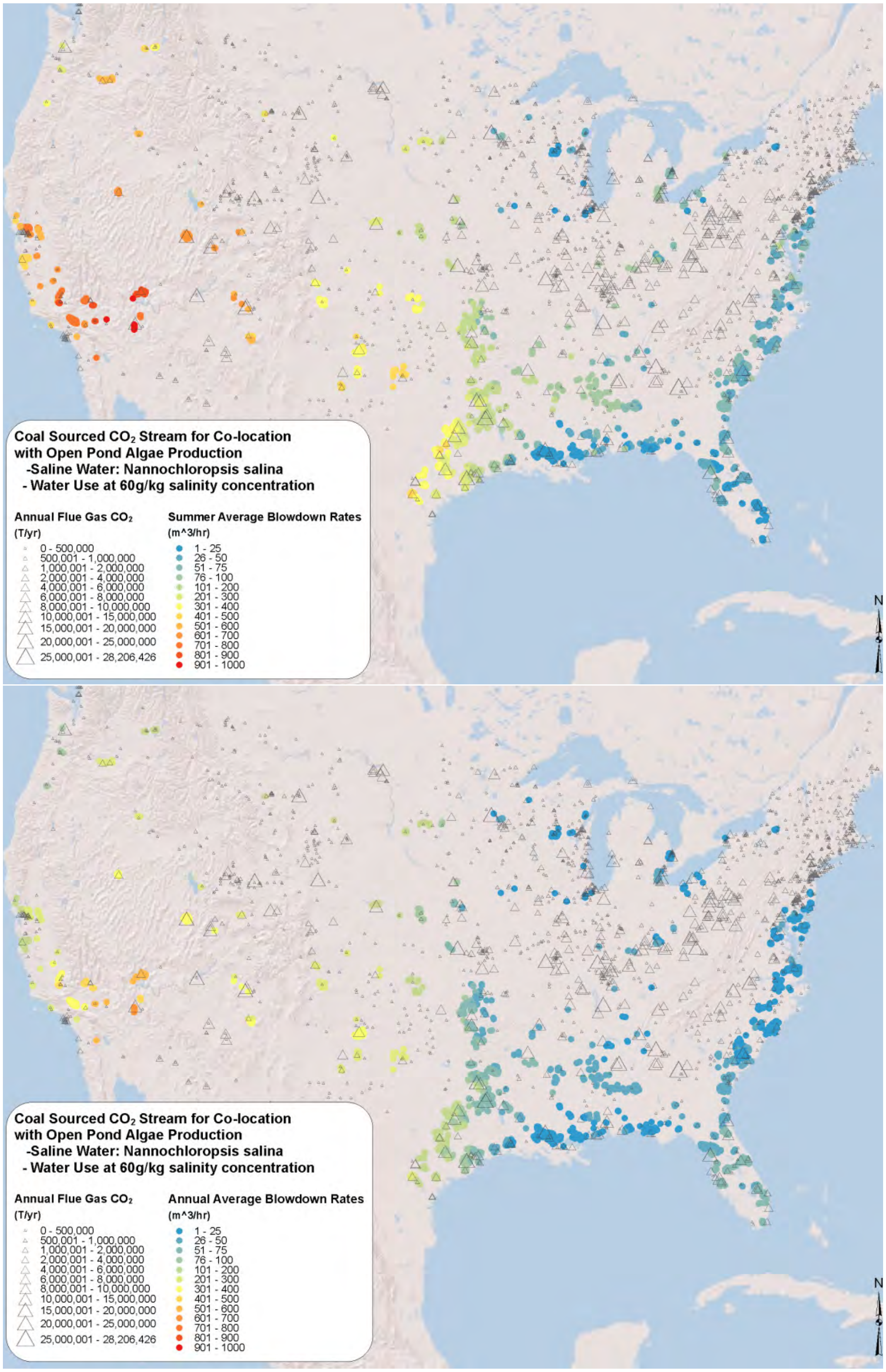


Figure 12.9 | Blowdown rates for 405-ha saline open ponds co-located with natural gas–based power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

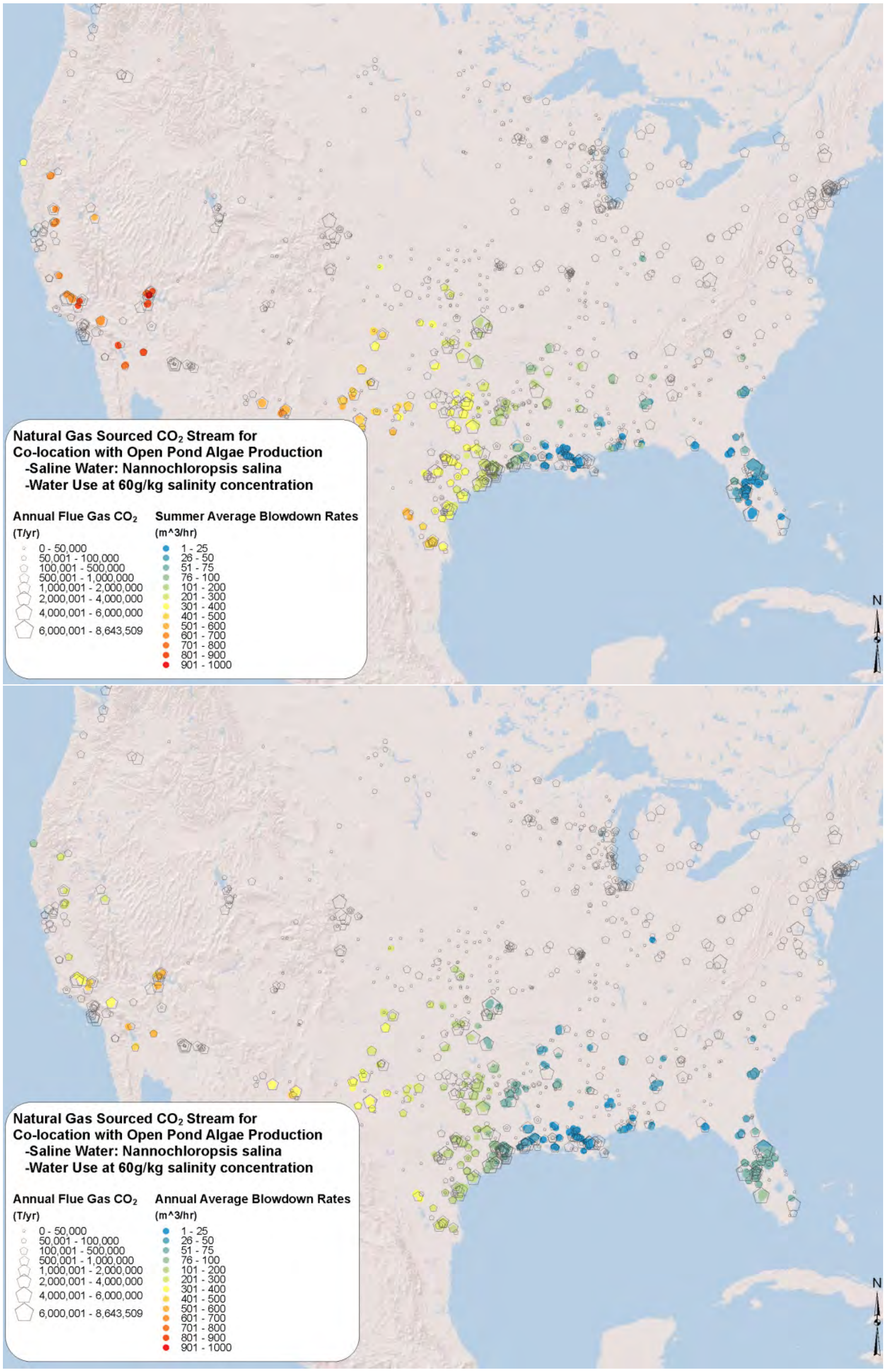


Figure 12.10 | Consumptive freshwater use for 405-ha freshwater open ponds co-located with ethanol production plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

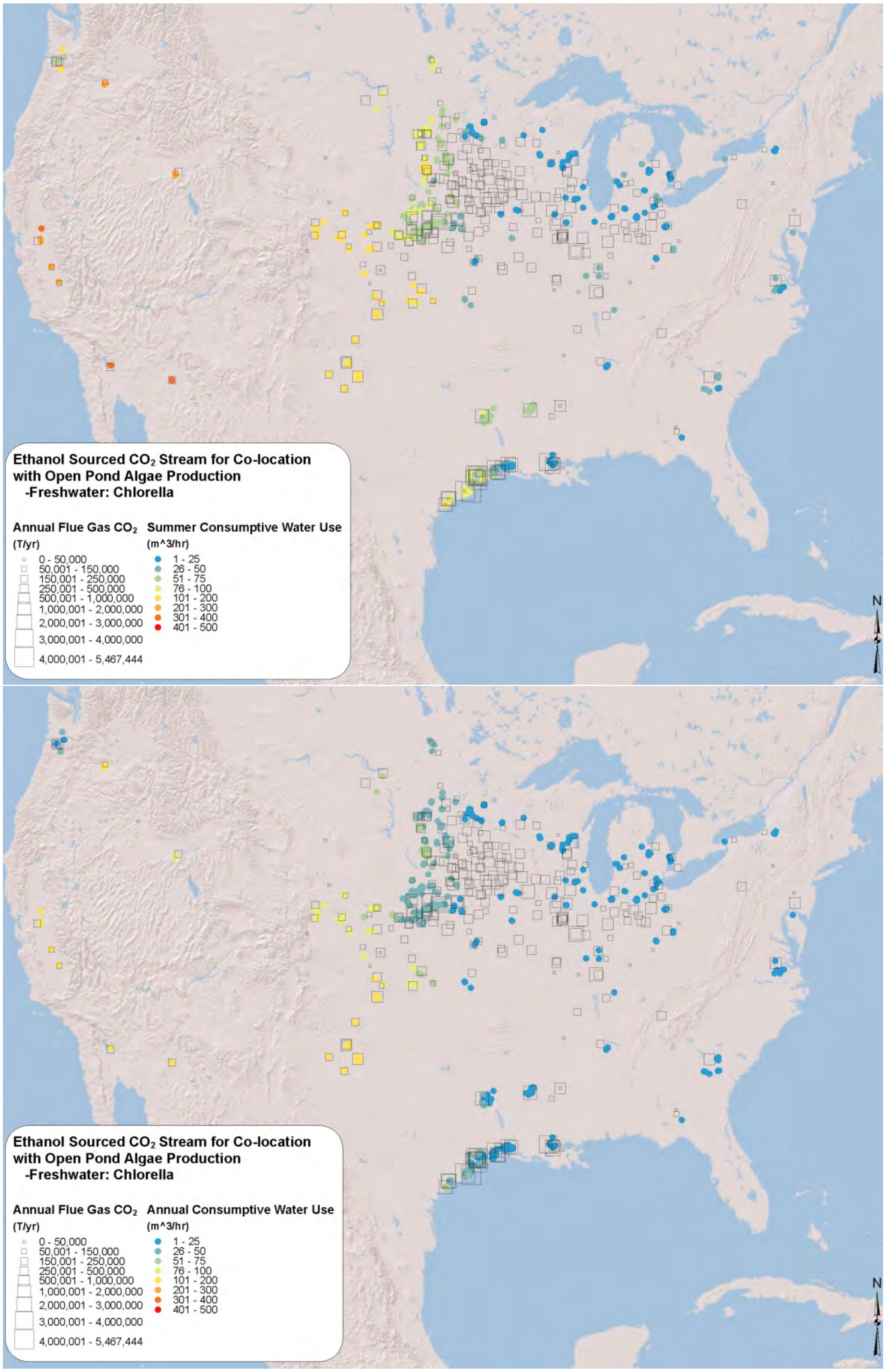


Figure 12.11 | Consumptive freshwater use for 405-ha freshwater open ponds co-located with coal-based power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

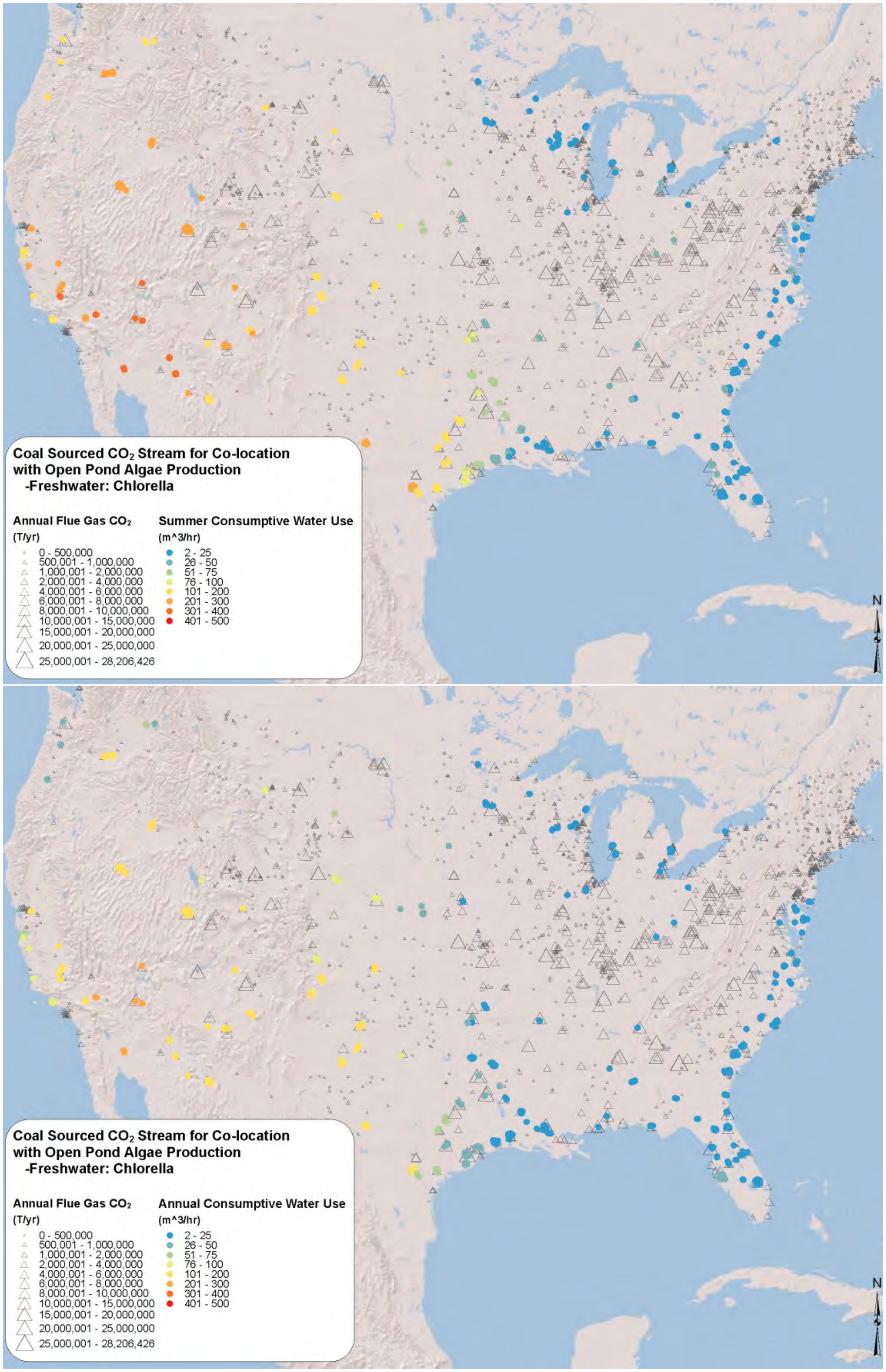
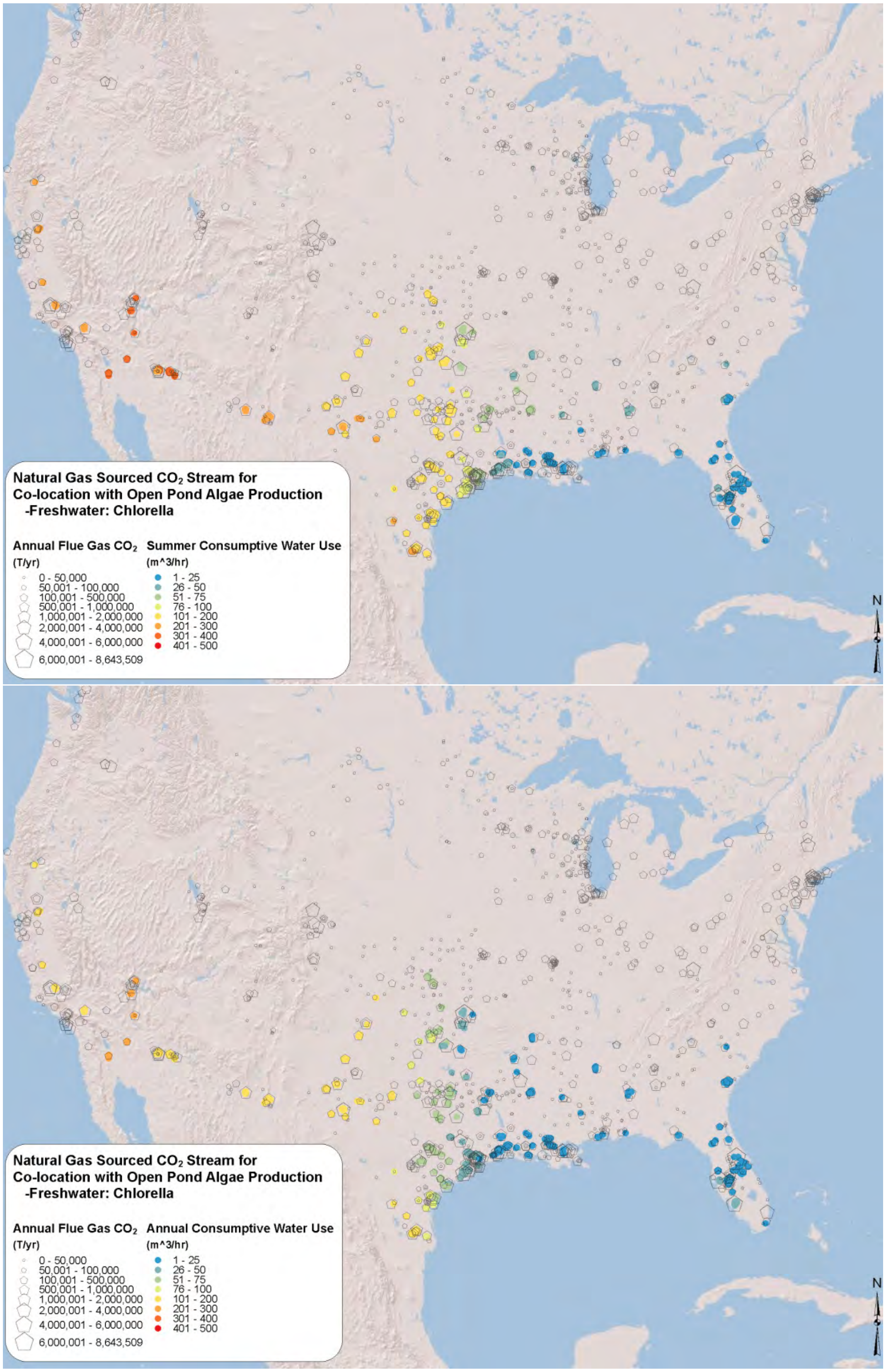


Figure 12.12 | Consumptive freshwater use for 405-ha freshwater open ponds co-located with natural gas power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)



cessing—thus, potentially driving more water losses through the system per unit of time. This largely depends on how a site or an enterprise of sites is configured and whether or not harvested biomass is processed on-site, within the enterprise, or moved at distance, creating "virtual water" (water embedded in a product and exported outside of the basin of origin) and likely allowing limited opportunity for water recycling.

Full Plastic Liners versus Minimal Plastic Liner

The extent of a pond liner may have implications for water quantity and quality. Many commercial- and research-scale algae-cultivation facilities install plastic or clay liners beneath cultivation ponds to prevent or mitigate water loss due to seepage, as well as to control the release of salts and nutrients into the subsurface or groundwater. Lined ponds include plastic liners (e.g., HDPE) that create barriers with very low effective permeability ($<10^{-12}$ cm/s), high longevity, and resistance to chemical and ultraviolet light degradation (Ng 2008).

Yet, liners are expensive (see chapter 7, volume 1 of *BT16*), and are not always needed. Venteris et al. (2014) proposed that natural soil properties, particularly soils with low saturated hydraulic conductivity (Ks), could be used to avoid the costly installation of plastic liners.

Evidence from the laboratory and field illustrates that liners are not always needed if the objective is to avoid leaching of cultivation fluids. The DOE-funded *Aquatic Species Project* tested unlined ponds for 2 years at a site in New Mexico without observable leakage (Brown and Sprague 1992). Studies of animal waste settling ponds and related industrial ponds show that underlying soils can exhibit reduced hydraulic conductivity over time, reducing or eliminating the need for HDPE or clay liners (SNTC 1993). Numerous lab-scale and field-scale studies have demonstrated rapid development of low-conductivity seals in soils beneath animal waste settling ponds (Cihan et al. 2006; Culley and Phillips 1982; Barrington et al. 1987a, 1987b; Rowsell 1985; Hills 1976; Chang et al. 1974). The mechanisms are physical and microbial pore clogging that may occur due to rearrangement of soil particles and growth of microbial biomass and buildup of metabolic products (Barrington et al. 1988; Barrington et al. 1987a, 1987b; Chang et al. 1974).

Vandevivere and Baveye (1992) established that various strains of bacteria differ in growth and metabolic-production rates, resulting in different degrees of decreased hydraulic conductivity of porous media. Numerous studies since the late 1940s indicate two to three order-of-magnitude reductions of saturated hydraulic conductivity can be credited to growth of such bacteria (Thullner 2010).

Soils ranging from commercial-grade "play sand" to clay loam soils develop seals that readily converge to a similar hydraulic conductivity (Cihan et al. 2006; Barrington et al. 1988; Barrington et al. 1987a, 1987b; Cihan et al. 2006) within days to weeks after construction. The soil-plugging process may be insensitive to soil texture (Hills 1976; Culley and Phillips 1982; Rowsell et al. 1985; Barrington et al. 1987a, 1987b). Thus, site discrimination according to hydraulic conductivity (Venteris et al. 2014) may be unnecessary (i.e., many soils can exhibit reduced hydraulic conductivity at $<10^{-7}$ cm/s).

For soil liners containing most types of wastes, hydraulic conductivities of $1x10^{-7}$ cm/s are required to prevent leaking of nutrients or contaminants into the subsurface (Daniel and Benson 1990). Best management practices are needed to achieve this water quantity (and water quality) target.

Research is needed to develop a mechanistic understanding of the processes that seal soils. Research is also needed to provide an experimental basis for understanding the characteristics of soils that will and will not seal sufficiently to allow unlined or minimally lined ponds.

12.4.2.3 Water Consumption in the Context of Agricultural Crops

Water consumption values need to be considered in the context of competing uses and regional availability. Here we consider the water requirements of agricultural crops.

A method of comparing water use between traditional agricultural crops and cultivated microalgae is made by considering the water use per mass of crop yield, i.e., the water footprint, consistently across the United States. Both precipitation or rainfall (sometimes termed "green water") and water withdrawn from surface and/or groundwater sources (sometimes termed "blue water") are presented for 11 terrestrial crops, varying from oil seed crops, to grains, to nuts, to the modeled freshwater algal strain, *Chlorella sorokiniana*. The water footprint for microalgae captures long-term annual total evaporative loss from the modeled 30-cm deep open pond and assumes 85% of pond water removed during harvest is recycled back to the pond. Methods and assumptions for modeling algae production are described above. Water use data were converted from units of L/ha to m^3/ha, and modeled biomass was converted from kg/ha to tons/ha to allow for comparison with terrestrial crops.

The water use data for terrestrial crops are sourced from Mekonnen and Hoekstra (2011) and are derived using a grid-based soil-water balance model and calculation of crop- and location-specific evapotranspiration (ET). The total annual water use is divided by the total annual crop yield to achieve a common water volume per mass produced (m^3 'withdrawn surface and/or groundwater' + 'direct precipitation'/ ton of harvested crop/year)[5] that is averaged within a state-level boundary. ET is calculated daily using crop coefficients throughout the growing season considering available soil moisture in the rooting zone, plant growth stage, and meteorology (see Allen et al. 1998; Chapagain and Hoekstra 2004). For this comparison, we are considering crop yield and not necessarily the whole biomass produced. This provides a common unit across crops, since for oil seed crops, only the seeds are used, and for tree nuts, only the nuts are harvested; however, for algal biomass, the whole biomass is harvested. Results are reported in the following units: m^3 'withdrawn surface and/or groundwater' + 'direct precipitation' / ton crop yield. Table 12.5 provides a state-level assessment of annual average 'withdrawn surface and/or groundwater' + 'direct precipitation' crop water use by state for 11 common terrestrial crops and freshwater microalgae. Appendix 12-D provides state-level maps to visualize these results.

With the assumptions in the analysis, such as water recycle, microalgae consumptive water use per biomass yield ranges from 5–953 m^3/ton, which is favorable in comparison to several other crops, where consumptive water use per crop yield is generally higher across the U.S., such as sunflower (2,615–4,265 m^3/ton), rye (2,041–4,265 m^3/ton), and rapeseed/canola (519–2,899 m^3/ton). Further analysis of water use would consider a selection of bioenergy-potential crops where the whole biomass is used and converted to an end product (ethanol or biodiesel) where the water consumption per energy unit can be assessed. Algae may be a high-quality source of proteins. Water consumption for algae may be more favorable if effects of food and fuel are considered together.

12.4.2.4 Water Consumption and Timing of Supply

The consideration of water-resource availability with respect to timing of supply and demand is important for evaluating competitive use. Many water-scarcity and water-stress indices (see below) are measured at the annual scale, which overlooks the critical monthly-to-seasonal aspects of the systems, particularly regarding the large allocation of consumptive water use to the agricultural sector during the growing

[5] Withdrawn surface and/or groundwater in this study is equivalent to blue water; direct precipitation in this study is equivalent to green water in Mekonnen and Hoekstra (2011) and other studies.

Table 12.5. | Annual Average 'Direct Precipitation' (Green Water) + 'Withdrawn Surface and/or Groundwater' (Blue Water) Crop Consumptive Water Use by State for 11 Common Terrestrial Crops and 1 Freshwater Microalgae Crop (*Chlorella sorokiniana*)

Green + Blue Water Use (in m^3 water/ton of crop yield)												
State	Corn	Sugarbeet	Soybean	Chickpea	Almonds	Walnuts	Rapeseed/ Canola	Sunflower	Wheat	Rye	Sorghum	Microalgae-Chlorella
Alabama	647	-	1,879	2,086	1,097	-	1,879	-	1,225	3,226	1,097	32
Arizona	851	536	-	-	1,843	-	536	-	1,247	-	1,307	585
Arkansas	617	-	1,940	1,971	1,119	-	1,940	-	1,566	-	1,119	49
California	579	519	-	2,743	1,535	3,419	519	4,275	1,248	2,589	1,015	552
Colorado	734	905	1,994	2,354	2,078	-	2,899	3,889	2,929	3,771	1,173	661
Connecticut	462	-	1,215	1,625	-	-	1,215	-	-	2,699	-	-
Delaware	512	-	1,472	1,834	752	-	1,472	-	1,404	2,886	752	26
Florida	563	-	1,805	2,125	-	-	1,805	-	1,198	3,236	-	35
Georgia	576	-	1,724	2,016	908	-	1,724	-	1,368	3,150	908	32
Idaho	951	915	-	1,952	915	3,660	915	-	2,284	-	-	692
Illinois	584	-	1,618	1,887	870	3,123	1,618	3,289	1,449	3,014	870	39
Indiana	533	-	1,604	1,833	843	3,038	1,604	-	1,375	2,927	843	35
Iowa	554	-	1,617	1,885	948	3,031	1,617	2,970	1,546	2,968	948	83
Kansas	744	-	1,815	-	1,100	-	1,815	4,023	2,119	4,070	1,100	333
Kentucky	584	-	1,602	1,783	915	3,421	1,602	-	1,383	-	915	37
Louisiana	643	-	1,915	2,334	1,047	-	1,915	-	1,367	-	1,047	27
Maine	377	-	949	1,312	-	-	949	-	1,031	2,041	-	14
Maryland	518	-	1,515	1,784	775	3,107	1,515	-	1,471	2,945	775	30
Massachu-setts	457	-	1,220	-	-	-	1,220	-	1,318	2,751	-	26
Michigan	480	535	1,475	1,515	535	2,621	2,010	-	1,394	2,585	-	36
Minnesota	529	576	1,575	1,769	576	-	2,151	2,867	1,216	2,783	-	133
Mississippi	589	-	1,920	2,094	1,033	-	1,920	-	1,473	-	1,033	34
Missouri	618	-	1,678	1,970	992	3,304	1,678	3,579	1,474	2,162	992	51

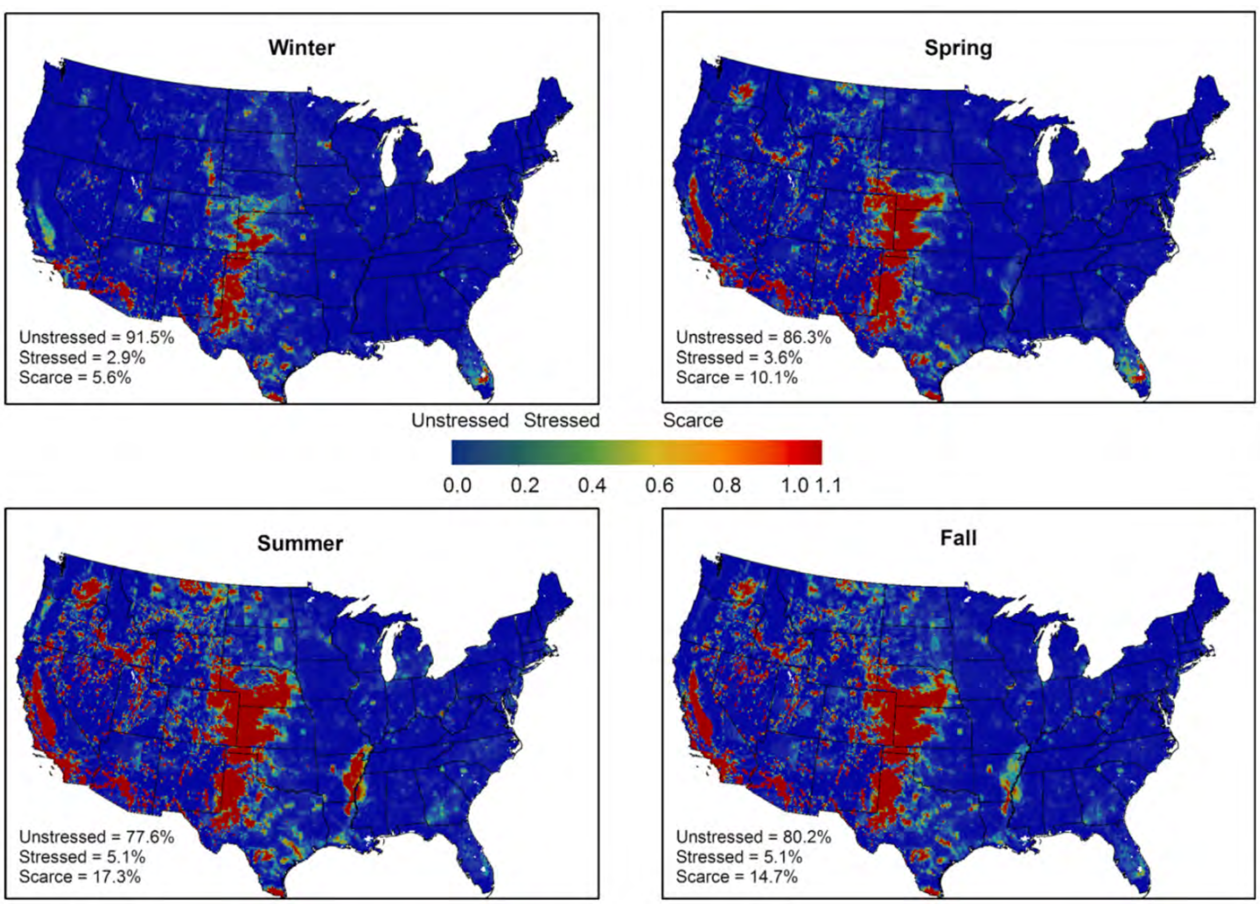
Green + Blue Water Use (in m^3 water/ton of crop yield)												
State	Corn	Sugarbeet	Soybean	Chickpea	Almonds	Walnuts	Rapeseed/ Canola	Sunflower	Wheat	Rye	Sorghum	Microalgae-Chlorella
Montana	596	772	-	1,959	772	-	772	3,386	2,406	3,170	-	623
Nebraska	634	852	1,802	2,127	1,895	3,207	2,654	3,696	2,316	3,403	1,043	289
Nevada	885	-	-	2,513	-	-	-	-	2,659	-	-	923
New Hampshire	394	-	-	1,164	-	-	-	-	-	-	-	14
New Jersey	449	-	1,377	1,773	762	-	1,377	-	1,663	2,929	762	19
New Mexico	793	-	1,830	3,264	1,177	-	1,830	4,046	2,618	4,115	1,177	567
New York	449	-	1,358	1,565	-	-	1,358	2,683	1,373	2,550	-	30
North Caro-lina	532	-	1,600	-	730	3,071	1,600	-	1,426	2,783	730	16
North Dakota	532	597	1,569	1,749	597	-	2,166	2,999	1,257	2,935	-	323
Ohio	493	573	1,495	1,706	1,397	2,799	2,068	-	1,413	2,692	824	23
Oklahoma	771	-	2,224	2,819	1,176	-	2,224	3,845	2,308	4,265	1,176	282
Oregon	690	1,009	-	2,350	1,009	2,702	1,009	-	2,342	3,092	-	662
Pennsylvania	457	-	1,466	1,610	728	2,721	1,466	2,797	1,603	2,742	728	5
Rhode Island	470	-	-	1,599	-	-	-	-	-	2,686	-	-
South Car-olina	549	-	1,701	2,013	760	-	1,701	-	1,336	3,021	760	28
South Dakota	603	-	1,599	1,919	958	3,156	1,599	3,400	1,854	3,129	958	321
Tennessee	652	-	1,708	-	989	-	1,708	-	1,459	3,091	989	23
Texas	810	-	2,002	3,039	1,206	-	2,002	4,276	2,339	2,723	1,206	298
Utah	746	860	-	3,285	860	3,909	860	-	2,608	-	-	863
Vermont	412	-	1,137	-	-	-	1,137	-	1,229	2,317	-	36
Virginia	529	-	1,614	1,869	764	3,052	1,614	-	1,469	2,796	764	28
Washington	636	922	-	2,056	922	3,144	922	-	2,390	-	-	372
West Virginia	442	-	1,450	1,656	-	2,952	1,450	-	1,507	2,694	-	-
Wisconsin	468	-	1,438	1,771	-	2,675	1,438	2,615	1,517	2,578	-	19
Wyoming	656	791	-	2,090	1,792	-	791	3,641	2,902	-	1,001	953

Terrestrial crop water use sourced from Mekonnen and Hoekstra (2011); microalgae water use sourced from Wigmosta et al. (2011)

season. The use of these indices as environmental metrics presents a challenge for evaluating competing water use with respect to algae production, for which the summer months generally have a higher consumptive water use. There are regional exceptions to high summer crop irrigation requirements; for example, in Florida, summer precipitation negates the need for crop irrigation, and the nature of the dominant crops (citrus) requires more irrigation in the fall, winter, and spring. Also, for algae production, higher levels of humidity in the summer months reduce the evaporation loss from open ponds, compared to loss in other seasons.

To highlight the seasonal changes, Moore et al. (2015) calculated local water scarcity based on the water scarcity and the classification method of Sun et al. (2008a, 2008b) (see fig. 12.13), where the following categorizations are defined: <0.2=unstressed; ≥0.2–<0.4=stressed; and ≥0.4=scarce.[6] In the Midwest, the Ogallala Aquifer is consistently in a water-scarce classification, largely due to limited precipitation and large agricultural water use (primarily due to groundwater pumping). Many agricultural regions in the western United States and, notably, the Mississippi River Plain move to a water-scarce condition during the growing season. The cultivation of algal biomass would have to take seasonal water requirements for competitive uses of water into account for siting and planning.

Figure 12.13. Seasonal local water scarcity for the conterminous United States. The indicated percentages of unstressed, stressed, and scarce reflect the fraction of 1/8° cells that fall in one of these three categories (Image credit: Moore et al. 2015).



6 Although Sun et al. (2008a, 2008b) term their method a "water stress index," it is a water scarcity index by the UN definition above, since it considers volumetric supply of water and not water quality, accessibility, and environmental requirements.

12.4.2.5 Water Consumption and the Supply Chain

This assessment herein has focused on the cultivation aspects of microalgae production and not on the water aspects through the full supply chain. This has partly been due to the limits of the study in producing algal biomass to the so-called 'pond gate' and not considering the various downstream extraction, processing, and fuel upgrading pathways for the feedstock. Additional analysis with considerations and variants from the cultivation process and operations through to fuel products is required to shape the water considerations in the full supply chain.

The differences in water use can vary significantly starting with the cultivation operations where open pond operations are site-specific and specific to environmental conditions and may or may not include pond water recycle, and may harvest at different concentrations where water is lost with each harvest and may or may not be recycled back into the system. Beyond the harvesting, different technological approaches to dewatering are further dependent on the downstream processing and product end point.

The use of algae for fuels can involve a lipid-only extraction. Alternatively, a whole algal biomass slurry can be processed into a bio-oil intermediate through hydrothermal liquefaction (HTL), after which oil is upgraded and fractionated into a variety of end fuel products. The HTL process can recycle most of the water that is put into the system. The fuel upgrading, whether by an HTL bio-oil or algae lipid, requires water for the refining process (i.e., cooling water, boiler, steam, backwash/rinse) (Wu et al. 2009; Luo et al. 2010). Design considerations as to whether lipid extraction or HTL processing are occurring on-site or are transferred elsewhere can have an impact of total site water use and feasibility for water recycle.

The use of PBRs will potentially minimize consumptive water use due to evaporative loss, but spray cooling is sometimes used to control the temperature of the media (NRC 2012). Algae-based ethanol secretion methods will also have a different consumptive water use where water losses are minimized in these closed systems. Direct ethanol secretion avoids a dewatering step and continually maintains a live algae culture.

12.4.3 Water Quality

Water quality of effluents from algal-biofuel facilities and receiving waters is influenced by the source of the water, nutrients and other amendments, and by the efficiency of nutrient use. Water quality of natural groundwater or surface water outside of the cultivation system is important for regional environmental objectives.

Nutrient-related indicators are important metrics of water quality for all biomass production, including algae (table 12.1). The net effect of algae cultivation systems on water quality depends on the baseline land management system, the quality of water released to natural environments, and the use of algae to modify water quality of associated systems, such as wastewater treatment. Eutrophication is caused or exacerbated by runoff from traditional agricultural systems; algae cultivation systems that produce a protein coproduct could replace other agricultural systems.[7] Overtopping or slow leakage of cultivation media from ponds to groundwater or surface water may occur in many ecosystems. (The risk would be lower for PBRs.)

If treated wastewater from wastewater treatment or concentrated animal feeding operations is used as a co-located nutrient source (not considered in the *BT16* volume 1 scenarios), downstream concentrations of nutrients in streams, as well as freshwater needs, should be positively affected by algae cultivation. As Chiu and Wu (2013) note, between 3% and 91% of water at the state level could be displaced by municipal wastewater. However, the risks to productivity from variable water chemistry and added

[7] In the scenarios discussed, however, croplands are not used for algae production.

microbes from some wastewater have yet to be overcome at a large scale (Shurin et al. 2013).

Downstream process elements could have a positive effect on water quality. Recycling of nutrients and algae would reduce nutrient loadings to streams (Murphy and Allen 2011). On-site water treatment would also reduce nutrient concentrations in effluents.

The use of impure gases from co-located power plants (coal-fired power plants and natural gas plants in volume 1 scenarios) could increase metal-contaminant loads in cultures and, ultimately, in natural waters. The potential accumulation of flue-gas-related contaminants in cultivation systems is not well understood but is beginning to be investigated. Examples include the incorporation of metals from coal-based flue gas in *Scenedesmus obliquus* (Napan et al. 2015) and *Desmodesmus communis* (Palanisami et al. 2015). Metal concentrations were not recommended as a generic indicator of water quality for algae biofuel systems in Efroymson and Dale (2015), but if algae affect levels of metals in surface or groundwater, then metal concentrations could be employed as an environmental indicator.

Because algae may be grown in coastal waters or saline or brackish groundwater, salinity of groundwater or surface water will sometimes be an important environmental indicator (table 12.1), as recommended by the NRC (2012). Unintentional leakage from open ponds, withdrawal effects, or injection of saline waste into the ground could lead to the possible salinization of groundwater or surface water in some environments. However, such salinization is hypothetical and has not been demonstrated. Water-quality effects could result from the construction and operation of pipelines to transport coastal waters to inland cultivation systems.

Water quality effects of ponds with plastic liners versus minimal or no plastic liners are described in the previous section on water quantity. Essentially, if permeability is very low (conductivity $<10^{-7}$ cm/s) due to physical and microbial clogging, then unlined ponds should not leach water or nutrients into underlying soils. If hydraulic conductivity is higher, adverse effects on groundwater quality could occur.

12.4.4 Other Environmental Indicators

12.4.4.1 Soil Quality

Soil quality is an important aspect of environmental effects of terrestrial biomass crops, which draw nutrients from the soil. Unlike vascular plants, algae do not extract nutrients or water from local soil, so soil nitrogen and phosphorus have not been proposed as environmental indicators for algae (table 12.1), as they are for terrestrial crops. Soil quality affects productivity of vascular bioenergy crops and ecosystems but not algae used for biofuels.

The main linkages of algal biomass production to soil quality are via excavation for construction and ultimate decommissioning. Therefore, bulk density has been proposed as an environmental indicator for algal biomass (table 12.1). If construction of ponds is performed at commercial scale, the top layers of soil may be compacted or removed (Davis et al. 2016), affecting soil density, potentially affecting soil carbon, and potentially creating a barrier between the surface soil and subsoil.

If unlined or partially lined ponds are used, soil nutrients are more likely to be affected by algae biomass production. With respect to the full supply chain, if residual algal biomass is used as fertilizer or a soil amendment, it has the potential to provide benefits to soil quality, particularly carbon and nitrogen.

12.4.4.2 Biodiversity

Algal biofuel production could affect aquatic or terrestrial biodiversity, but little research exists to support hypotheses related to algal biomass and biodiversity outside of the cultivation system. It is reasonable to assume that extensive freshwater or saltwater ponds across the landscape could affect populations of vertebrates. If wildlife were to drink from algal biomass ponds, potential toxic exposures to individuals could come from metals accumulated from flue gas, salinity, or toxins from opportunistic cyanobacteria (Kotut et al. 2010). Moreover, the high productivity of algae per acre, combined with the potentially large yields of protein coproducts, could result in decreased pressure for deforestation (and decreased pressure on forest biodiversity), compared that which could be associated with terrestrial crops.

Breaches or overtopping events could lead to large quantities of algae and nutrients released to aquatic ecosystems, causing some algal taxa to bloom, and potentially causing changes in the native community. However, it is reasonable to assume that existing or future best management practices would prevent or lower the risk of these events. Algae biomass production in marine waters is not considered in this study, and potential implications of production in coastal ecosystems have not been studied.

The selection and interpretation of biodiversity indicators should be specific to the region where they are applied. Therefore, the indicators' "presence of taxa of special concern" and "habitat area of taxa of special concern" (table 12.1) would be regional.

None of the variables from the scenarios in volume 1 are directly related to biodiversity effects, though releases of different strains and releases of saline versus freshwater would have different effects in different ecosystems.

12.4.4.3 Air Quality

Air quality indicators relate to regional human health, occupational health, or ecosystems. Air emissions can occur during feedstock production, but also during processes such as drying and extraction, refining, and transportation and use. A suite of four indicators has been proposed to measure air quality related to algal biomass production, namely tropospheric ozone, carbon monoxide, total particulate matter (PM) less than 2.5 micrometers (µm) in diameter (PM2.5) and total particulate matter less than 10 µm (PM10) (table 12.1 and Efroymson and Dale 2015). These are the same as the air-quality indicators recommended for terrestrial biomass by McBride et al. (2012). However, even less information is available on these indicators in the context of algal biomass production. The NRC Committee on Sustainable Development of Algal Biofuels (NRC 2012) suggested that air quality indicators may include concentrations of volatile organic compounds (VOCs) and odorous secondary metabolites for open-pond systems. For later steps in the supply chain, the NRC (2012) suggested particulates for active drying processes, air concentrations of solvent used for extraction processes; and particulates, hydrocarbons, and acid gases for pyrolysis, if used, as air-quality-related metrics. The GREET model estimates emissions of six EPA criteria pollutants: CO, VOCs, nitrogen oxides, sulfur oxides, PM10 and PM2.5 (Frank et al. 2011), without a judgment about their relative importance compared to other measures.

However, little evidence exists of emissions of these chemicals and materials from the cultivation process. VOCs have been detected as emissions from open ponds (personal communication from Paul Zimba in NRC 2012).

12.4.4.4 Primary Productivity

Productivity is a measure of the efficiency of biomass or biofuel production, and it may also be an economic or environmental measure (Efroymson and Dale 2015). For photosynthetic organisms, yield of biomass (and ultimately, fuel) is related to primary productivity, i.e., net flux of carbon from the atmosphere to the organisms per unit time. Whether productivity of algae represents an environmental indicator of bioenergy sustainability relates to the extent to which algal biomass cultures are connected to the ecosystem. The ecosystem context of the unit farms in volume 1 of *BT16* is important but outside of the scope of this chapter.

In *BT16* volume 1, the productivities of algae were modeled using the Pacific Northwest National Laboratory's Biomass Assessment Tool, with current productivity and future high-productivity scenarios. Productivity of algae is influenced by abiotic environmental conditions, including temperature and light; biotic conditions such as algae strains; microbial community structure; and the abundance of predators, pathogens, and self-shading by other algae (Kazamia et al. 2012; Shurin et al. 2013). In *BT16* volume 1, higher productivities were observed in warmer, sunnier regions of the United States, and seasonal cycles were projected. Higher productivities are related to higher profitability, and as described above, higher EROI, lower GHG emissions per biomass, and lower water consumption per biomass.

Primary productivity from vegetation on land that was removed to transition to algae production is an important consideration, not only for estimating changes in greenhouse gas emissions (Arita et al. 2016), but also for ecosystem functions. Using lands with high primary productivity could affect higher trophic level animals. Algae cultivation can use land that is marginally productive, reserving highly productive or biodiverse lands for other uses.

12.5 Summary and Future Research

Little information is available to support a quantitative analysis of the environmental effects of algae cultivation. Few examples of commercial algae production exist, and few environmental indicators have been measured for those systems. More specifically, environmental effects of the scenarios from *BT16* volume 1—namely those that involve one of three CO_2 co-location sources (coal-fired power plants, natural gas plants, and ethanol plants), freshwater or saltwater strains (*Chlorella sorokiniana* and *Nannochloropsis salina*), full plastic pond liners or minimal liners, and current or future productivities—have not been measured. GHG emissions from scenarios similar to the *BT16* scenarios have been estimated by Davis et al. (2016), but the full supply chain and appropriate baselines would have to be selected for net emissions to be estimated. Water consumption has been estimated and described for the scenarios in *BT16* volume 1, but the context of those estimates with respect to competitive use has not been determined.

Some conclusions about the scenarios in *BT16* volume 1 are clear. Increasing productivity has benefits for water consumption on a per-unit-biomass basis. Information is available to allow analysts to quantify the difference in GHG emissions between co-location scenarios and carbon-capture scenarios and between supplying pure, captured CO_2 to algae facilities in tanks and piping dilute, impure gas a short distance to algae-cultivation systems. Similarly, information is available to allow the quantification of carbon emissions from plastic liner production. However, quantitative estimates of the GHG emissions of biomass alone are not possible for an algal biomass system that is highly integrated, so a life-cycle analysis would need to evaluate the whole supply chain for CO_2 co-location scenarios. Cultivation systems that use saline or brackish water media have the potential to consume less water than freshwater systems. However, realizing this water consumption benefit would be dependent on the method used to handle blowdown. If evaporation ponds are used, saline water systems will have a relatively high consumptive use.

Some of the indicators require more regional context than others. Water flows are regional and season-specific. Measuring water consumption does not depend on regional variables, but interpreting the environmental significance is a regional exercise. Biodiversity indicators must be selected with the region in mind because particular species and habitats are valued in specific regions. However, GHG emissions have more global significance than regional significance, and understanding the regional context is not important for modeling or measuring emissions from facilities. Similarly, the temporal context of water withdrawals and water consumption is important within a given region. Understanding the relationship between regional soil biogeochemistry and the probability of sealing or leakage of unlined ponds is an important research need tied to water quantity, water quality, GHG emissions and profitability.

As with any environmental assessment, it is important to define a baseline or reference scenario. A business-as-usual scenario was not evaluated in *BT16* volume 1. Therefore, the only qualitative comparisons that can be made here are between the scenarios that were evaluated. The three co-location scenarios in volume 1 do not represent all of the potential algae production co-location strategies. Additional sources of CO_2 may be available (e.g., ammonia plants and cement plants), which would allow more biomass to be generated from potential GHG emissions, and co-location with wastewater is an opportunity to improve water quality of natural waters.

While tradeoffs were not specifically evaluated for the scenarios, it is clear that the cultivation of freshwater algae species uses much more water than the cultivation of saline water species. However, to maintain salinity targets, freshwater may be needed to dilute saline water, or salt may need to be added to brackish water. Much more energy may be spent pumping saline water overland from the sea or from deep saline aquifers to maintain salinity targets than the energy needed to withdraw and transport fresh surface water.

Of course, the exclusion of PBRs has a large effect on the potential national biomass, and enclosed systems have very different environmental advantages and disadvantages than those described here. For example, PBRs often use less water per unit of algal biomass produced, given that they are not subject to appreciable evaporation. However, spray cooling can increase that water use.

The broad spectrum of energy and food production and its intrinsic tie to water and energy use leads to an increasing need to evaluate aspects of sustainability and implement planning strategies. Research needs include quantifying uncertainty in surface and groundwater sources; evaluating, with a spatial and temporal emphasis, the available supplies and non-stationarity climate and extreme events that impact those supplies; identifying existing and proposed food and energy uses; and identifying interactions between uses and sources of water (GAO 2012; Bauer et al. 2014).

In addition to further research on GHG emissions and water quantity (both consumption and natural stream flows), research, including field studies and modeling, is needed to evaluate potential aquatic and terrestrial biodiversity, air quality, water quality, and primary productivity effects of growing diverse species of algae at the commercial scale. A better understanding of environmental effects will allow future resource analyses to quantify the potential availability of more environmentally sustainable biomass, rather than all potential biomass. Such an understanding will help industry place facilities in the best locations and continue to develop good management practices.

Research needs for algae production include quantifying the environmental effects that are only described in qualitative terms in this report and estimating environmental effects in additional contexts to those in the scenarios. As algae-produced food (protein) and feed becomes commercially viable, understanding the interactions between the profitability, food security, energy security, and water quantity will become paramount, just as current research is investigating the water-energy-food nexus.

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Appendix 12-A: Water Resource Indices

Water Resources Vulnerability Index

The Water Resources Vulnerability Index, often referred to as the "withdrawal to availability (WTA) ratio," is probably the simplest and most widely used of the water resources indices. All other indices described here are variants of the basic ratio of water supply to demand (Rijsberman 2006). The Water Resources Vulnerability Index is simply defined as a ratio (equation 12A.1).

Equation 12A.1:

W is total annual water withdrawals, and Q is the sum of available water. In general, a ratio >0.4 indicates water

$$\frac{W}{Q}$$

stress (Raskin et al. 1997; Alcamo et al. 2000). This simple equation can be expanded in a number of ways to include sector-specific water demand (including environmental flows) represented as withdrawals, with a weighting term to indicate estimated fraction of consumptive use and, thus, allowing for a water-reuse term. In addition, the index could be applied to a monthly time scale and at a user-defined spatial scale.

Water Supply Stress Index (WaSSI) and Water Supply Stress Index Ratio (WaSSIR)

The Water Supply Stress Index (WaSSI), originally proposed by Sun et al. (2008a, 2008b) provides a measure of the relative supply and demand of water at a monthly time step for eight-digit Hydrologic Unit Code (HUC) watersheds. The WaSSI is defined as equation 12A.2.

Equation 12A.2:

$$WaSSI_x = \frac{WD_x}{WS_x}$$

WD is water demand, *WS* is water supply, and *x* represents any number of different simulations that might impact water availability. In the original context of the equation, *x* identified baseline conditions and simulations around future changes to climate, land use, population, and various combinations of these changes. Traditionally, the use of this index is hinged on the use of the sector-specific U.S. Geological Survey (USGS) 5-year water-use data (e.g., Maupin et al. 2014), though other water-demand data could be used if they are available. *WD* then is defined by equation 12A.3.

Equation 12A.3:

$$WD = \left(\sum_{i=1}^{n} WU_x\right) + \sum PB_x$$

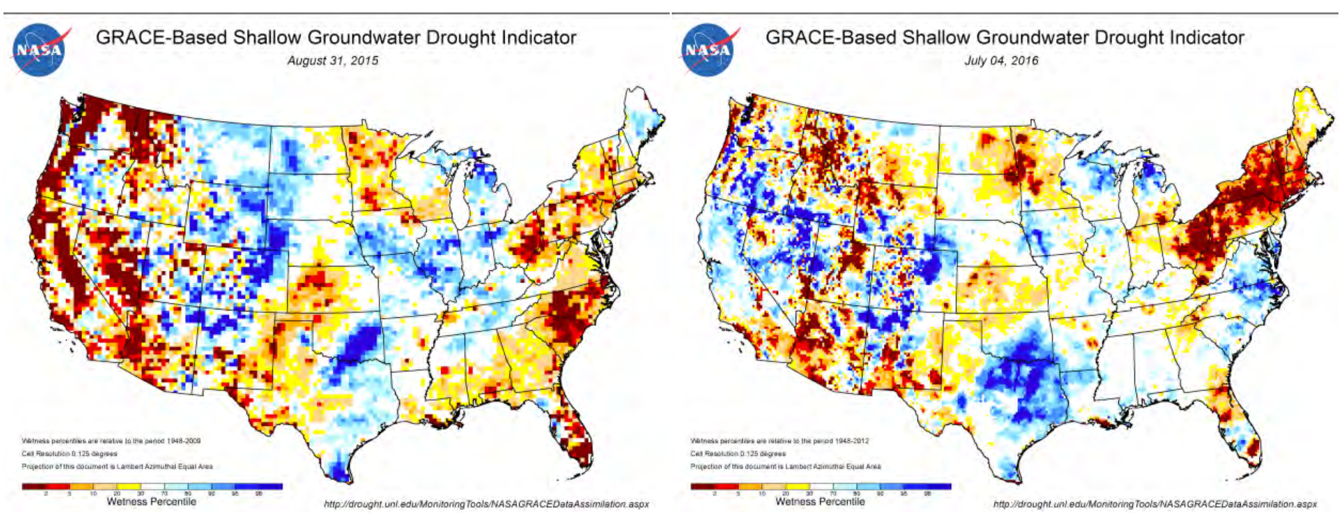
WU is the summation of water use by USGS-defined water-use sector n for each simulation, *x*, and *PB* is public use that also considers inter-basin water transfers by evaluating the difference between public-use withdrawals and public water supply. Water supply (*WS*) is defined by equation 12A.4.

Equation 12A.4:

$$WS = SS + GS + \sum RF_x$$

For a given watershed boundary, *SS* is the surface-water supply volume (e.g., eight-digit HUC), *GS* is the groundwater supply, and *RF* is the return flow volume for each sector *x*. The surface-water supply volume is a measure of total precipitation in the basin less the evapotranspiration lost out of the basin. Evapotranspiration can be calculated in one of a number of ways: Penman-Monteith, mass/energy balance models, models using observed satellite data (Moderate Resolution Imaging Spectroradiometer [MODIS] – MOD16 Global Evapotranspiration; Surface Energy Balance Algorithm for Land [SEBAL]; Mapping EvapoTranspiration at high Resolution with Internalized Calibration [METRIC]). The groundwater supply can be estimated in many ways (i.e., models, direct and remote-sensed observations [fig. 12A.1]); however, for sustainable use, Alley, Reilly, and Franke (1999) define this as “development and use of groundwater in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic, or social consequences,” requiring more local-to-regional definition. Sun et al. (2008a) and McNulty et al. (2010) suggest use of withdrawal records from the USGS 5-year water-use data (Solley et al. 1998; Maupin et al. 2014).

Figure 12A.1. | Regular monitoring of shallow groundwater supply using a data assimilation of the National Aeronautics and Space Administration’s GRACE (Gravity Recovery and Climate Experiment) satellite data, direct observations, and mass/energy numerical models (National Drought Mitigation Center, University of Nebraska – Lincoln; Houborg et al. 2012)



Finally, the return flows can be estimated as a fraction of the sector-specific water use, and it is suggested that this fraction be established regionally per sector.

To complement the WaSSI, the Water Supply Stress Index Ratio (WaSSIR) provides a ratio between a baseline and the various simulations of the WaSSI. The WaSSIR is simply defined as equation 12A.5.

Equation 12A.5:

$$WaSSIR_x = \frac{WaSSI_x - WaSSI_b}{WaSSI_b}$$

Where $WaSSI_x$ is a given WaSSI scenario and $WaSSI_b$ is the baseline, negative ratio values indicate reduced water stress, and positive values indicate increased water stress as compared to baseline. The further positive deviation from 0 there is, the greater the stress.

Potential simulations using the WaSSI can not only incorporate future projected climate, population, and land-use chance (LUC), but can also specifically address and evaluate the potential water sustainability of potential microalgae production from scenarios in the *2016 Billion-Ton Report* (*BT16*) volume 1. The site-specific, modeled, consumptive water-use requirements (Wigmosta et al. 2011) for each carbon dioxide (CO_2) co-located site can be aggregated from the site scale to the watershed scale, and the WaSSIR can be assessed to determine the potential water stress.

Water Scarcity Index (Wsci)

Asheesh (2007) established the Water Scarcity Index (Wsci) as a method to identify gaps that would prevent a balance in water supply and water demand amongst a complex relationship of variables, referred to as the Water Equality Accounting System. The Wsci is defined by equation 12A.6.

Equation 12A.6:

$$Wsci = \left(\frac{\alpha}{\left(\left[\frac{100}{100 - p} \right] \beta e^{\lambda \Delta t} (\varepsilon + \gamma + \delta) \left(\frac{100}{100 - \kappa} \right) + h + b \right)} \right) - 1$$

Where the following are true:

α is the annual total freshwater availability

p is the industrial water demand (%)

β is the current population

λ is the population growth rate

Δt is the length of time considered (years)

k is the estimated annual water system losses

ε is the annual domestic per capita demand

γ is the annual per capita demand for green areas (dependent on population growth)

δ is the annual irrigation demand

h is the annual evapotranspiration

b is the annual environmental water requirement.

While there is no specific spatial scale identified in the Wsci equation, this could be evaluated on a watershed-to-regional scale where information is available to support the terms. In addition, the equation was intended to be run at an annual scale with a future projection of population, and it would be possible to evaluate the use of the Wsci at the monthly time-step to better reflect water opportunities for microalgae production. Modeled, net consumptive water use of open pond or photobioreactor (PBR) systems can be incorporated into a new, independent term for the denominator of this equation.

Water Stress Indicator

Smakhtin et al. (2005) provide a simple environmental water-scarcity method forward that considers the relationship of water withdrawals to the environmental water requirement: the Water Stress Indicator (WSI). The WSI is represented by equation 12A.7.

Equation 12A.7:

$$WSI = \frac{W}{MAR - EWR}$$

Where *W* is the total water withdrawal in a basin, *MAR* is the naturalized, long-term mean annual runoff volume that represents the total water supply, and *EWR* is the annual environmental water requirement. The WSI method is intended for a global-scale analysis, but as with other indices included in this chapter, it can be modified for use at finer temporal or spatial scales to help understand the water-use impacts that microalgae production might have in the spatiotemporal context. The environmental water requirements can be defined in a number of ways, as described below (environmental flow requirements). The classification of the WSI is described in table 12A.1.

Table 12A.1. | Classification of the Water Stress Indicator (WSI) as Defined by Smakhtin et al. (2005)

WSI	State of Basin Environmental Water Scarcity
WSI > 1	Overexploited (current water use is tapping into EWR)— environmentally water-scarce basins
0.6 ≤ WSI < 1	Heavily exploited (0%–40% of the utilizable water is still available in a basin before EWRs are in conflict with other uses)—environmentally water-stressed basins
0.3 ≤ WSI < 0.6	Moderately exploited (40%–70% of the utilizable water is still available in a basin before EWR are in conflict with other uses)
WSI < 0.3	Slightly exploited

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Appendix 12-B: Environmental Flow Requirements

As described in chapter 7, environmental flow requirements are a key target for environmental indicators. The International Union for Conservation of Nature and Natural Resources defines environmental flows as "…the water regime provided within a river, wetland, or coastal zone to maintain ecosystems and their benefits where there are competing water uses and where flows are regulated. Environmental flows provide critical contributions to river health, economic development, and poverty alleviation. They ensure the continued availability of the many benefits that healthy river and groundwater systems bring to society" (Dyson et al. 2003). Thus, indicators of environmental flow may be considered indicators of biodiversity, as well as indicators of water quantity, and need to be considered in sustainable use of water resources. In addition, environmental water requirements identified in several of the water-resource indices previously described can be assessed using a variety of methods noted below.

A fundamental tenet in developing environmental flow requirements is the understanding that hydrologic conditions have intrinsic variability over a range of time scales, primarily as a function of meteorology and longer-term climate. The management of environmental flows involves understanding the components of flow and their relationship to the specific ecosystem need—namely magnitude, frequency, duration, variability, timing, and rate of change—which can impact the overall structure and function within an ecosystem (Poff and Ward 1989; Richter et al. 1996). More specifically, fluxes of nutrient and prey availability; habitat development and maintenance; life-history flow requirements that support migration freshets, spawning, and nursery environments; flushing flows for sediment cleaning, transport, and redistribution; hydrologically connected/fragmented habitats; water quality; quality of aquatic habitats; riparian and wetland function; and more can contribute to the environmental flow requirements (O'Keeffe and Quesne 2009).

Peak flow and minimum base flow are described as basic environmental indicators for water quantity in table 12.1, but more complex measures may be needed to incorporate the regional context. Tharme (2003) identified more than 200 methods available to assess environmental flows, and generally, they can be classified as hydrological, hydraulic rating, habitat simulation, and holistic methodologies. The methods can be distinguished as follows:

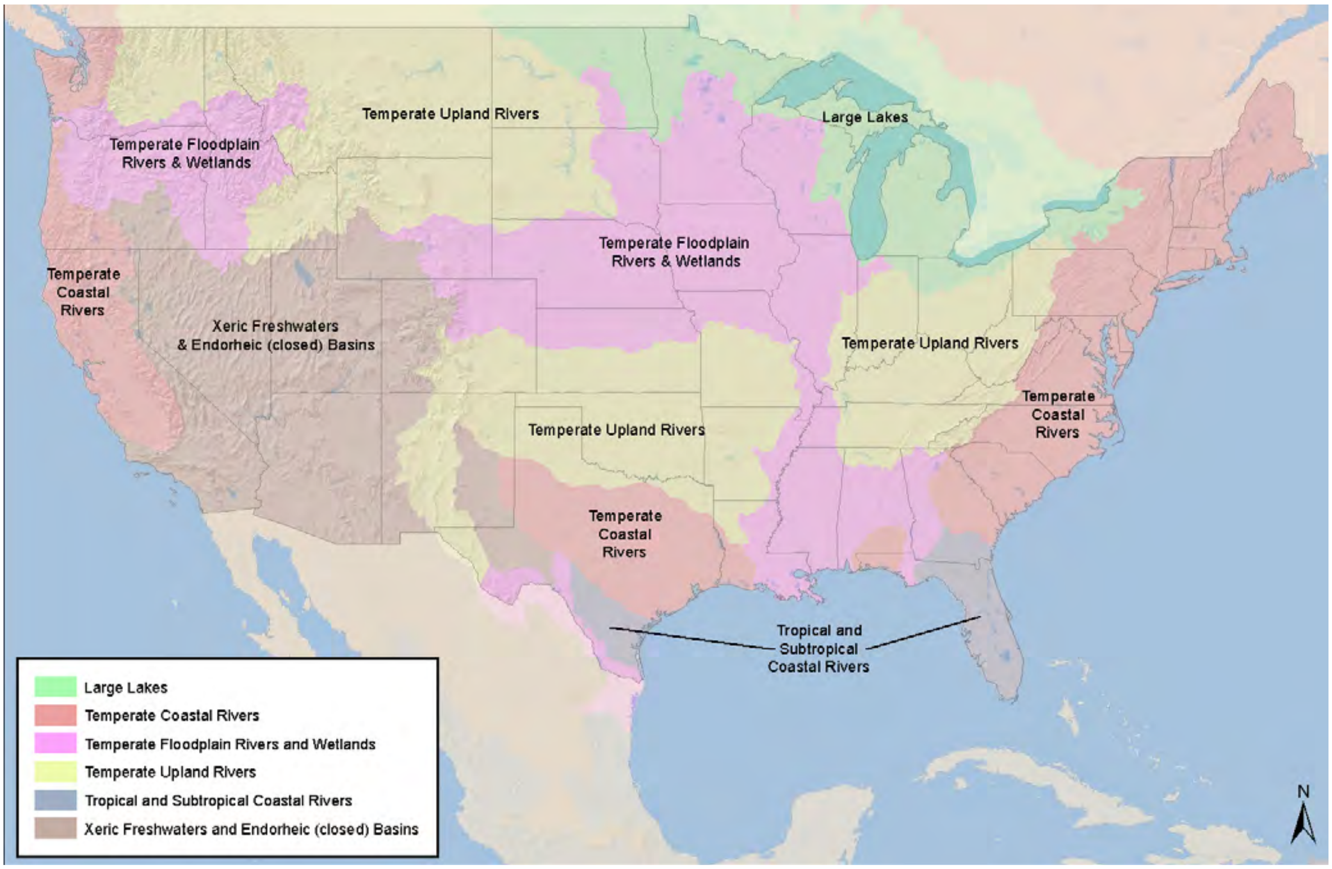
- **Hydrological:** Typically, the hydrological methods are the simplest within the four groups of environmental flow–assessment methodologies. These methods primarily use observed or modeled flow data as in the form of naturalized (unimpaired) flow, historical daily or monthly flow records. Some methods also incorporate a combination of catchment-scale variables and geomorphic and biological indices. In general, these methods are appropriate for planning-level applications and can use readily available data. A few examples of these models follow: (1) The Q_{95} Index environmental flow is defined as the flow that exists ≥95% of the time during the period of record. (2) The Tennant Method (Tennant 1976) assesses environmental flow as a percentage of mean annual flow that relates to the desired habitat quality, where 10% is poor quality, 30% is moderate, and >60% is excellent; however, these values were established statistically for rivers and streams in the midwestern United States and would need to be adjusted for other regions. (3) The Indicators of Hydrologic Alteration (Richter et al. 1996) use numerous input parameters and more than 60 resulting indicators to provide a statistical characterization of the system of interest.

- **Hydraulic Rating:** This class of methods relies on more-detailed and site-specific information, requiring channel cross-sections or three-dimensional bathymetry to assess and relate changes in simple hydraulic variables (e.g., wetted perimeter) to define habitat or aquatic-system requirements and objectives. Environmental flows then are established by defining habitat quality thresholds by relating discharge to habitat indices of concern. The flows generally follow a response curve, or flows are set according to a fixed percentage to reflect an acceptable loss of habitat and/or habitat function. Much of the work done under the hydraulic rating methods is preparatory for more complex method groups of habitat simulation and holistic approaches. The R2Cross method is an example of a hydraulic-rating method (Espegren 1996; Armstrong, Todd, and Parker 2001).
- **Habitat Simulation:** Detailed, local-scale analyses that incorporate hydrological data (i.e., flow magnitude, frequency, duration, variability, timing, and rate of change), hydraulic data (i.e., depth, velocity, shear stress, etc.), and biological data (i.e., habitat-suitability index for specific species, assemblage of species, life stages) are used together to produce habitat-discharge curves and habitat-exceedance probability curves. Well-known simulation models here include the Instream Flow Incremental Methodology (IFIM) model/Physical Habitat Simulation (PHABSIM) model (Bovee et al. 1998; Milhous and Waddle 2012) and variants, including the River Hydraulics and Habitat Simulation (RHYHABSIM) model (Jowett 1989), the Riverine Habitat Simulation (RHABSIM) model (Payne 1994), and the Mesohabitat simulation (MesoHABSIM) model (Parasiewicz 2001, 2007). This approach brings together a more detailed emphasis on flow thresholds that best support system biodiversity. Generally, because of the data and effort required, an approach such as this is conducted at a more local scale.
- **Holistic:** The goal of holistic methodologies, such as the building block method (King, Tharme, and de Villiers 2000, 2008), is to consider the various aspects—ecological, geomorphological, and social—of an entire riverine ecosystem in order to develop appropriate levels of environmental flow. These are often constructed across a range of possible conditions that require expert judgment from multiple subject-matter experts and may ultimately move toward an adaptive management plan. Two general approaches are taken: (1) top-down, where environmental flows across a range of different conditions are defined as an acceptable departure against naturalized or reference flows, as categorized by critical flow events from the perspective of multiple subject matter experts; and (2) bottom-up, where defined objectives of individual elements (ecological, geomorphic, socioeconomic, etc.) are built up and assessed at a finer temporal resolution (i.e., monthly or seasonally) in order to achieve a multi-objective flow regime. As with the habitat simulation approaches, the holistic methodologies are time- and resource-intensive and are not effectively used at a regional or national scale.

Because each aquatic system has specific requirements and objectives, deriving physical and ecological interrelationships requires expert opinion and/or field-collected data to fully parameterize more local-scale hydraulic and habitat models. It also requires appropriate regional- to national-scale analyses of environmental flow requirements, often using a "desktop method" (also known as "historic flow," "discharge," and "lookup table" methods) that only requires the use of consistent and readily available data, such as that found in the national sets of hydrologic data from the USGS's National Water Information System (http://waterdata.usgs.gov/nwis). It is possible, however, to incorporate an ecological component into some hydrologic-based environmental flow methods via weighting factors or percent water allocations. A biogeographical classification dataset such as the

Freshwater Ecoregions of the World (Abell et al. 2008) provides a regional perspective that incorporates aquatic biodiversity, endemism, and hydrologic fragmentation (see fig. 12B.1), thus allowing varying ecological functions and conditions to be evaluated separately.

Figure 12B.1. | A global freshwater ecoregion classification that incorporates aquatic biodiversity, endemism, and hydrologic fragmentation (Source Data: Abell et al. 2008)



Pastor et al. (2014) evaluated several hydrologically based environmental flow–assessment methods, including Smakhtin, Revenga, and Döll (2004), Tennant (1976), Hoekstra et al. (2012), Tessmann (1980), and their own variable monthly flow (VMF) and Q90_Q50 (flow that is present 90% of the time; flow that is present 50% of the time) methods. The Tessmann (1980) and VMF approaches are the only methods that consider monthly flow, low-flow, high-flow, and intermediate-flow indicators. These two methods are based on a temporal resolution that reflects inter-annual cycles; they represent a more detailed flow regime, fit the requirement of using readily available data for basin-, regional-, or national-scale analyses, and have been demonstrated with a high correlation to locally developed environmental flows amongst several characteristically varying basins. We discuss these two approaches below.

Variable Monthly Flow (VMF) Method

The VMF method is one suggested approach for estimating environmental flow. It uses the variability in monthly flow to quantify a reasonable level of ecological protection with the ability to support other water-use activities, such as agriculture, industry, and domestic use. The VMF provides temporal detail (monthly) that is appropriate for planning around the flows throughout the year and allocates required environmental flows as a percentage of the naturalized mean monthly flow. When combined with estimates of existing consumptive water use across all sectors, these estimates provide a high-level assessment of available freshwater resources that is appropriate for use at the regional and national scales for the purpose of resource use and planning; however, particular local conditions and policies will always need to be evaluated. To reflect the variability and ecological need in seasonal flows, in general, a smaller percentage of flows can be allocated for socioeconomic use during low-flow periods; during high-flow periods, a greater fraction of the water resource can be used. Allocation rules can be adjusted according to regional conditions and need across both environment and socioeconomics; general guidelines are provided in table 12B.1.

Table 12B.1. | General Flow Allocation Guidelines for the VMF Method Assessed Monthly (Pastor et al. 2014).

	% of Mean Annual Flow	% Water for Environmental Flow	% Water for Socioeconomic Use
Low-Flow	<40%	60%	40%
Intermediate-Flow	40%–80%	30%–60%	40%–70%
High-Flow	>80%	30%	70%

Tessmann Method

The Tessmann (1980) method to assess environmental flow is a modification of an earlier method established by Tennant (1976), which is based on the field assessment of nearly a dozen rivers in Montana, Nebraska, and Wyoming. A fraction of the mean annual flow is required, where, generally, 10% is the minimum flow and intended only for short-term use to sustain the aquatic environment, and ≥30% of the mean annual flow is what is required to sustain the ecological integrity of the aquatic ecosystem. Additional guidance was provided by Tennant (1976) for low-flow and high-flow seasons, fall and spring, respectively, to maintain the aquatic ecosystem (see table 12B.2).

Table 12B.2. | Tennant (1976) Recommendations for Environmental Flows Parsed by Low- and High-Flow Seasons

Aquatic Ecosystem Maintenance	% of Mean Annual Flow	
	Low-Flow (Oct.–Mar.)	High-Flow (Apr.–Sep.)
Flushing Flows, or Maximum	200%	200%
Optimum Range	60%–100%	60%–100%
Outstanding	40%	60%
Excellent	30%	50%
Good	20%	40%
Fair/Degrading	10%	30%
Poor/Minimum	10%	10%
Severe Degradation	<10%	<10%

Tessmann (1980) uses the same principles as Tennant (1976), but instead of using two flow regimes (low-flow, high-flow) and mean annual flow, Tessmann (1980) uses a ratio of mean monthly flows to mean annual flows and, accordingly, assigns flow rules to one of three categories (see ruleset in table 12B.3). With more temporal detail, Tessmann (1980) offers an approach that can be used in a variety of hydrologic systems throughout the world, though in general, the environmental flow guidelines are more conservative than other methods discussed here (i.e., it keeps more flow in the river).

Table 12B.3. | The Tessmann (1980) Rules for Environmental Flow Based on Naturalized Mean Monthly Flow (MMF) and Naturalized Mean Annual Flows (MAF)

Naturalized Flow Condition	Environmental Flow Requirement
if: MMF < 40% MAF	*then*: MMF
if: MMF > 40% MAF and 40% MMF < 40% MAF	*then:* 40% of MAF
if: 40% MMF > 40% MAF	*then:* 40% of MMF

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Appendix 12-C: Contributions of Sectors to Total Consumptive Water Use

Figure 12C.1 illustrates the percentage of each consumptive water-use category contributing to the total consumptive water use from 1985 to 2000. All years except 2000 contain consumptive use data; therefore, a trend for each category was determined as a ratio of consumptive water use to total water use and extrapolated to determine the sector-specific consumptive water use for 2000 (Moore et al. 2015). Comparing consumptive water use to water withdrawal provided in the U.S. Geological Survey (USGS) 2010 water-use report (Maupin et al. 2014), one can see the significant fractional difference in consumptive use vs. withdrawals, where, for example, thermoelectric represents 3.9% of all consumptive use; for withdrawals, this same sector accounts for ~45% of all withdrawals (i.e., a significant amount of water is pulled for thermoelectric use, but is returned to the system) (fig. 12C.2).

Figure 12C.1. | Percentage of average annual consumptive water use for each USGS-defined sector, 1985–2000

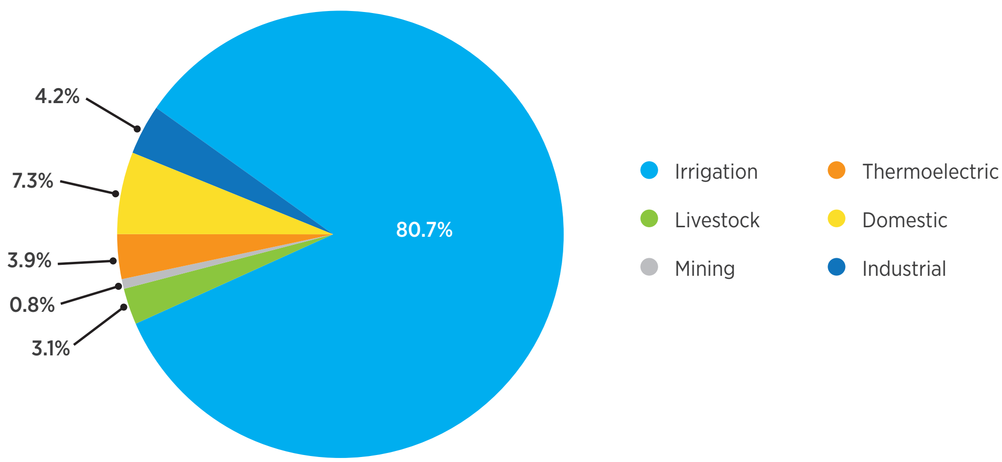
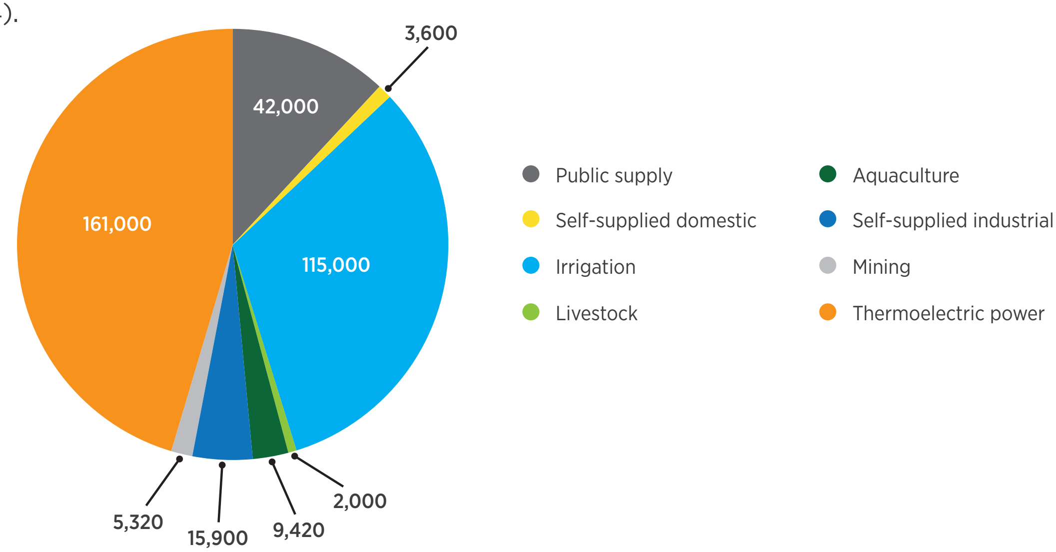
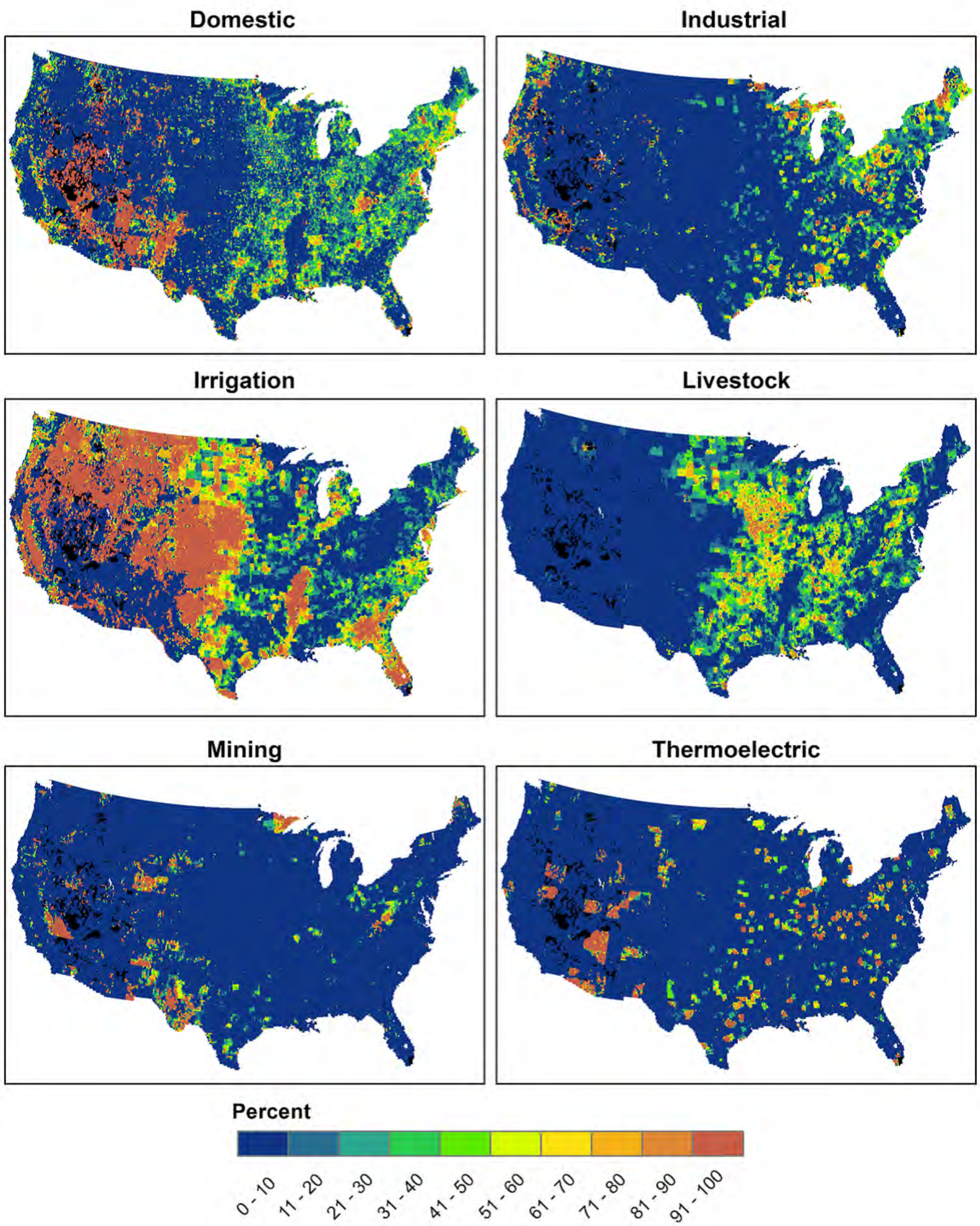


Figure 12C.2. | The pie graph represents the USGS 2010 estimates of water withdrawals per sector averages, 2005–2010. Note the additional sectors in the 2010 data as compared to the 1985–2000 data. (Image credit: Maupin et al. 2014).



To help understand the distribution of consumptive water use amongst sectors, Moore et al. (2015) developed a spatially explicit view of the percent annual consumptive use relative to the total as an average from 1981 to 2000 (fig. 12C.3).

Figure 12C.3. | A spatially explicit high-resolution (1/8°) sector view showing percentage of annual consumptive use relative to the total consumptive water use. Values represent an average from 1981–2000 and highlight primary consumptive water-use sectors within the United States. (Image credit: Moore et al. 2015).



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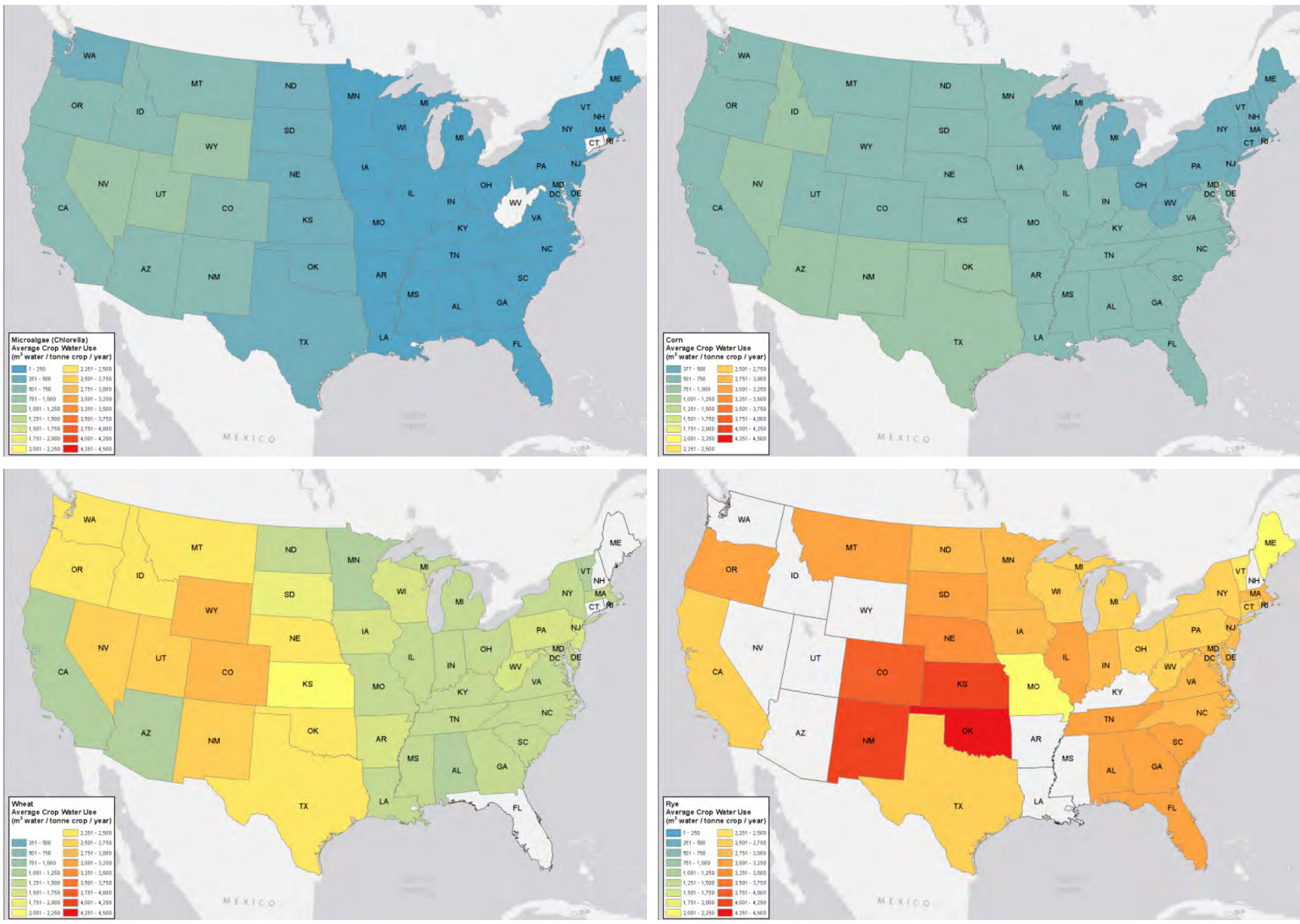
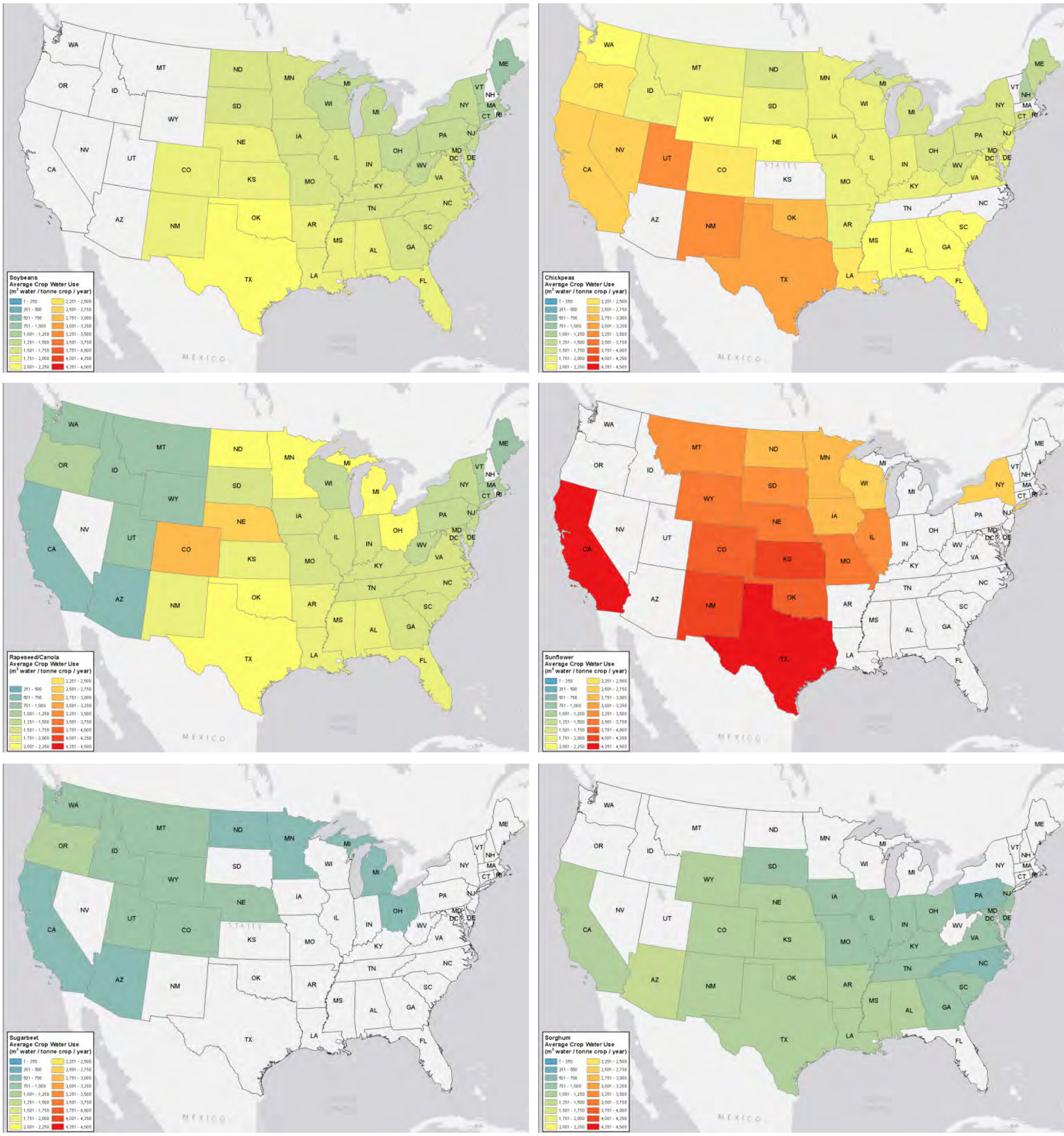
Appendix 12-D: Comparison of Water Use by Selected Terrestrial Crops and Microalgae Water Use—Geographic Analysis

Figure 12D.1. | Annual average green + blue crop water use by state. Terrestrial crop water use from Mekonnen and Hoekstra (2011); microalgae crop water use from Wigmosta et al. (2011).




Soybeans
Average Crop Water Use
(m³ water / tonne crop / year)
Chickpeas
Average Crop Water Use
(m³ water / tonne crop / year)
Rapeseed/Canola
Average Crop Water Use
(m³ water / tonne crop / year)
Sunflower
Average Crop Water Use
(m³ water / tonne crop / year)
Sugarbeet
Average Crop Water Use
(m³ water / tonne crop / year)
Sorghum
Average Crop Water Use
(m³ water / tonne crop / year)
1 - 250
251 - 500
501 - 750
751 - 1,000
1,001 - 1,250
1,251 - 1,500
1,501 - 1,750
1,751 - 2,000
2,001 - 2,250
2,251 - 2,500
2,501 - 2,750
2,751 - 3,000
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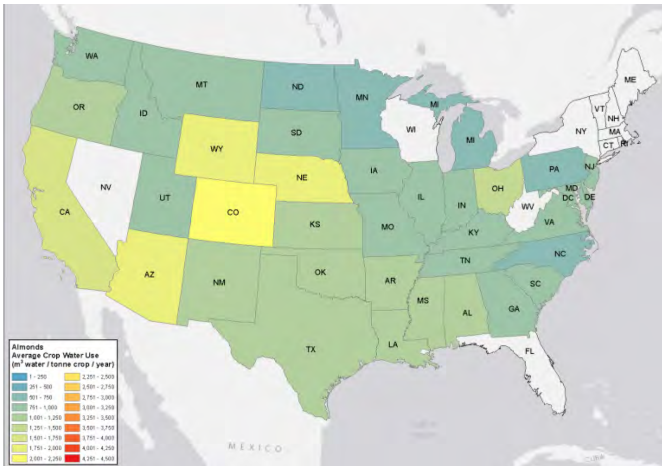
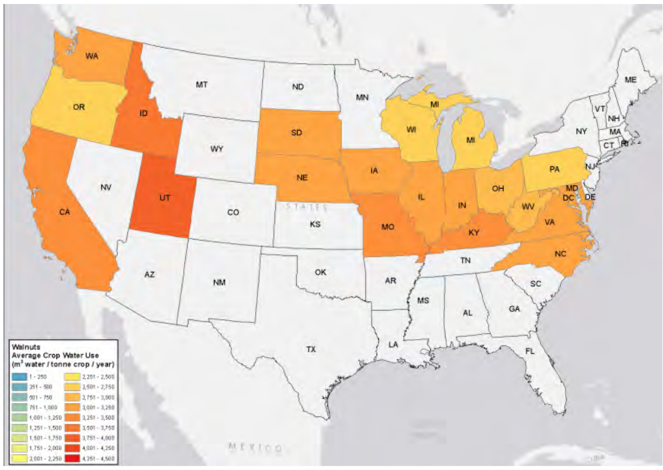
3,751 - 4,000
4,001 - 4,250
4,251 - 4,500
MEXICO





References

Mekonnen, M. M., and A. Y. Hoekstra. 2011. "The Green, Blue and Grey Water Footprint of Crops and Derived Crop Products." *Hydrology and Earth System Sciences* 15 (5): 577–1600. doi:10.5194/hess-15-1577-2011.

Wigmosta, M. S., A. M. Coleman, R. J. Skaggs, M. H. Huesemann, and L. J. Lane. 2011. "National Microalgae Biofuel Production Potential and Resource Demand." *Water Resources Research* 47 (3): W00H04. http://dx.doi.org/10.1029/2010WR009966.

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13 | Climate Sensitivity of Agricultural Energy Crop Productivity

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13.1 Introduction

Bioenergy, including biofuels and biopower, has received significant attention as a technology for increasing U.S. energy security and offsetting greenhouse gas (GHG) emissions from fossil energy (Schneider and McCarl 2003; Adler, Grosso, and Parton 2007; Campbell et al. 2008; Field, Campbell, and Lobell 2008). However, the potential effect of climate change on biomass production has received comparatively little attention (Jones and Dalton 2012; Wilbanks et al. 2012; Tuck et al. 2006; Schröter et al. 2005; Haberl et al. 2011; Poudel et al. 2011; de Lucena et al. 2009; Dominguez-Faus et al. 2013). For example, recent assessments of the implications of climate change for U.S. energy systems acknowledge the potential climate sensitivity of biomass (CCSP 2007; Wilbanks et al. 2012), but contain little discussion of the timing and magnitude of future climate effects on different biomass resources.

As with all agricultural and forestry production, biomass resources for bioenergy are highly exposed and sensitive to weather and climate (Wilbanks et al. 2012), and thus, they may be more vulnerable than other energy sources to climate change (Eaves and Eaves 2007). Given projections that some extreme weather events will increase in frequency, duration, and/or intensity (Ortman and Guarneri 2009), climate risk to biomass derived from agricultural and forest enterprises would also be expected to increase. Yet, future changes in climate could also create opportunities for enhanced yields of particular energy crops in areas that are not currently climatically suitable for production of those crops. Greater attention to the implications of climate change for the production of biomass resources is therefore warranted.

This chapter differs from other chapters in this report in that it evaluates the effects of climatic changes on potential future biomass production, rather than evaluating environmental effects of biomass production. Thus, it does not apply the production scenarios evaluated in the other chapters. The objective of this chapter is to assess the sensitivity of U.S. cellulosic biomass to climate change by presenting initial empirical estimates of the implications of alternative climate-change scenarios for a number of illustrative energy crops. In doing so, the chapter seeks to address the extent to which future changes in climate variables (e.g., temperature and precipitation) are projected to drive significant changes (positive or negative) in the yields of energy crops at the national, regional, or county level. In addition, this chapter addresses the implications of those changes for biomass production, as well as key knowledge gaps arising from this assessment and its methods, which could be addressed with future research. Because this chapter analyzes the climate sensitivity of biomass without consideration for changes in management practices, other changes in environmental conditions, or the economics of production, results should not be treated as future predictions. Rather, the biomass projections based on particular climate scenarios help in (1) identifying the areas where production of different energy crops is anticipated to benefit or to be harmed in response to climate change and (2) prioritizing future research needs.

13.2 Methods

13.2.1 Scope of Assessment

This assessment estimates the implications of climate change for the geographic distribution and yields of potential cellulosic energy crops. Yields were modeled at the county level for the continental United States for the current climate and in response to future climate conditions as simulated by multiple Earth system models (ESMs) and model configurations (i.e., different versions of a particular ESM). The modeling also incorporates four different scenarios of future GHG concentrations in the atmosphere to capture the uncertainty in global GHG emissions. The assessment includes seven energy crops:

1. Conservation Reserve Program (CRP) mix of grasses, forbes, and legumes[1]
2. Energy Cane
3. Miscanthus
4. Poplar
5. Sorghum
6. Switchgrass (lowland and upland)
7. Willow

Forest biomass is not included in this assessment.

Yields for these energy crops were estimated for two future time periods, 2050 and 2070; that is, the assessment looks further into the future than the other modeling conducted for *BT16*. Changes in climate over shorter time frames (e.g., 2030) may be difficult to distinguish from natural climate variability. Hence, results reflect yield changes that would be anticipated in response to changes in climate conditions for U.S. counties over the long term. This long-term temporal extent enables near-term developments in biomass production to be considered in the context of long-term uncertainty in future climate change. Results do not account for changes in the intensity, frequency, or duration of extreme weather events; indirect effects of climate change such as pests or disease; fertilization effects associated with higher atmospheric carbon dioxide (CO_2) concentrations; or changes in management practices or biotechnology. Therefore, the results reflect first-order estimates of biomass for the purpose of identifying energy crops that may be particularly vulnerable or resilient to changes in climate variables, as well as identifying possible range shifts. Also, because of inherent uncertainties in long-term economic trends and the dynamics of biomass and bioenergy markets, this assessment does not consider the economic drivers of energy crop production or interpret the results in the context of different price assumptions.

13.2.2 Description of Modeling Approach

Yields for the biomass crops were modeled using a two-stage process. First, relative yields of particular energy crops for current climate conditions were modeled using the PRISM (Parameter-elevation Relationships on Independent Slopes Model) Environmental Model (PRISM-EM) (Halbleib, Daly, and Hannaway 2012; DOE 2016). PRISM-EM is an empirical model for estimating production potential for selected energy crops under various water balance, temperature, and soil constraints based on extrapolation of field trial data. Relative yield represents the fraction of the theoretical maximum physiological yield that can be achieved for a particular energy crop in a location given environmental constraints. Relative yield values range from 0% (no production) to 100% (maximum production). Although not a direct measure of absolute yields (i.e., tons per acre) of energy crops, increases in relative yields are indicative of increases in absolute yields while decreases in relative yields are indicative of decreases in absolute yields. The two key inputs for PRISM-EM are climate conditions from the PRISM historical climate data set (Daly et al. 2008) and soil conditions from the Soil Survey Geographic database (USDA 2016). Because PRISM-EM is based on historical climate information, it does not currently model the effects of future changes in climate. To extrapolate the results from PRISM-EM into the future, the historical

[1] CRP land was not included in potential biomass production areas in the BT16 volume 1 but is a potential source of biomass described in section 13.3.2.1.

climate, soil, and relative yield information used by and generated from PRISM-EM was used to develop Bayesian statistical models for each of the aforementioned energy crops. The Bayesian models emulate PRISM-EM by using the quantitative relationships among temperature, rainfall, soil conditions, and energy crop yields to generate expected relative yields for particular crops and a given combination of environmental conditions.

Bayesian models were trained by using 30 years of PRISM-EM results aggregated to the county level in conjunction with annual average minimum temperature (T_{min}), annual average maximum temperature (T_{max}), and total annual precipitation for each year, as well as the soil conditions for each county. A comparison of county-level, aggregate-yield results from the Bayesian models indicated that they perform well in capturing the magnitude and spatial distribution of energy crop yields (see appendix A for validation and uncertainty metrics). Sensitivity analyses conducted on the Bayesian models indicated precipitation was the dominant variable influencing yield, followed by temperature. The one exception was energy cane, for which temperature (T_{min} and T_{max}) was more important. In most instances, modeled yield had a greater sensitivity to T_{min} than T_{max} (fig. 13.1). In these models, yield was rather insensitive to soil variables relative to climate variables (fig. 13.1).

In the second stage of the modeling, Bayesian models trained with PRISM-EM results were used to project relative yields of energy crops in response to alternative climate information and scenarios (see fig. 13.2), based on the assumption that the relationships

Figure 13.1 | Sensitivity of Bayesian yield models to input variables. Sensitivity was calculated as the variance reduction (expressed as a percentage) associated with each input variable (Marcot 2012). Higher variance reduction scores reflect greater sensitivity to specified input variables.

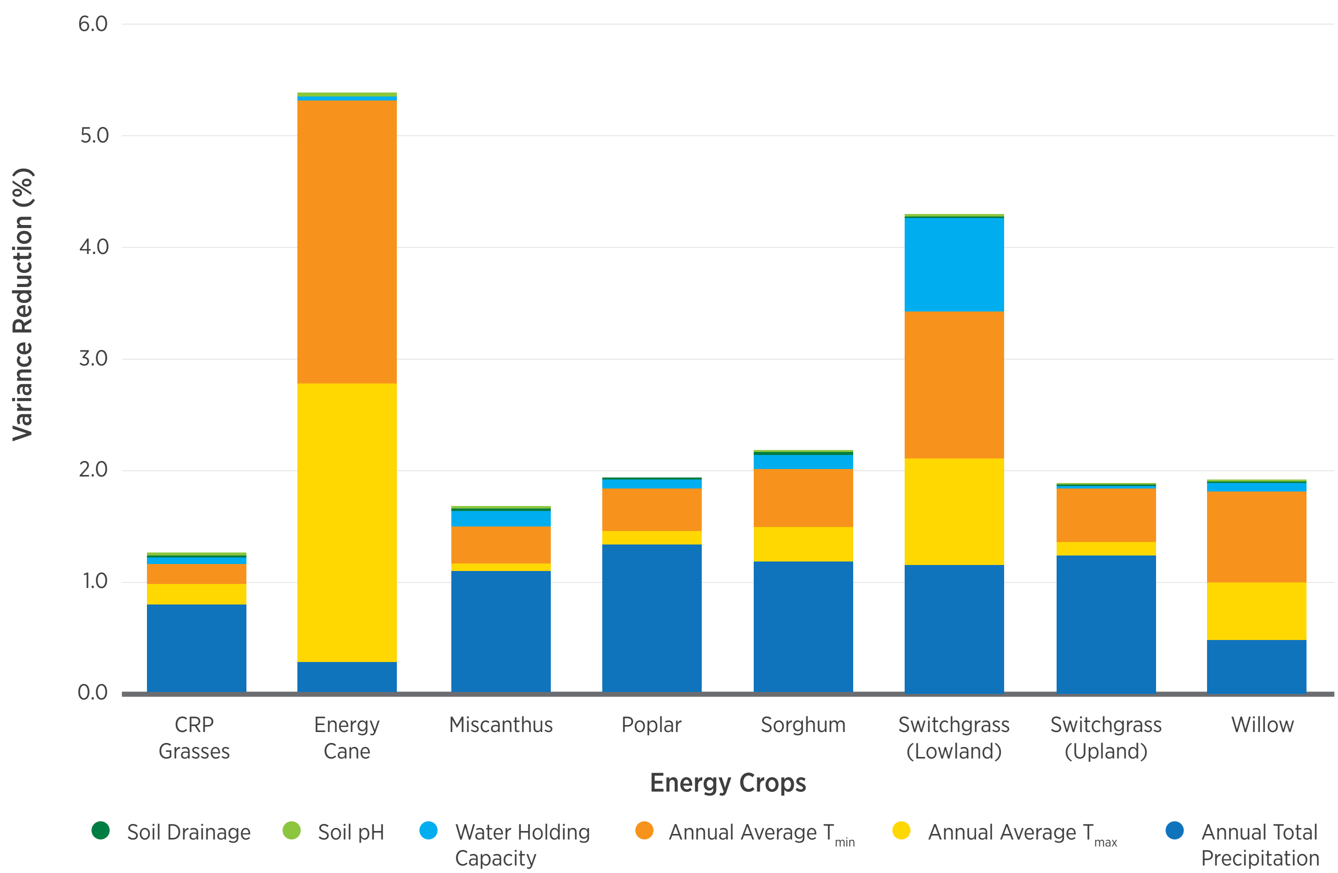
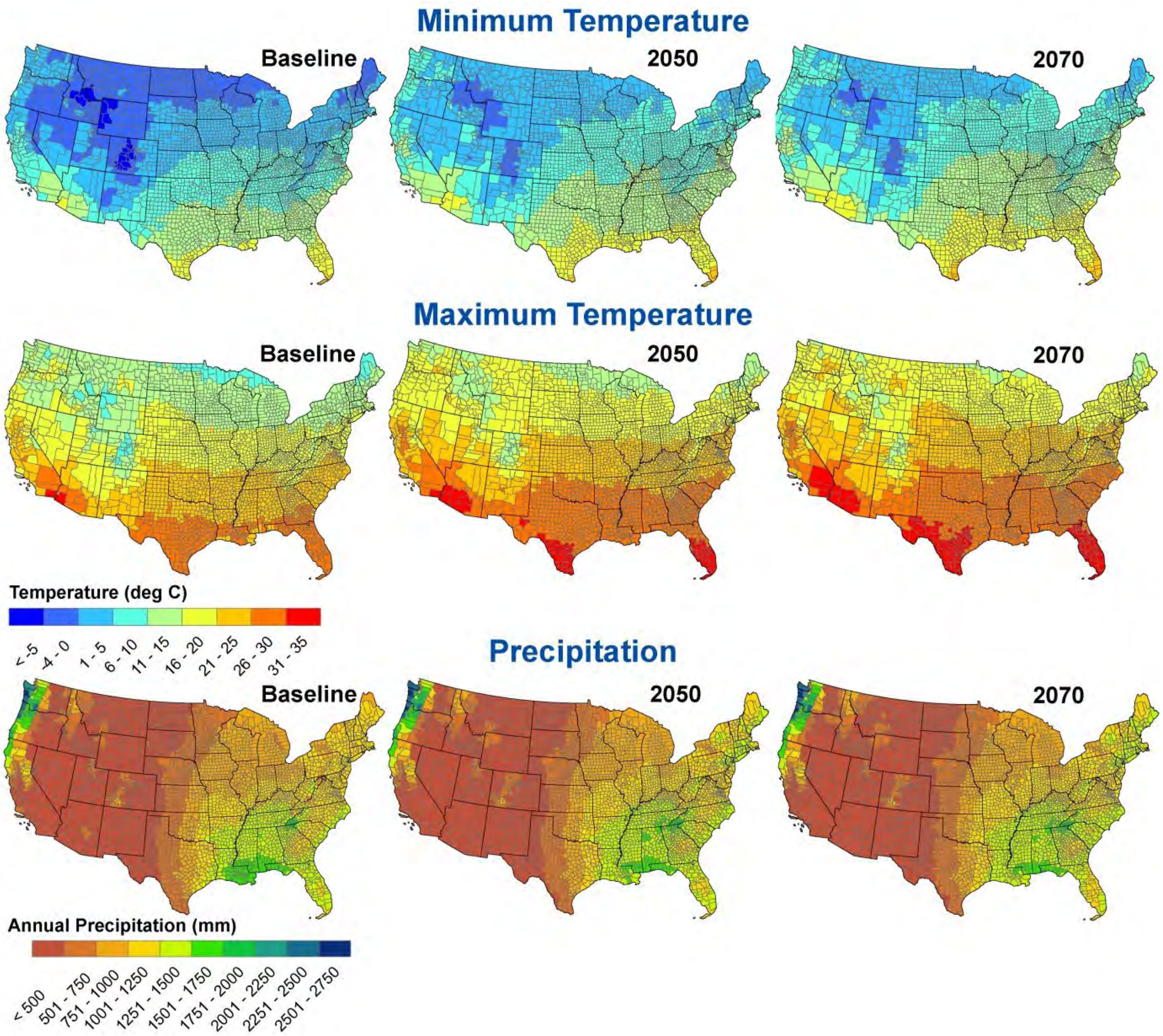


Figure 13.2 | Geographic distribution of baseline T_{min}, T_{max}, and annual precipitation for U.S. counties and projected changes for 2050 and 2070 for the Representative Concentration Pathway (RCP) 8.5 scenario. Maps represent the average of results for 11 different ESM configurations.



between climate variables and relative yields within PRISM-EM would continue to be valid into the future.

Scenarios of current and future climate from ESMs were based on the WorldClim project, which developed global, high-resolution data for historical climate conditions (Hijmans et al. 2005). The data for the current climate represent interpolated surfaces using weather stations from around the world, as well as elevation information to account for the influences of topography on climate. Variables used for modeling energy crop yields for the baseline period of 1950–2000 include annual average T_{min}, annual average T_{max}, and total annual precipitation. Annual averages for each variable in each year of the 1950–2000 baseline period were averaged to generate a 51-year climatology of baseline conditions. When aggregated to U.S. counties, the spatial gradients in temperature and rainfall across the United States are clearly visible (fig. 13.2). For example, WorldClim captures the latitudinal gradient in temperature associated with both T_{min} and T_{max}, as well as the effects of mountains such as the Appalachian Mountains in the Southeast and the Rocky Mountains in the West. In addition, the wetter regions of the Southeast and coastal Pacific Northwest are contrasted against the drier regions of the West.

For projections of future climate, WorldClim generates scenarios by downscaling simulations of ESMs from different international modeling groups using the historical WorldClim climatology. The future climate for any given U.S. county is difficult to project with confidence because of uncertainties in future GHG emissions, as well as uncertainties in how the climate will respond to those emissions. To account for this uncertainty in projections of future climate, WorldClim data for 11 different ESM configurations were used (table 13.1 and fig. 13.3). In addition, each ESM configuration was used with four different atmospheric GHG-concentration scenarios, known as the Representative Concentration Pathways (RCPs). The RCPs represent a wide range of alternative assumptions regarding future global GHG emissions and their accumulation in the atmosphere. Each RCP is identified by a number representing the radiative forcing in watts/m^2. Lower radiative forcing (i.e., RCP 2.6) is associated with lower magnitudes of future climate change relative to higher radiative forcing (i.e., RCP 8.5). WorldClim aggregates ESM simulations for two different time periods, 2050 and 2070, with each time period representing a 20-year average centered on that year (i.e., 2050 is the average of the years 2041–2060, and 2070 is the average of 2061–2080). Therefore, climate change-related relative yields for each energy crop and county include a baseline estimate for the current climate (1950-2000) as well as 44 estimates of relative yields (based on 11 ESM configurations, each using four emissions scenarios) for each county in 2050 and 2070, respectively.

Each of the 11 ESM configurations generates a different distribution of temperature and precipitation changes for U.S. counties (fig. 13.2). Most counties experience temperature increases of 4°–6°C by 2070 relative to the baseline period across different ESMs

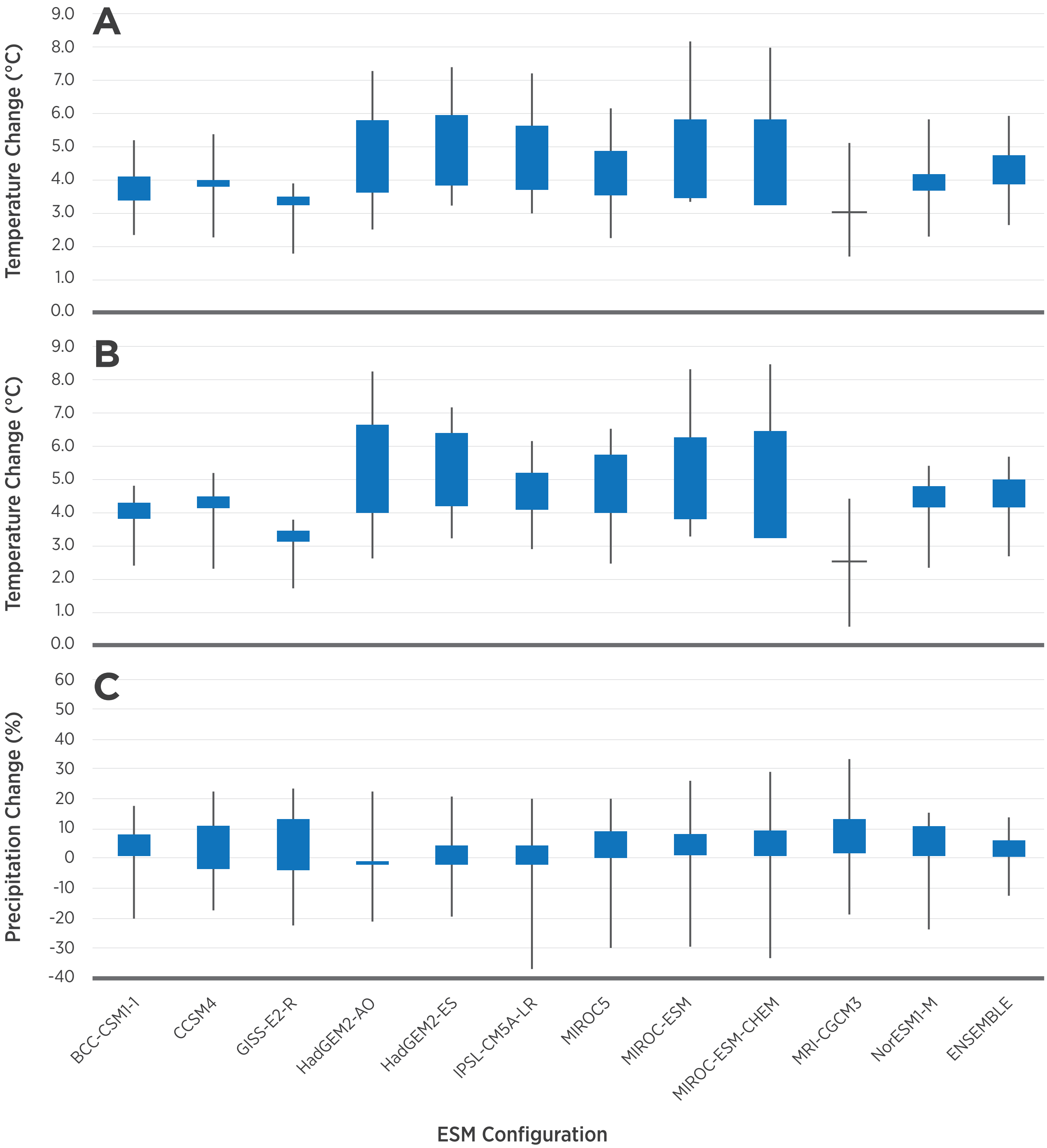
Table 13.1 | ESM Configurations Used in Estimating Energy Crop Yields in Response to Climate Change

ESM	Model Origin
BCC-CSM1-1	China
CCSM4	USA
GISS-E2-R	USA
HadGEM2-AO	United Kingdom
HadGEM2-ES	United Kingdom
IPSL-CM5A-LR	France
MIROC-ESM-CHEM	Japan
MIROC-ESM	Japan
MIROC5	Japan
MRI-CGCM3	Japan
NorESM1-M	Norway

for RCP 8.5. However, increases in temperature in excess of 8°C are projected for some counties. Meanwhile, counties closer to coastal regions experience more modest increases of 2°–3°C. For RCP 2.6, which assumes that atmospheric concentrations of GHG emissions stabilize and then decline over the 21st century, the temperature changes with respect to the baseline are similar for both 2050 and 2070. On average, climate change causes increases in both T_{min} and T_{max} throughout the continental United States. These higher temperatures, and, in particular, higher minimum temperatures, are an important factor influencing the potential future relative yields of different energy crops in different U.S. regions.

While all the ESM configurations project that temperatures increase in all counties with respect to the baseline period (fig. 13.2 and fig. 13.3), changes in rainfall vary significantly in magnitude. Furthermore,

Figure 13.3 | Comparison of the distribution of a) T_{min}, b) T_{max}, and c) precipitation for each of the 11 ESM configurations used to assess the sensitivity of bioenergy feedstock yields to climate change, as well as the ensemble average. Boxes represent the 25th and 75th percentile changes for U.S. counties in 2070 for RCP 8.5. Whiskers represent the minimum and maximum values.



the direction (i.e., increase or decrease) of change in rainfall is difficult to interpret from model results, because differences in rainfall changes among various ESM configurations are masked when results are averaged together. For example, changes on the order of ±40% by 2070 are projected in individual counties for individual models for RCP 8.5, but the average across ESMs is within ±10%. Counties across the northern United States would tend to experience increases in annual rainfall, particularly in the Northeast, while counties in the South would experience declines, particularly the Southwest. These results are consistent with other assessments of model projections of future precipitation changes (Walsh et al. 2014). However, analyses based on a different combination of ESMs generate different results. Furthermore, changes in precipitation are projected to vary among seasons (Walsh et al. 2014), which is an important factor affecting biomass yields.

To estimate future changes in relative yields, the county-level aggregate WorldClim data for each ESM configuration and RCP were used as input to the Bayesian models, resulting in maximum likelihood estimates of relative yields. Yield estimates for each ESM configuration and RCP were subsequently averaged. Analysis of variance was used to test for differences between changes in relative yields for individual ESM configurations and RCPs compared with the WorldClim baseline results. In addition, county-level results were aggregated to the national level using a weighted average, with the weights based on the area in each county identified as cropland in the U.S. Department of Agriculture's (USDA's) 2015 cropland data layer (NASS 2015).

13.3 Results

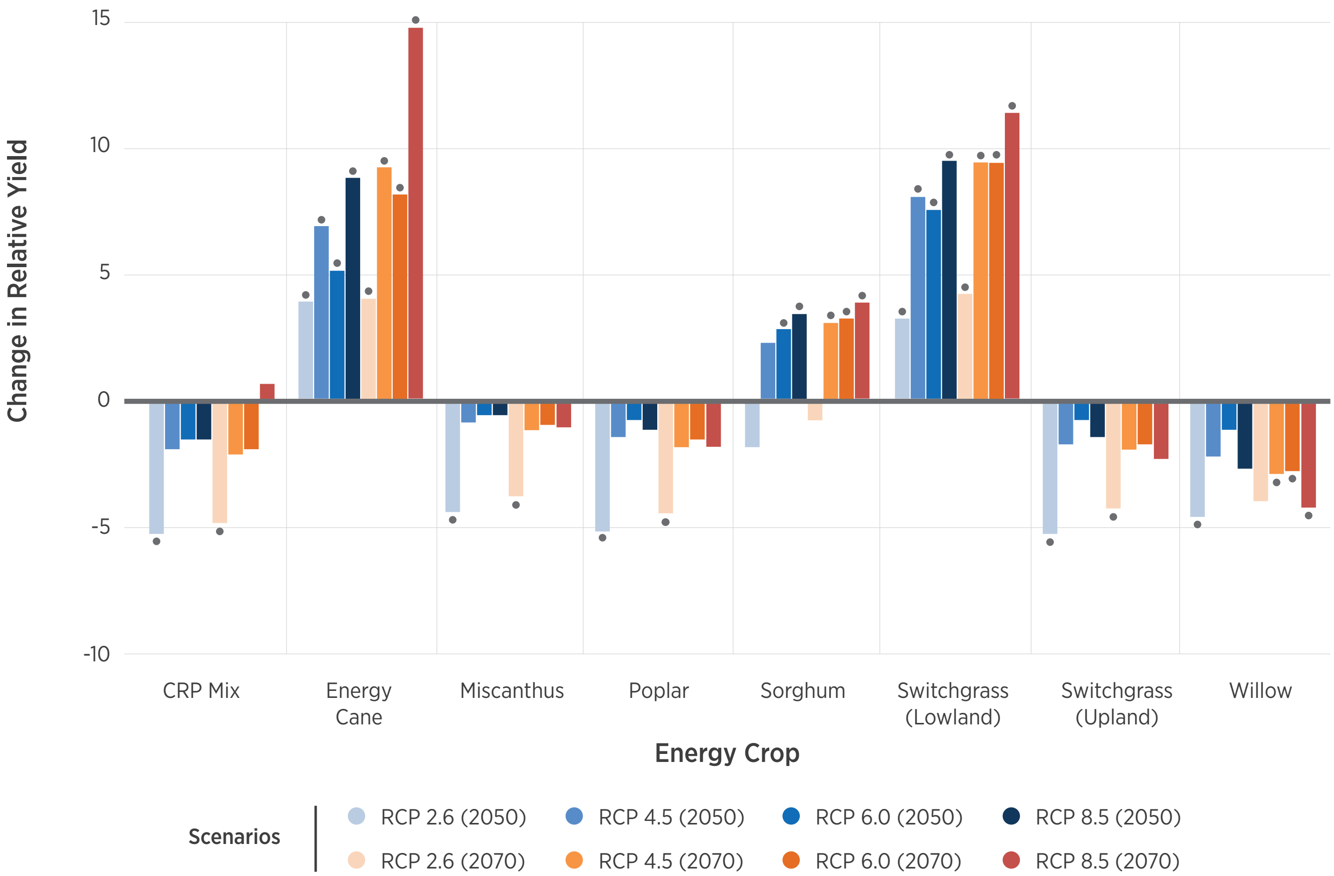
Results of the assessment of climate change effects on cellulosic-energy crop relative yields are summarized here, starting with the presentation of aggregate national results across the different crops. This is followed by the presentation of county-level results for individual energy crops to highlight regional patterns of potential yield effects.

13.3.1 National-Level Results

The aggregate national results (weighted by cropland area) reflect the geographic range of different energy crops as well as the differential sensitivities of crop yields to climate conditions (fig. 13.4). For example, because the most productive areas for energy cane and, to a lesser extent lowland switchgrass, are currently restricted to the warmer climate of the southern United States, these energy crops benefit from climate change and, in particular, higher temperatures. In addition, the benefits increase over time and/or with higher atmospheric concentrations of GHGs (i.e., RCPs). For energy cane, increases range from 4 to 15 percentage points by 2070 among the different RCPs.[2] Similarly, increases for lowland switchgrass range from 4 to 12 percentage points by 2070. These increases are attributable to large increases in yields in the southern

[2] Because relative yield is a percent value by definition, all energy crop modeling results for climate change scenarios are reported as percentage point changes in relative yields as compared with the 1950-2000 baseline.

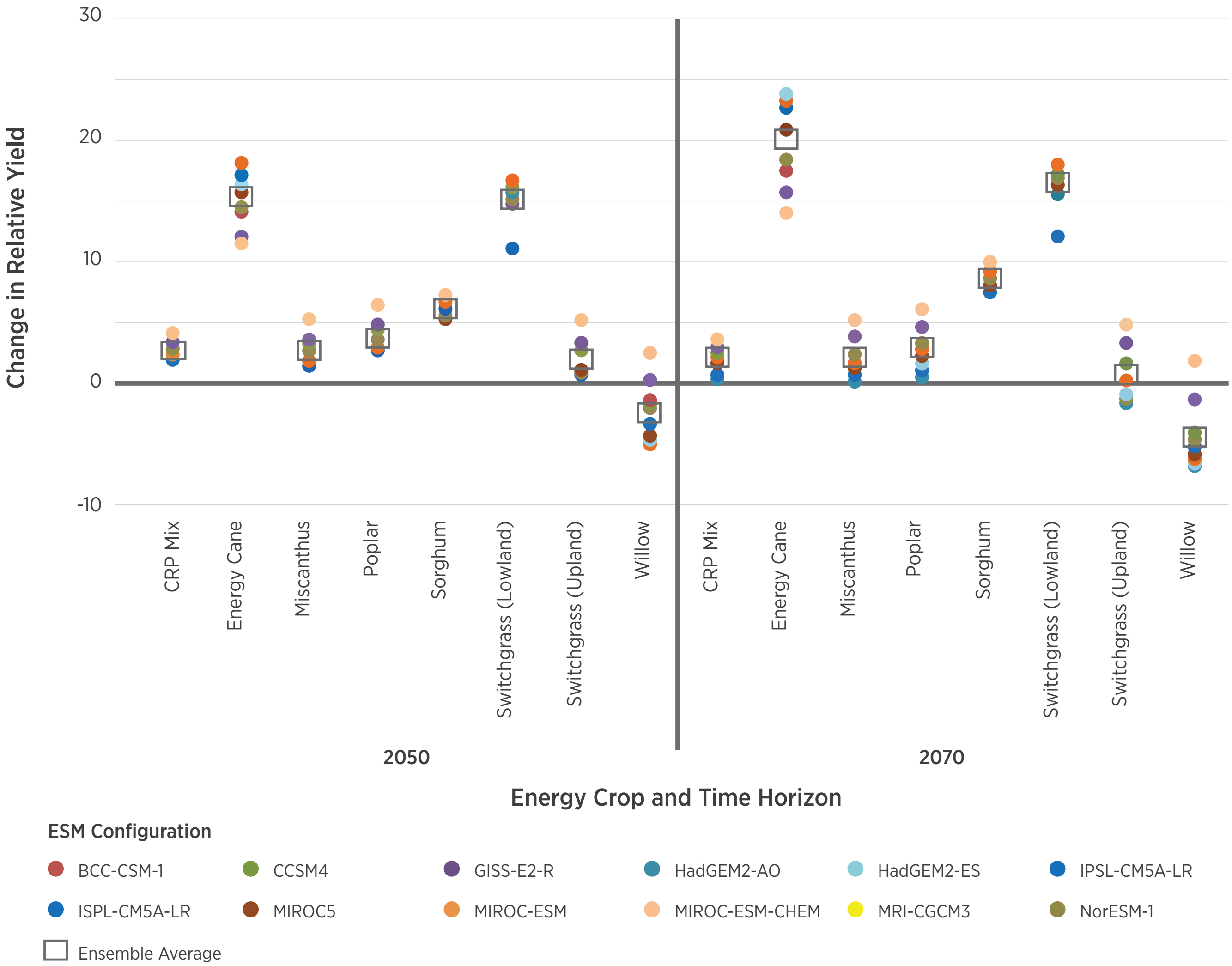
Figure 13.4 | Changes (percentage points) in aggregate national relative yields (weighted by county crop area) relative to baseline (1950–2000) estimates for alternative climate change scenarios. The grey dot (•) indicates a significant difference ($p<0.05$) in relative yields compared with the baseline climate.



United States, Mid-Atlantic, and Midwest (see sections 13.3.2.2 and 13.3.2.6), which reflects a general northward shift in the productive range of energy cane.

In contrast, energy crops that are restricted to cooler climates of the United States, such as upland switchgrass and willow, experience little change or modest reductions in yields. The relatively modest effects of climate change at the national aggregate level are a function of declining yields in some counties for certain energy crops being offset by increases in other counties. This suggests that the long-term changes in average U.S. climate conditions and the associated shifts in the geographic distribution of biomass yields are not necessarily a threat to biomass production at the national level. However, as illustrated in the county-level results, the suitability of a given energy crop for a particular region may change significantly over time. Furthermore, changes in seasonal conditions or changes in extreme events and disturbances may be even more related to biomass yields than long-term changes in average temperature and rainfall.

Figure 13.5 | Changes (percentage points) in aggregate national average relative yields (weighted by county crop area) compared with the 1950–2000 baseline. The figure includes results for different energy crops in response to alternative climate change conditions in 2050 and 2070 as represented by different ESM configurations for RCP 8.5.



For a number of energy crops, variability in model results existed among different ESM configurations (fig. 13.5). As a consequence, results for an individual ESM configuration could differ from the ensemble average by up to ±5 percentage points. For most energy crops, the direction of change compared to the baseline was the same among the different ESM configurations. However, for willow and upland switchgrass, different ESMs generated relative yields both higher and lower than those estimated for baseline conditions (fig. 13.5). This suggests greater uncertainty regarding the aggregate sensitivity of these energy crops to changes in climate conditions.

13.3.2 County-Level Results

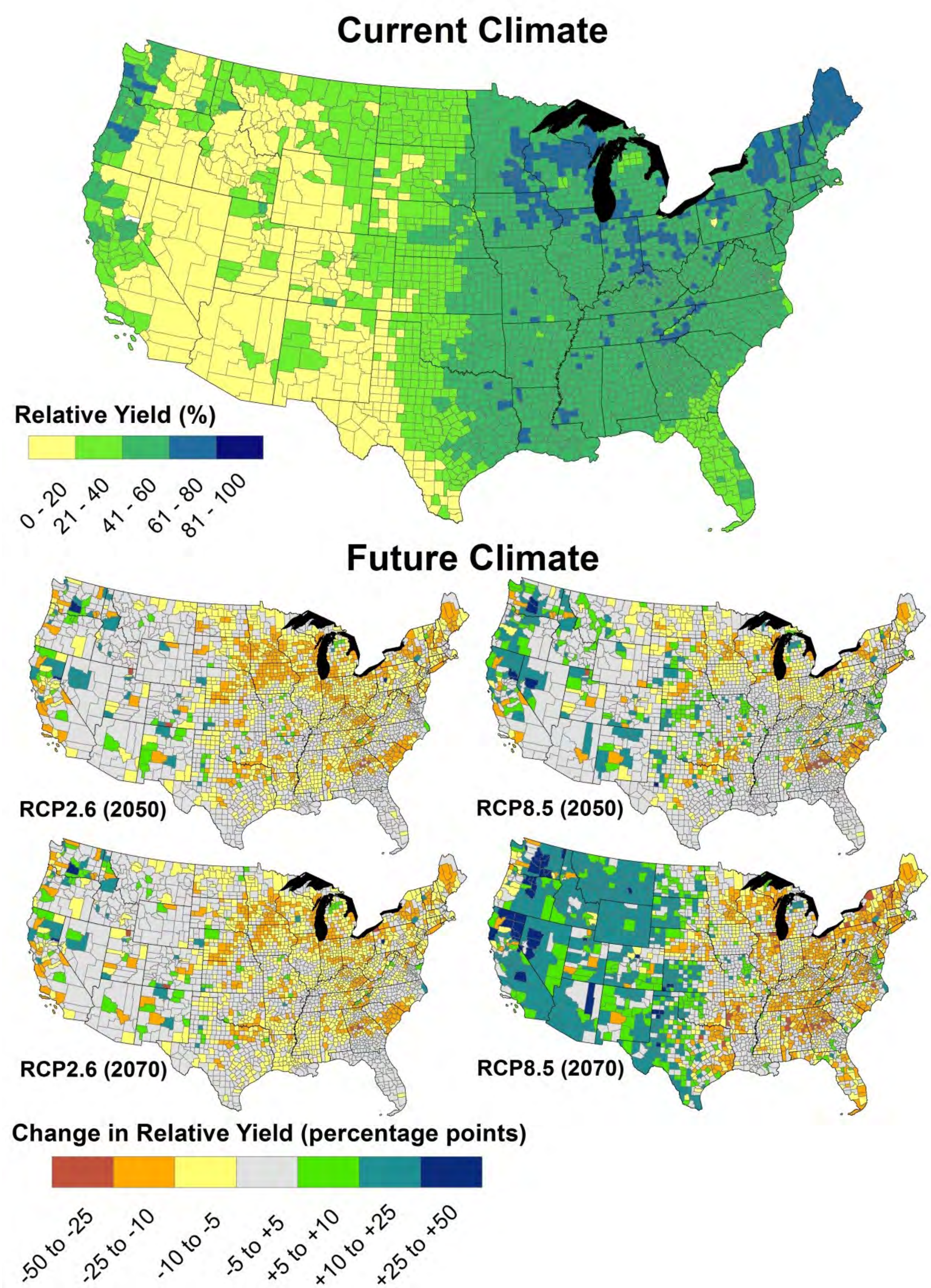
13.3.2.1 CRP Grasses

USDA's CRP encourages farmers to convert highly erodible cropland or other environmentally sensitive land area to vegetative cover. The goal of the program is to reduce soil erosion, enhance water quantity and quality, and provide habitat for wildlife. A wide variety of species and varieties of vegetation are found within CRP seed mixes, with different mixes used in different regions of the United States. Increasingly, a mix of perennial grasses is being explored as a means of maximizing biomass production on CRP lands (Zamora et al. 2013; Venuto and Daniel 2010; Mapemba et al. 2007). Such grasses could be deployed as a biomass resource on other land as well.

At present, much of the eastern United States is conducive to relatively high yields from CRP grasses, as is much of the coastal Pacific Northwest (fig. 13.6). This suggests that areas with higher rainfall and temperature are most conducive to the development of high-yield CRP mixes. The climate change projections reflect a clear east-west division with respect to changes in yields of CRP grasses. For RCP 2.6, relative yields across much of the United States are within ±10 percentage points of baseline values (fig. 13.6). However, yield reductions of 10–25 percentage points are projected in isolated areas of the South (e.g., coastal Carolinas and central Georgia), as well as in the upper Midwest (e.g., Iowa and Wisconsin) and New England (e.g., Maine). In contrast, yield increases of 10–25 percentage points are projected for Appalachia and other isolated areas of the country. Yield effects under RCP 8.5 suggest sharp contrasts between the eastern and western United States , with yield declines of 10–25 percentage points throughout much of the eastern states and yield increases across much of the western states, particularly by 2070 (fig. 13.6).

Although the climate projections suggest there is potential for significant increases in relative yields for CRP mixes across the West, these percentage increases occur in areas with low absolute baseline yields. Therefore, the projected declines in relative yields in the eastern United States are potentially more significant, as these areas have higher absolute yields. In many instances, the yield reductions are less than 10 percentage points; however, larger reductions are projected for some areas, particularly under RCP 8.5. It should also be noted that as CRP vegetative cover comprises a broad mix of species, there may be significant opportunities for adapting the mix of species used in a particular region to reduce adverse consequences and enhance potential benefits of climate change to yields.

Figure 13.6 | Relative yields for a CRP mix of grasses under baseline climate conditions, as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



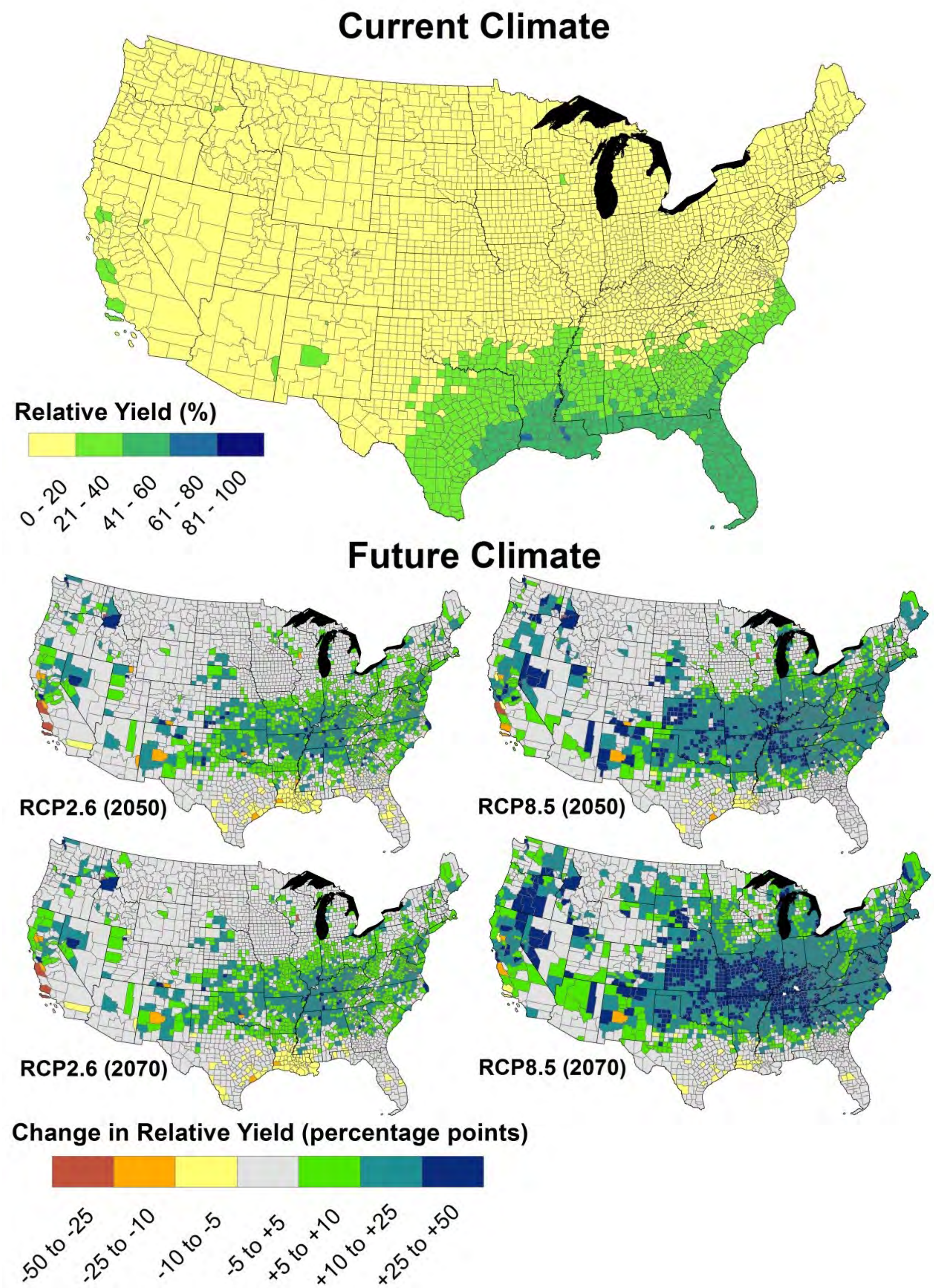
13.3.2.2 Energy Cane

The perennial grass energy cane is a variety of sugar cane selected for high fiber content that enhances biomass yield, making it suitable for use as an energy crop (Matsuoka et al. 2014). Due to poor frost resistance (Sandhu and Gilbert 2014), potential land areas suitable for energy cane production are currently found exclusively in the deep, sub-tropical South (fig. 13.7). The implications of climate change for energy cane yields are most significant for the Southeast. The current high-yield zone along the Gulf Coast remains largely unchanged at ±5 percentage points of baseline values, regardless of the time horizon or RCP considered (fig. 13.7). This result, however, may be an artifact of the modeling. The climate projected for the Southeast by 2050 is unprecedented in the context of other regions of the United States, and thus, there are limited analogues for training the model. However, the physiology of energy cane is known to have temperature thresholds beyond which germination success and photosynthesis plateau or decline. Therefore, higher temperatures across the southern United States may not necessarily drive continual increases in energy cane yields, particularly given the potential for rainfall reductions.

Model results indicate that the current range of energy cane may expand northward significantly under the climate change scenarios. Much of the southern United States, Midwest, and Mid-Atlantic regions are projected to experience an increase in energy cane yields of 5–25 percentage points under RCP 2.6 (2050 and 2070) (fig. 13.7). Such yield increases would likely expand the land area that is viable for cultivation of energy cane as an energy crop. Under RCP 8.5, the yield increases by 2050 are more substantial and widespread—increasing on the order of 10–25 percentage points. By 2070, relative yields increase 25–50 percentage points from northern Georgia, Alabama, and Mississippi, westward to southern Illinois, Kansas, and Oklahoma (fig. 13.7). Although climate change is projected to enhance the suitability of other U.S. regions for energy cane production, for most regions, the increases in relative yields would be less than 5 percentage points. Given that relative yields for much of the rest of the United States are effectively zero, this level of increase is relatively insignificant in the context of cost-effective biomass production.

The limited frost tolerance of energy cane is a significant barrier to the expansion of this high-yielding energy crop into other areas; therefore, a key research challenge is to pursue selective breeding and hybridization to enhance energy cane's frost tolerance (de Siqueira Ferreira et al. 2013; Sandhu and Gilbert 2014). For example, miscane is a hybrid of sugarcane and miscanthus with greater frost tolerance and disease resistance (de Siqueira Ferreira et al. 2013). The projected changes in energy cane yields suggest that climate change will also enhance the ability to expand the range of commercially viable energy cane production in future decades.

Figure 13.7 | Relative yields for energy cane under baseline climate conditions as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



13.3.2.3 Miscanthus

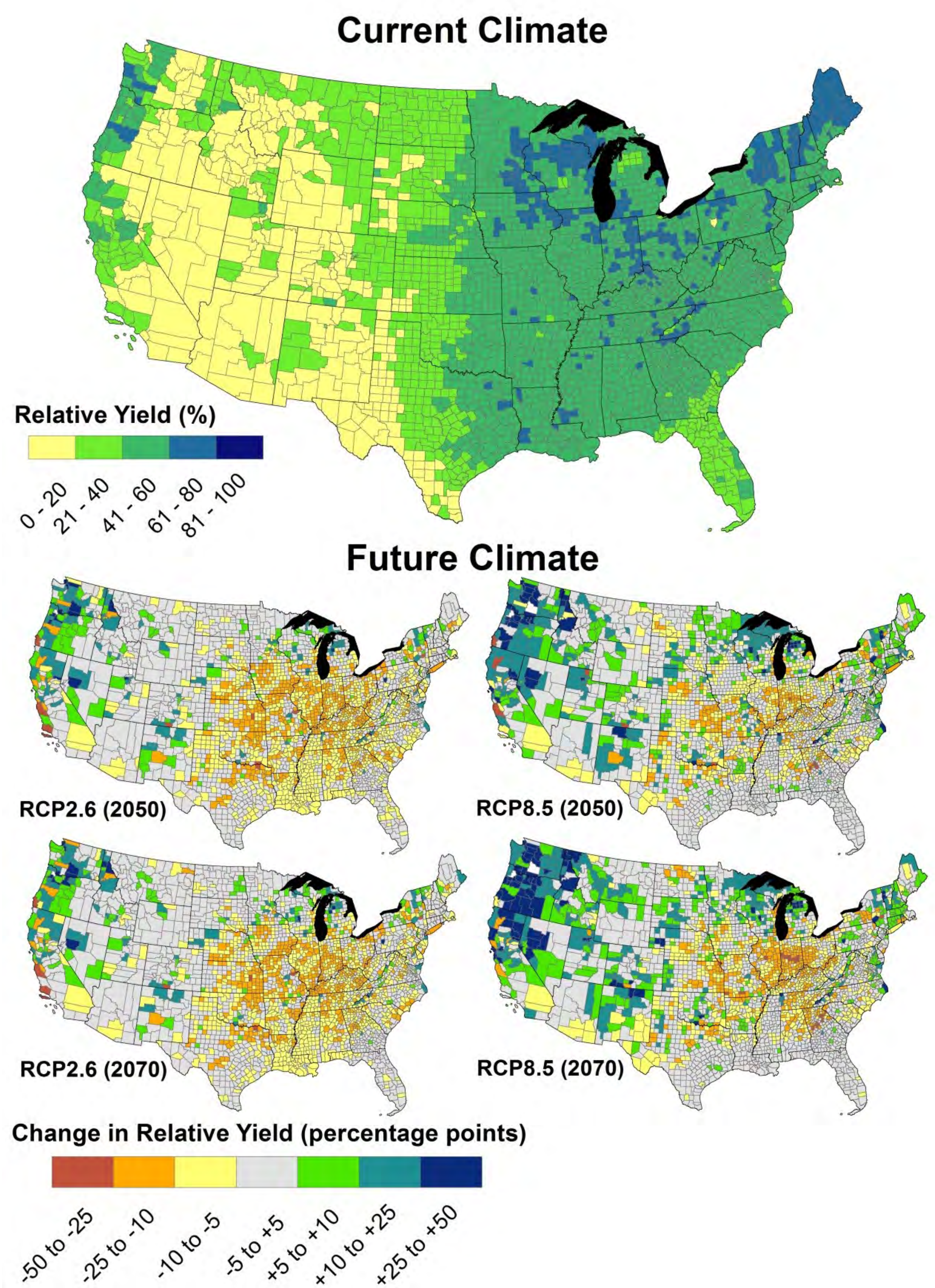
Miscanthus is a large, high-yield perennial grass that is increasingly being developed as a bioenergy resource in the United States for direct combustion as well as for conversion to ethanol (Khanna, Dhungana, and Clifton-Brown 2008; Heaton et al. 2004). Miscanthus is being explored as a biomass energy crop in field trials in various locations around the United States. Modeling suggests that relative yields in excess of 40% can be realized throughout much of the eastern United States under baseline climate conditions (fig. 13.8). Higher relative yields in excess of 60% may be achievable in parts of the Midwest and Northeast.

Modeling of the effects of future climate change on miscanthus yields in U.S. counties suggests that effects of climate change may be modest and transient. Yield changes for much of the continental United States are projected to be within ±10 percentage points of baseline values in both 2050 and 2070 (fig. 13.8). More counties experience positive yield changes under RCP 8.5 compared to RCP 2.6. However, reductions in relative yields of 10–25 percentage points are projected in parts of the Midwest and isolated counties across the South in 2050 and 2070. A number of counties along the West Coast—northern California, Oregon, and Washington—are projected to experience large increases in relative yields.

These increases are larger under RCP 8.5, particularly in 2070. Many of these counties have the potential for modest potential yields in the current climate (fig. 13.8), and thus, could represent new zones for viable production of miscanthus as an energy crop in future decades.

Miscanthus is considered to be an energy crop with moderate tolerance to a range of climatic stressors (Quinn et al. 2015), which explains its potential for widespread cultivation across the eastern United States (fig. 13.8). Selective breeding and hybridization of miscanthus can help address potential problems with survival through the winter during the first year of growth (Quinn et al. 2015; Clifton-Brown and Lewandowski 2000), while also expanding heat tolerance. Projections of changes in miscanthus relative yields in response to climate change suggest that higher temperatures may enhance winter survival, particularly in northern latitudes. However, higher temperatures may also contribute to greater heat stress during summer.

Figure 13.8 | Relative yields for miscanthus under baseline climate conditions as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



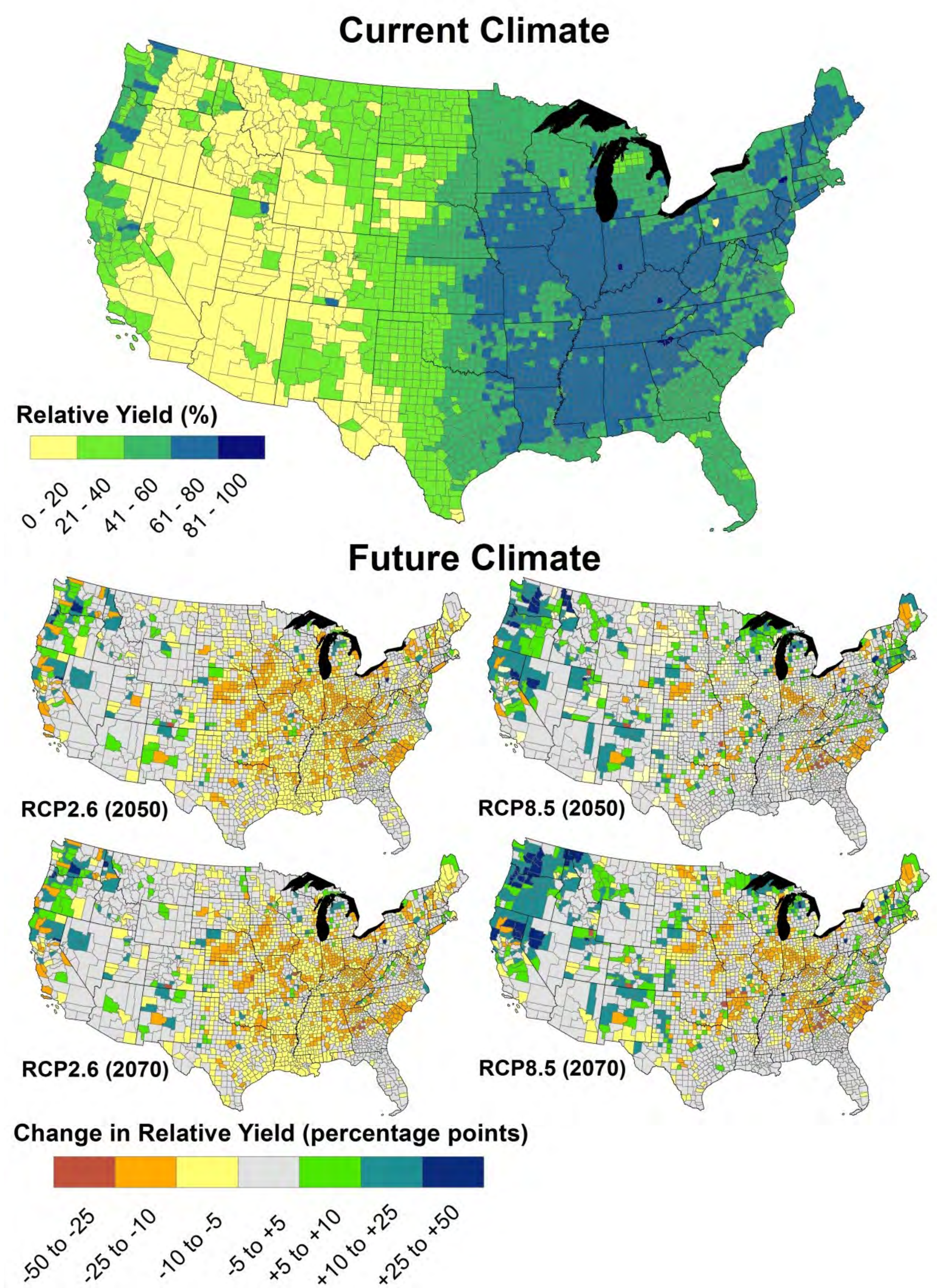
13.3.2.4 Poplar

Poplar (including hybrids of various types) is one of the fastest-growing tree types in the temperate United States. Their rapid growth, ability to grow under a range of site conditions, ability to resprout after harvest, and low requirements for chemical inputs has made them a popular species for commercial forestry (Smith et al. 2009). However, these same qualities also make poplar suitable as a source of woody biomass for bioenergy. Modeling of the distribution of poplar relative yields under the current climate indicates high yields are possible throughout the eastern United States, from Louisiana to Maine (fig. 13.9). In addition, conditions are suitable for relatively high poplar yields along the West Coast.

Projections of changes in relative yields for poplar in response to climate change vary significantly when different assumptions are used regarding future atmospheric GHG concentrations. For example, under RCP 2.6, changes in yields are within ±10 percentage points of baseline values throughout much of the continental United States (fig. 13.9). Generally, changes in the Southeast and Midwest tend to be more negative, while changes in the West tend to be more positive. However, a number of counties in the Southeast and Midwest are projected to experience more substantial declines on the order of 10–25 percentage points. In contrast, increases of 10–25 percentage points are projected for parts of Appalachia and some counties in the Pacific Northwest. Differences between 2050 and 2070 under RCP 2.6 are negligible. For the higher GHG concentrations associated with RCP 8.5, yield effects are spatially heterogeneous. By 2050, yield declines of 10–25 percentage points appear in isolated areas of the Midwest, Southeast, and New England. Yet, yield increases of 10–25 percentage points are projected as well. By 2070, adverse yield effects persist, but overall, yields are more positive across the United States and, in particular, the Pacific Northwest.

The genus *Populus* comprises species that are generally tolerant of a range of environmental conditions (e.g., Wang et al. 2012), creating opportunities for the selection of particular species of *Populus* to suit specific sites. Nevertheless, model results (fig. 13.9) suggest that future productivity of poplar is sensitive to changes in rainfall, as well as rising temperatures that could increase the risk of prolonged heat stress. However, model results also suggest there may be trade-offs in yields over different time scales, spatial gradients, and trajectories of future GHG concentrations.

Figure 13.9 | Relative yields for poplar under baseline climate conditions as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



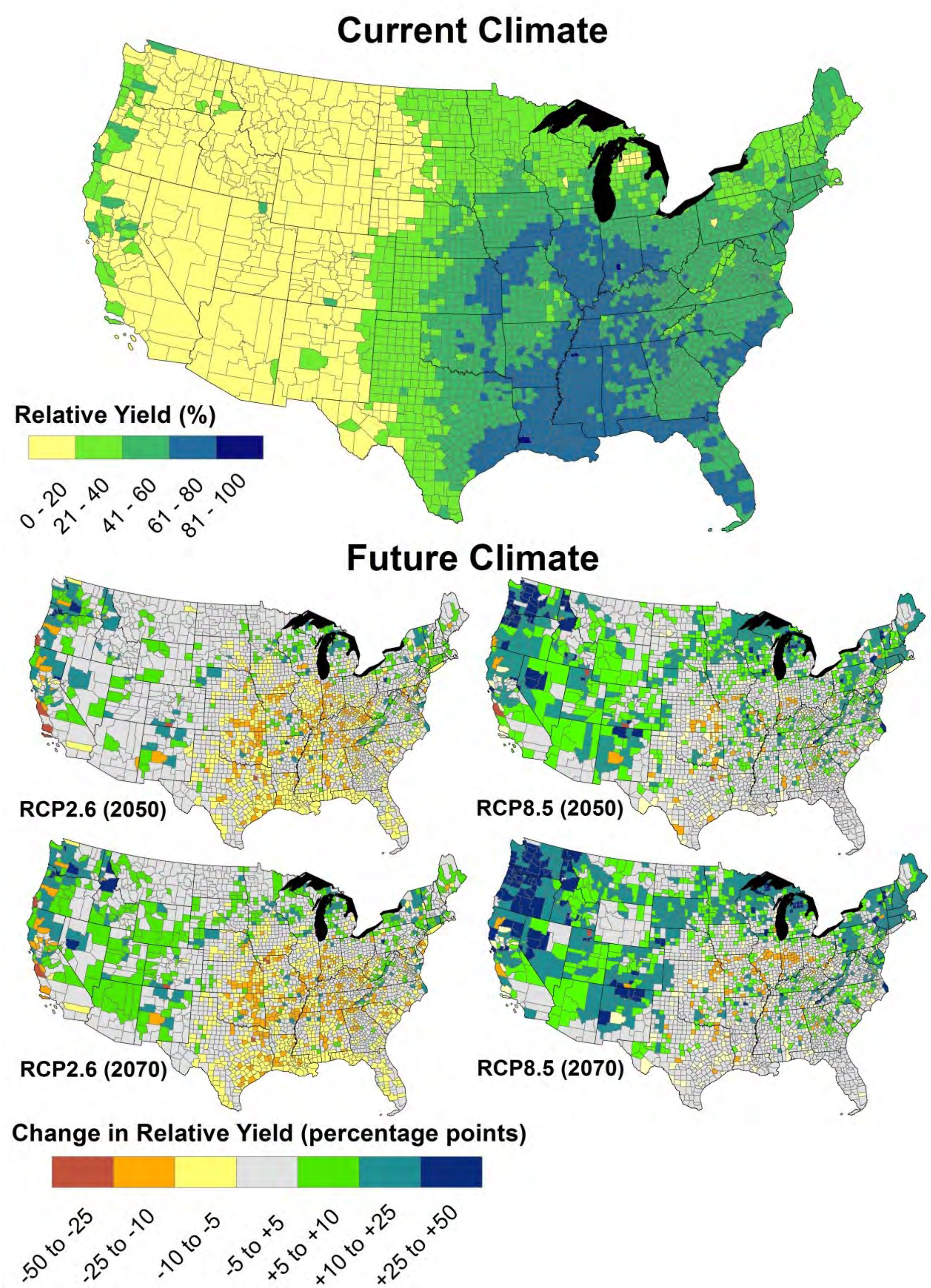
13.3.2.5 Sorghum

Sorghum is an annual, C4 grass with high photosynthetic efficiency. Sorghum is currently produced for livestock feed and table syrups, with an estimated 8.8 million acres of U.S. land allocated for its production in 2015 (NASS 2015; Braun, Karlen, and Johnson 2007). Almost 90% of this area is allocated toward grain sorghum in the arid plains states, from Kansas southward into Texas (USDA 2009; NASS 2015). However, several characteristics make biomass sorghum a useful energy crop for conventional agricultural systems, as well as underutilized agricultural and rural lands. Biomass sorghum tolerates a range of soil conditions, uses nutrients efficiently, and is relatively drought-tolerant due to a deep root system (Regassa and Wortmann 2014; Shoemaker and Bransby 2010).

At present, much of the eastern half of the continental United States has climatic conditions suitable for the growth of biomass (i.e., forage) sorghum (fig. 13.10). The highest yields are associated with the Midwest, lower Mississippi River Valley, coastal Gulf of Mexico, and the coastal Carolinas (fig. 13.10). This distribution is indicative of a preference for mild to warm conditions with plentiful rainfall. Under RCP 2.6, projected relative yields of sorghum change little (i.e., ±10 percentage points) from current baseline values in both 2050 and 2070. However, declines in relative yields of 10–25 percentage points are simulated among some central plains counties in Kansas, Oklahoma, Missouri, and Arkansas, which are currently a center for sorghum production (USDA 2009). Meanwhile, increases of 10–25 percentage points in relative yields of sorghum are projected for a number of counties along the West Coast. This general pattern of response to climate change is also reflected in model results for RCP 8.5. However, relative yields tend to be higher for RCP 8.5 across the United States relative to RCP 2.6, particularly in New England, Appalachia, the northern Plains States, and the West, where increases are frequently in excess of 25 percentage points. Hence, some of the areas that are projected to experience reductions in relative yields with RCP 2.6 are projected to experience increases with RCP 8.5 (for both 2050 and 2070).

The suitability of biomass sorghum for a broad range of climatic conditions increases the resilience of the crop as the climate changes. This is evidenced by the projected modest effects of climate change on relative yields of biomass sorghum, even assuming high atmospheric concentrations of GHGs and relatively long (i.e., 2070) time horizons. However, it is interesting to note that some counties where sorghum yields are projected to decline the most (i.e., Kansas and neighboring vicinities) also comprise the region currently associated with the highest concentration of grain sorghum production. Furthermore, those areas that are identified as having the greatest yield potential for biomass sorghum in the baseline climate (fig. 13.10) are not necessarily those where production is currently concentrated. While sorghum performs well relative to alternative crops in the more arid West, other crops may be more economically viable in areas of the eastern United States that receive more rainfall. However, when sorghum is grown for forage on underutilized agricultural land rather than for grain in conventional agricultural production, such competition is alleviated.

Figure 13.10 | Relative yields for sorghum under baseline climate conditions as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



13.3.2.6 Switchgrass (Lowland and Upland)

Switchgrass, a perennial herbaceous plant, has been described by the U.S. Department of Energy as a "model" high-potential energy crop (Patt et al. 2010). Productivity of switchgrass is dependent upon the selected cultivar and the environmental conditions under which cultivars are grown. Lowland cultivars tend to have higher yields but reduced cold tolerance, relative to upland cultivars, which limits the northern geographic limit of viability for the former cultivars. Upland cultivars also have a higher drought tolerance (Stroup et al. 2003). As a consequence, lowland cultivars are anticipated to be most productive in the Southeast and, in particular, the Gulf Coast States of Louisiana, Mississippi, and Alabama (fig. 13.11). Meanwhile, the most productive regions for upland cultivars are concentrated in the Midwest. However, high productivity is also anticipated farther south and in parts of New England (fig. 13.12).

Because relative yield, rather than absolute yield, was modeled, this assessment does not make direct comparisons of yields between lowland and upland cultivars in different regions of the United States. However, it is possible to compare the relative changes in yields between these two sets of cultivars for different U.S. regions (fig. 13.11 and fig. 13.12). This exercise shows that projected effects are consistent with the differential temperature limits of the two cultivars. Relative yields of lowland switchgrass remain largely unchanged in the South under RCP 2.6 (fig. 13.11). However, significant yield increases are projected for the northern U.S., from Minnesota to New England, because of increasingly mild conditions. For RCP 8.5, larger increases in relative yields on the order of 25–50 percentage points are projected for many counties in the North by 2050. In addition, the West is projected to experience significant increases in relative yields. These changes become more pronounced by 2070.

For upland cultivars, higher temperatures associated with a changing climate increase thermal stress, particularly in the Midwest and Southeast, which reduces yields for RCP 2.6 in both 2050 and 2070 (fig. 13.12), whereas the western U.S. is projected to experience modest increases in yields. For RCP 8.5, similar yield reductions are projected for the Midwest and Southeast, but larger increases on the order of 10–25 percentage points are projected for other U.S. regions by 2050, particularly on the West Coast. By 2070, relative yields increase by 25–50 percentage points above baseline values for all RCPs, and increases are more widespread throughout the West.

The clear differences in yield responses to alternative climate change scenarios between lowland and upland cultivars of switchgrass emphasize the importance of cultivar selection to the yields that are realized on landscapes. Cultivar selection is therefore a valuable tool for adapting the cultivation of biomass to a changing climate. Consideration for the performance of different cultivars in a changing climate may also help guide the prioritization of characteristics that are enhanced or suppressed through selective breeding and hybridization. For example, while higher temperatures in the North would be beneficial for lowland cultivars (fig. 13.11), they could enhance thermal stress and drought risk in the South. Hence, enhancing lowland cultivars' tolerance to drought and thermal stress may enable them to continue to be productive in the South, as well as become increasingly suitable in the North.

Figure 13.11 | Relative yields for lowland switchgrass under baseline climate conditions as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).

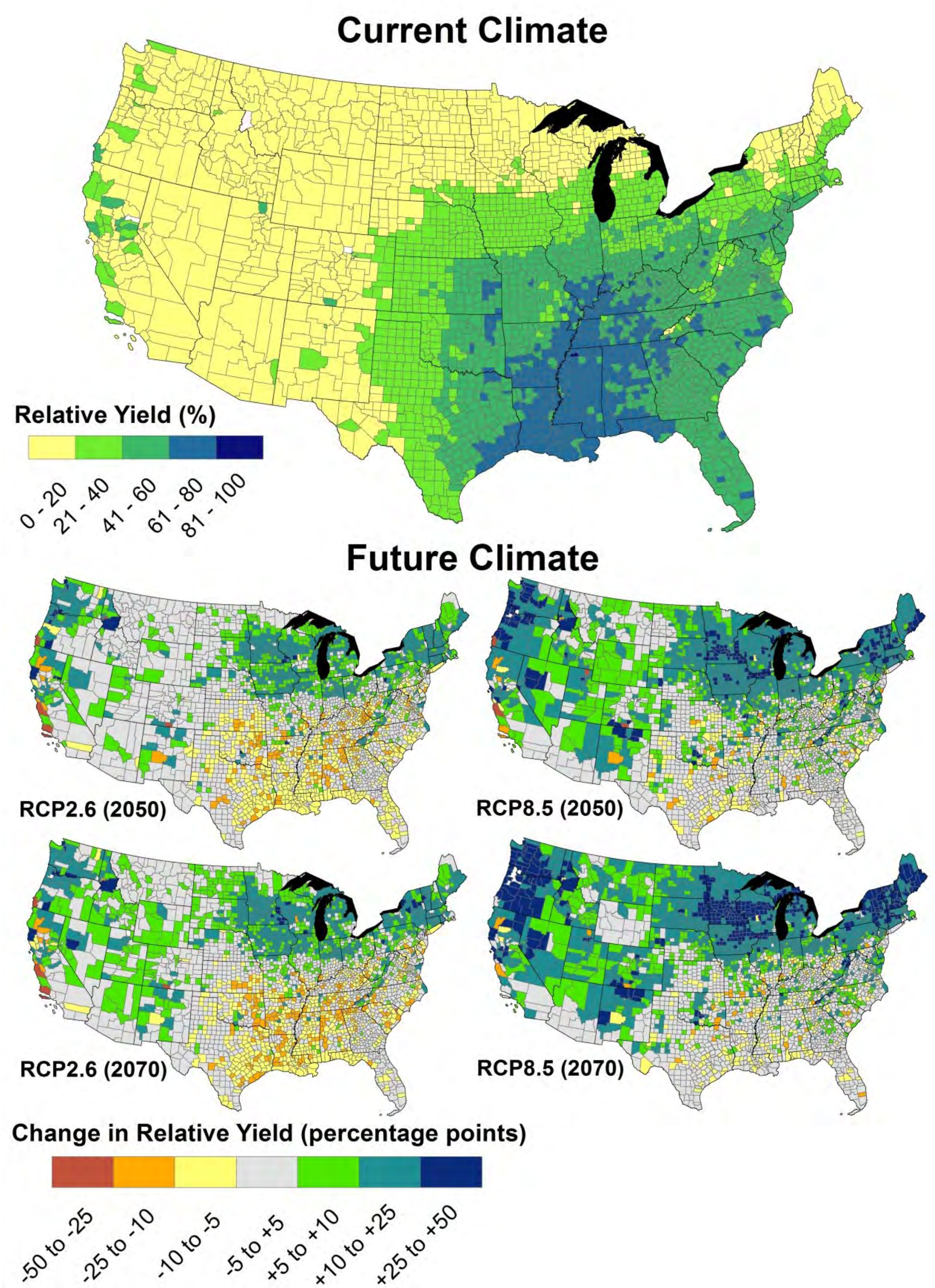
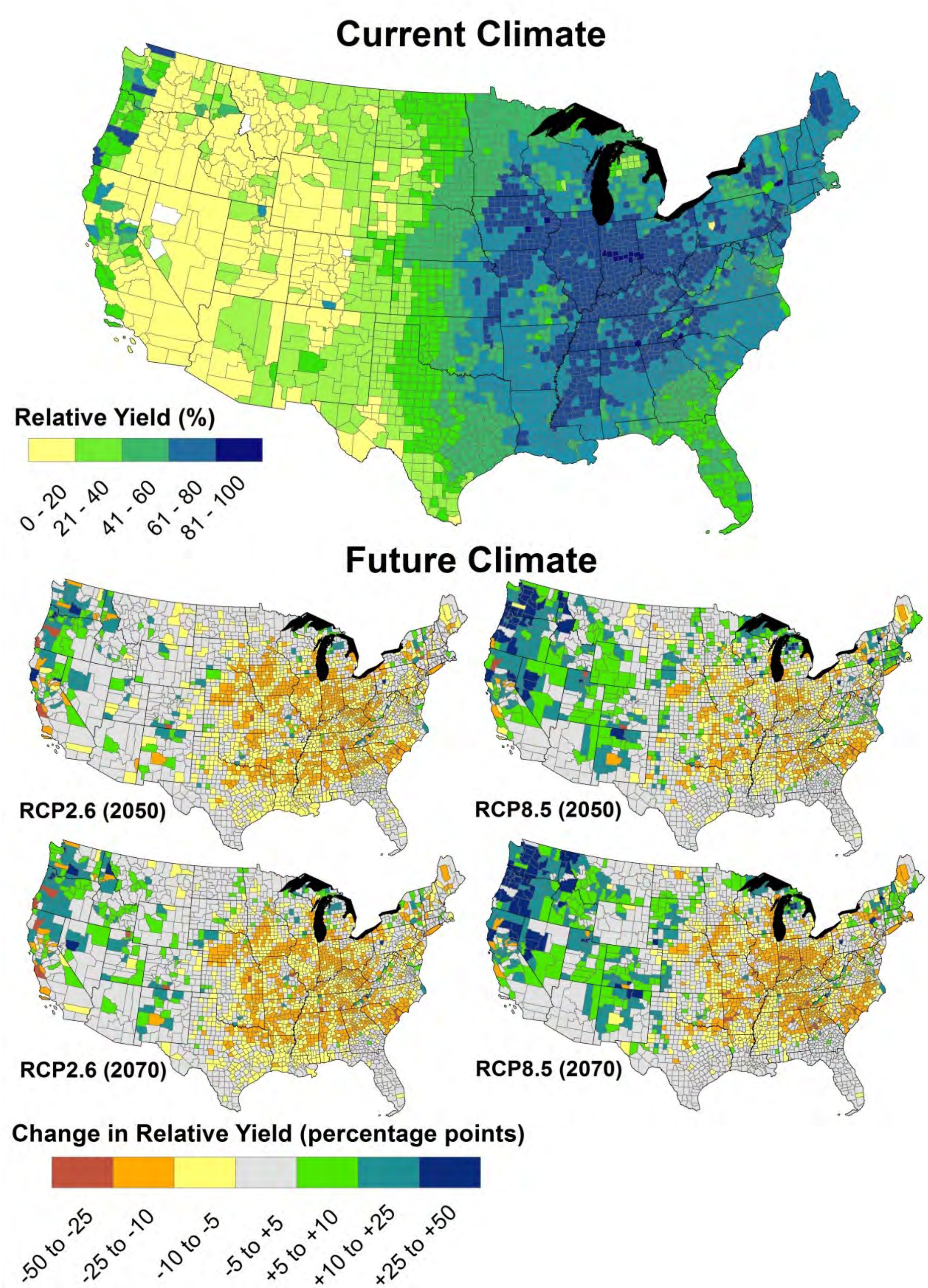


Figure 13.12 | Relative yields for upland switchgrass under baseline climate conditions as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



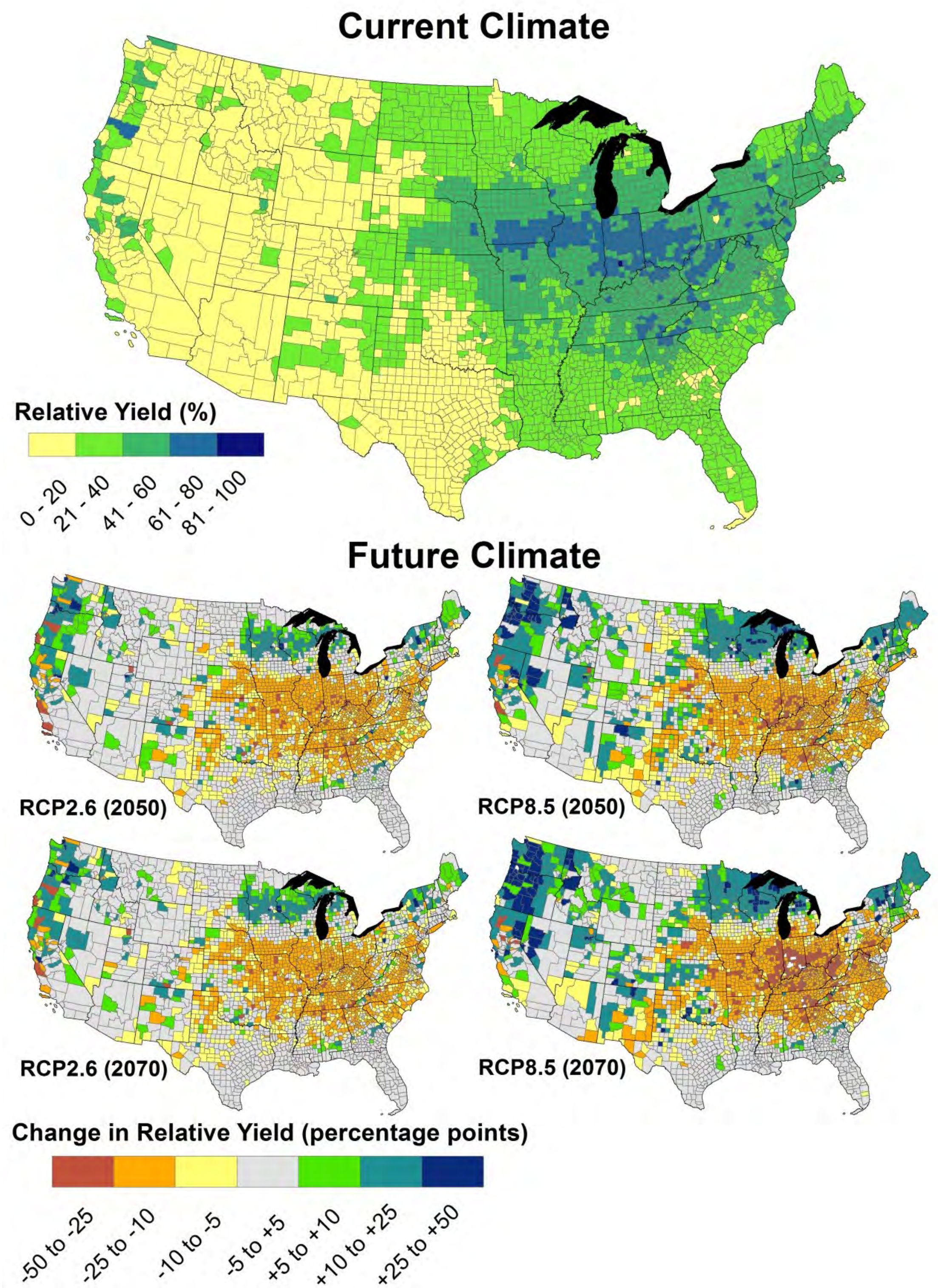
13.3.2.7 Willow

Shrub willow is a perennial hardwood that is considered to be particularly useful for biomass production on underutilized agricultural lands, including those with relatively poor drainage and nutrient content compared to conventional agricultural lands. Like poplar, willow resprouts after coppicing, allowing it to be harvested for 20 years. Under current climate conditions, willow is best-suited to the Midwest, Mid-Atlantic, and New England regions of the United States (fig. 13.13). However, favorable climatic conditions can also be found in northern California and parts of the Pacific Northwest.

The geographic restriction of high-yield willow cultivation to relatively cool climates suggests that willow may be adversely affected by climate change and, in particular, higher temperatures. By 2050, model results project that relative yields of willow could experience declines on the order of 10–25 percentage points in the Midwest and Mid-Atlantic under both RCP 2.6 and RCP 8.5, although the effects are greater under RCP 8.5. Meanwhile, significant increases of 10–25 percentage points or more are projected for the upper Midwest and northern New England. This same pattern arises under RCP 8.5, but is more pronounced. In addition, by 2070, significant increases in yields of 25–50 percentage points are projected for a number of coastal counties in the Pacific Northwest.

The response of willow to climate change indicates that a significant shift in the geographic distribution of willow could transpire over the 21st century. While some regions are projected to become significantly more productive and potentially open up new areas for significant cultivation of willow as a biomass resource, substantial declines in yields are projected over much of willow's current range of climatic suitability. Therefore, ongoing genetic improvements in shrub willow to enhance productivity, improve disease resistance, and reduce production costs (Smart et al. 2005; Smart and Cameron 2008) could be accompanied by efforts to enhance thermal stress and drought tolerance. This could contribute to extending the range of climatic and environmental conditions in which willow can generate high yields.

Figure 13.13 | Relative yields for willow under baseline climate conditions, as well as projected changes (percentage points) in relative yields for different time periods (2050 and 2070) and RCPs (2.6 and 8.5).



13.4 Uncertainties and Limitations

While the modeling results presented here provide some first-order insights into how different energy crops may respond to changes in temperature and precipitation, a number of relevant factors were not incorporated. The development of a more process-based understanding of energy crop responses to changing climatic conditions would assist in reducing uncertainties associated with purely empirical methods. For example, the methods here do not capture the physiological processes of energy crop growth and how climate interacts with each stage of development. Furthermore, the methods here reflect yields as a function of changes in long-term, average climate conditions for a selected group of ESMs. Different ESMs, and therefore ESM ensemble projections, produce different estimates of changes in temperature and, especially, rainfall. Although downscaling methods can help address uncertainties caused by topography, such as mountain ranges, they can also introduce additional uncertainties and biases into model projections (Lo, Yang, and Pielke 2008; Salathe, Mote, and Wiley 2007; Chen, Brissette, and Leconte 2011; Teutschbein, Wetterhall, and Seibert 2011).

Langholtz et al. (2014) argue that extremes of weather and climate are important factors that influence the effects of climate change on biomass. Significant uncertainties remain with respect to projections of changes in the frequency, intensity, or duration of climate extremes, and agricultural models often remain poorly equipped to assess their effects. Yet, capturing the effects of such extremes is an important aspect of understanding the implications of climate change for biomass. Similarly, the results do not account for the effects of changes in atmospheric CO_2 concentration on energy crop physiology and growth, which could have important implications for net energy crop responses to future changes in the climate (McGrath and Lobell 2013). This is particularly important for C3 plants such as poplar and willow (Bishop, Leakey, and Ainsworth 2014; Gielen et al. 2005).

In addition to the direct effects of climate variability and change in energy crop yields, indirect effects can also be important over different time scales. Like conventional crops, energy crops are susceptible to pests and disease, which are also likely to respond to a changing climate. These disturbances could have positive or negative effects on biomass, and those impacts may be region- and cultivar-specific. However, such indirect effects are not captured in the current assessment, and they are often poorly represented in agricultural modeling in general.

Finally, as a managed resource, biomass production systems can be improved and modified in response to new knowledge , innovation, and changing environmental conditions. Such adaptation can arise both autonomously and through strategic planning on behalf of the bioenergy industry and supporting institutions. However, the effects of potential management responses to a changing climate, on behalf of individual agricultural enterprises or the bioenergy industry, more broadly are often neglected in modeling the potential of bioenergy. The results presented here are no exception, as they reflect models of the biophysical response of energy crops but not technological, social, economic, or institutional responses. In particular, future decisions regarding water management for biomass production will have a significant influence on biomass productivity. As land managers gain experience with biomass-production systems, more information will become available regarding how energy crop yields respond to different management regimes or technological innovations.

13.5 Discussion

The modeling of energy crop responses to alternative climate change scenarios indicates that, much like conventional crops or other forms of vegetation, energy crops are sensitive to climatic conditions. The U.S. climate is projected to change significantly in coming decades, particularly for regions such as the Midwest and Southeast that are considered productive landscapes for the development of biomass resources (Walsh et al. 2014). Therefore, in considering the future potential of bioenergy as a significant energy resource for the United States, attention should be given to long-term changes in regional climatic conditions, particularly for energy crops with multidecadal lifespans.

Model projections of climate change effects on different energy crops indicate that climate change could alter yields and shift the geographic distribution of commercially important energy crops. However, responses to climate change among different crops are highly variable. This variability is a function not only of geographic variability in current climate and future climate change, but also variability in the inherent sensitivity of different energy crops and cultivars.

Based on changes in climate variables alone, both significant increases and decreases in energy crop yields are projected to occur in future decades given the current genetic composition of crops and levels of technology associated with crop production and the biomass supply chain. These changes may have greater significance at the regional level than the national level. As a managed resource, biomass-production systems can be improved and modified in response to new knowledge, innovation, and changing environmental conditions. Hence, there are significant opportunities for adaptation to maintain or even enhance the supply of biomass for energy. However, this can be aided by greater focus on the implications of climate change on the long-term strategic selection and deployment of energy crops across the U.S. landscape.

13.6 Summary and Future Research

Climate change is likely to drive changes in the geographic distribution of energy crops. However, there are significant opportunities for adaptation to maintain or even enhance the supply of biomass. This process can be aided by greater focus on the implications of climate change on the long-term strategic selection and production of energy crops across the U.S. landscape. For example, agricultural crop models and/or other physiologically and process-based models for projecting the responses of energy crops to climate change could be coupled with ESM projections of future climate change (Langholtz et al. 2014). However, this integration may require more focused efforts to incorporate knowledge generated by field trials associated with different energy crops and cultivars into agricultural modeling frameworks (Surendran Nair et al. 2012). Furthermore, more rigorous application of ESM projections could enable analysis of the transient response of energy crop yields over different time scales and in response to short-term climatic variability, as well as long-term average climate conditions and atmospheric CO_2 concentrations.

In addition to improving the modeling of energy crop responses to climate variability and change, there are also significant opportunities for adapting energy crops to a changing climate. These include the following (Langholtz et al. 2014):

- Continued investments in the genetic improvement of energy crops in general, as well as specifically for climate-related stress
- Improved management practices to reflect climate change implications for plant establishment, maturation, and harvesting
- Strategic planning for the deployment of different energy crops and cultivars to maintain biomass yields as the climate changes
- Evaluation of the implications of shifting energy crop yields and economic competitiveness for the biomass supply chain, including transportation and refining.

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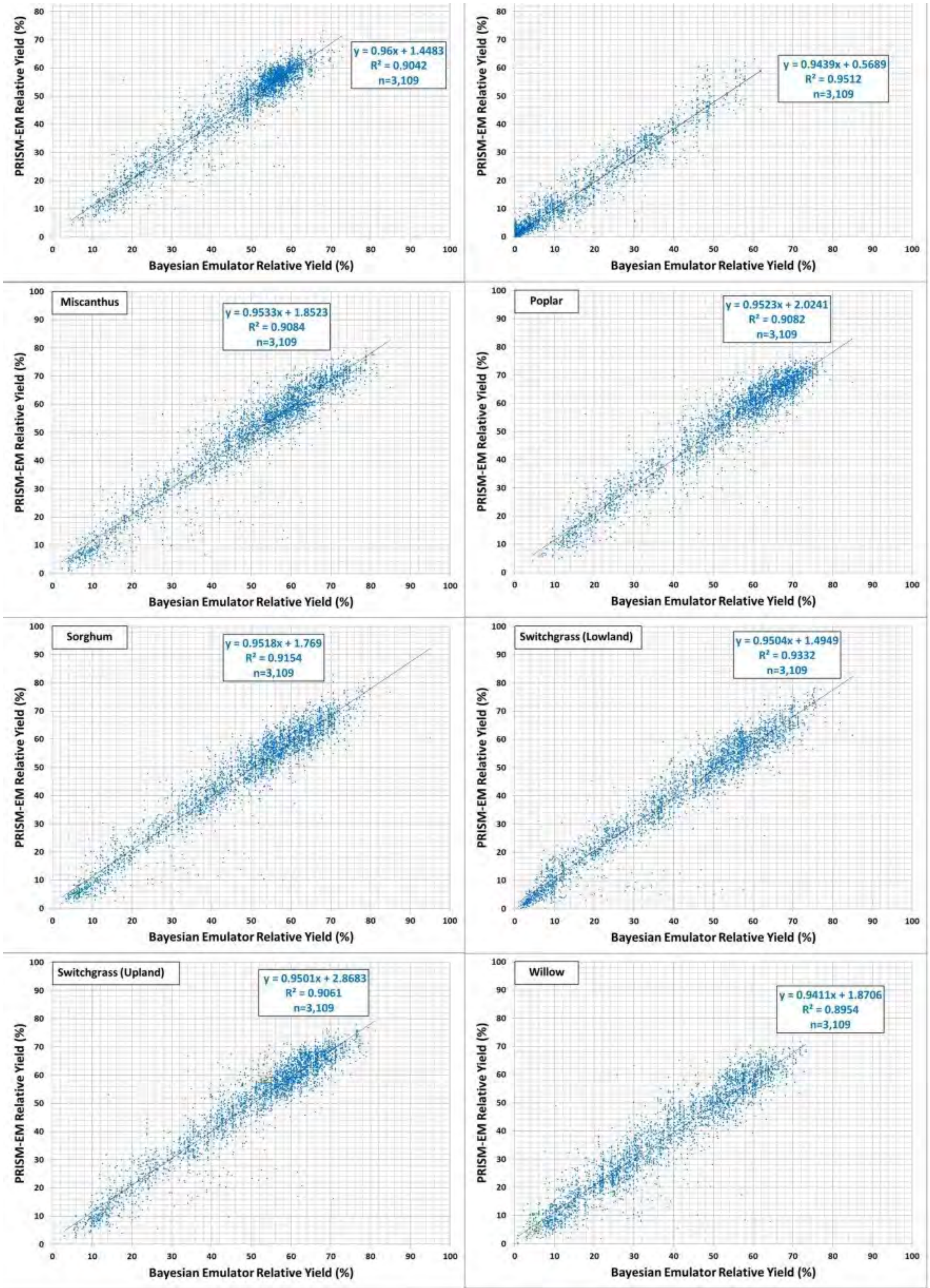
Appendix to Chapter 13: Additional Details on Model Validation

13A.1. Validation Statistics for Bayesian Models

In order to validate the performance of the Bayesian models, model results under the baseline climate (30-year averaged T_{min}, T_{max}, and total rainfall from PRISM) for annual relative yields for all counties considered in the analysis (3,109) were compared against estimates from the Bayesian models (fig. 13A.1). Bayesian models explained over 90% of the observed variance in relative yields for all energy crops (R^2 ranging from 0.90 to 0.95), and the slopes of the regression lines were close to 1 (ranging from 0.94 to 0.96).

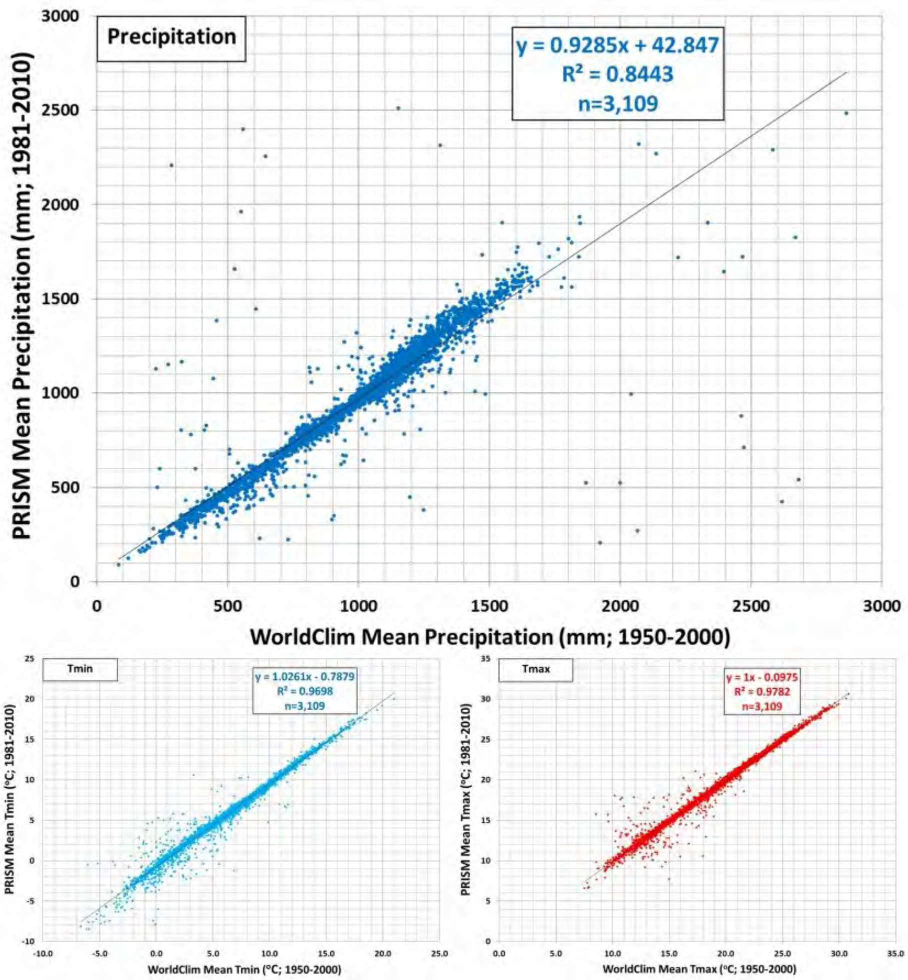
Figure 13 A.1 | Validation plots for Bayesian models of county feedstock relative yields. For each feedstock considered in this assessment, the figures plot the relative yield for PRISM-EM averaged over 30 years (1980–2013) against the yields predicted by the Bayesian graphical models using the PRISM historical climatology.



13A.2. Comparison of Worldclim and PRISM Historical Climatologies

The Bayesian models were trained with historical PRISM temperature and precipitation data, and relative yield data from PRISM-EM. However, projections of future climate conditions and yields were based on the WorldClim dataset, and projected yields were evaluated against those associated with the baseline WorldClim climate conditions. Because the baseline climate for WorldClim was developed using different methods, data, and time horizon than PRISM, it was necessary to test for homogeneity between the PRISM and WorldClim baseline climate conditions for T_{min}, T_{max}, and annual total precipitation. Significant discrepancies between the two data sets would raise questions as to whether the responses generated by the Bayesian models using WorldClim data are reasonable representations of the relationships between climate and yields within PRISM-EM. Comparison of T_{min}, T_{max}, and total annual precipitation between PRISM and WorldClim baseline data using least-squares linear regression indicates close agreement ($R^2 = 0.97$ for T_{min} and T_{max}, and 0.89 for total annual precipitation) (fig. 13A.2). However, significant discrepancies for precipitation were observed between the two data sets for a small number of counties, which likely explain outliers in Bayesian model yield projections observed in the validation of the Bayesian models (see section 13A.1).

Figure 13A.2 | Comparison of the distribution of T_{min}, T_{max}, and precipitation between the PRISM historical climatologies and the WorldClim historical climatologies.



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14 | Synthesis, Interpretation, and Strategies to Enhance Environmental Outcomes

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14.1 Introduction

This report investigates the potential environmental effects associated with select biomass production scenarios across the United States in the *2016 Billion-Ton Report: Advancing Domestic Resources for a Thriving Bioeconomy* (*BT16*), volume 1. *BT16* volume 1 (released in July 2016) evaluates potential biomass that could be available for use—at specified prices, assuming a future market for the biomass. *BT16* volume 2 is a first effort to analyze a range of potential environmental effects associated with select near-term and long-term biomass-production scenarios from volume 1. As with volume 1, this report does not assume particular policy conditions. This report takes the broad approach of including environmental indicators that would be of interest to a range of stakeholders. Environmental effects of biomass production that are modeled include effects on soil organic carbon (SOC), greenhouse gas (GHG) emissions, water quality, water quantity, air emissions, and biodiversity. Land-management changes associated with the scenario transitions are also described and discussed.

BT16 volume 2 seeks (1) to advance the discussion and understanding of environmental effects that could result from significant increases in U.S. biomass production and (2) to accelerate progress toward a sustainable bioeconomy by identifying actions and research that could enhance environmental benefits while minimizing negative impacts of biomass production. Therefore, this chapter synthesizes key results from the report, discusses this chapter synthesizes key results from the report, discusses key uncertainties and limitations, and then focuses on and then focuses on strategies to enhance environmental outcomes of commercial-scale biomass production.

This chapter returns to the initial questions from the Introduction (chapter 1):

- What are the land-use change (LUC) implications of the scenarios over time?
- What are the estimated values of environmental indicators and how do those compare among scenarios?
- What are the potential negative environmental effects, and how might they be managed or mitigated?
- What environmental benefits are possible, and under what conditions do they occur?
- Where is more research needed with regard to quantifying effects, enhancing benefits, and preventing negative consequences?
- How sensitive is feedstock productivity to climate?

This chapter describes many strategies to enhance environmental outcomes from biomass production, i.e., to enhance potential benefits and reduce potential adverse effects associated with the specific scenarios as well as biomass production more generally. The strategies include applying constraints that limit where and how biomass can be sourced (such as the constraints employed in modeling biomass in *BT16* volume 1); implementing mitigations for specific potential impacts identified in this volume; using waste (that would otherwise be landfilled or incinerated) for energy; applying best management practices (BMPs) and landscape design principles; and integrating biomass harvesting with other activities (e.g., mineland reclamation and invasive species control). Concepts of ecosystem services and monetary strategies are also introduced. Finally, future research needs are discussed.

14.1.1 Synthesis and Interpretation of Results

The analyses in this report begin to illustrate the environmental effects of biomass that could potentially be available for energy or other purposes in the future, given a market, a $60 price per dry ton of feedstock, available land, and many other assumptions that are described in chapter 2 and embedded in the economic production models used in *BT16* volume 1. Results should be interpreted in the context of *BT16*, which includes factors ranging from specific temporal and spatial resolutions of available data to broad national energy needs. Contextual factors to consider in an assessment of environmental effects typically include the purpose of the assessment, the biomass production and distribution system, end use, policy conditions, stakeholder values, location, temporal influences, spatial scale, baselines, and reference scenarios (Efroymson et al. 2013).

Quantitative results in *BT16* volume 2 are highly dependent upon the particular scenario comparisons that are used, but implications are relevant beyond these scenarios. The temporal aspects of *BT16* volume 2 are selected so that most analyses could focus on near-term harvests of residues and future potential production of energy crops. Comparisons of scenarios containing energy crops (e.g., BC1 2040, HH3 2040) with those that do not (BC1 2017) highlight the potential effects of those energy crops. Miscanthus and biomass sorghum, for example, contribute to gains in soil carbon in the 2040 scenarios. Some scenarios have been designed to facilitate interpretations of how environmental effects are influenced by annual yield increases. Higher-yield scenarios result in lower air emissions for terrestrial biomass on a per-ton basis, as well as a lower consumptive water use for algae. The wide variety of algae scenarios highlight effects of different types of cultivation systems and sources and purity of carbon dioxide (CO_2), which affect water consumption and GHG emissions, respectively.

To further interpret the importance of the environmental effects, they could be compared to effects under alternative land uses and alternative energy production systems. For example, the air emissions analysis (chapter 9) notes that biomass production activities may replace (rather than occur in addition to) current activities and, therefore, may not pose air quality challenges as results might suggest. While a complex business-as-usual scenario is beyond the scope of this report, reference scenarios, an agricultural baseline, and fossil energy comparisons are used in some analyses.

The analyses reflect effects of LUC (land management) transitions associated with simulated biomass production. LUC is important because all social, economic, and environmental indicators of sustainability can be affected by LUC (McBride et al. 2011; Dale et al. 2013). Since 2008, effects of LUC have dominated discussion of bioenergy sustainability because of their implications for GHG emissions, biodiversity, food security, and other aspects of sustainability.

The primary type of LUC associated with *BT16* biomass supply scenarios is the land management practices that accompany transitions of up to 45 million acres of annual crops to perennial cover by 2040. Replacing annual crops with perennial crops has multiple environmental advantages, such as reducing soil erosion, increasing carbon sequestration (chapter 4), improving water quality (chapter 5), and providing higher-value habitat for wildlife (Robertson et al. 2008; Dale et al. 2011). Unlike annual crops, perennial crops can generally be grown with minimal inputs of fertilizer, pesticides, and irrigation (chapter 8) (Chamberlain and Miller 2012; Dale et al. 2011). Management of perennial crops typically involves less-frequent physical disturbance (e.g., tillage, seeding, cultivation), and harvests can be timed to avoid critical life history events for wildlife (Gamble et al. 2015; Roth et al. 2005). Indeed, chapter 10 recommends perennial crop management of this type to mitigate potential habitat quality losses for particular bird populations. In this study, energy crops show favorable performance relative to conventional feedstocks.

Historical land use in different regions is a major element affecting scenario comparisons. For example, an increase in soil carbon (i.e., a carbon sink) is simulated when transitioning from historical cropland to energy crops, whereas a transition from pastureland to energy crops does not always increase soil carbon (except in the case of miscanthus and biomass sorghum). Land management changes on forestland are assumed to be minimal, involving thinnings and harvesting of whole trees and residues but not involving new road building or transitions into or out of forest.

The location and type of biomass have also been found to be major factors affecting the direction and magnitude of environmental changes that were estimated. Most counties analyzed in the scenarios show potential for a substantial increase in biomass production to support a growing bioeconomy with minimal or negligible effects on water quality, water quantity, avian diversity (as analyzed in chapter 10), or air pollutant emissions, under the biomass supply constraints assumed in *BT16*. Cellulosic biomass generally shows favorable performance relative to conventional feedstocks for the indicators investigated, with harvest of agricultural and forestry residues generally showing the smallest contributions to changes in certain environmental indicators. However, in some locations and under some biomass scenarios, challenges may arise for maintaining SOC levels, water quality, water availability, biodiversity, and air quality.

The regional context influences the significance of the environmental effects that are estimated in *BT16* volume 2, and it is also important to note that factors besides biomass production affect the environmental indicators investigated here. For example, the air emissions analysis (chapter 9) found that some counties already in nonattainment in 2015 for National Ambient Air Quality Standards could see emissions representing greater than 1% of the National Emissions Inventory for those counties. The chapter notes that the spatial distribution of modeled air emissions, including those not associated with biomass production, would need to be understood before an estimate of local air quality could be made. The water footprint analysis (chapter 8) discusses the importance of considering the context of water withdrawals, such as those from the Ogallala Aquifer, before fulfilling water needs of new activities. Similarly, loadings to waters would need to be placed in the context of local water-quality criteria. The algae chapter (chapter 12) reviews many of the indicators and indices of water quantity that incorporate regional needs, such as environmental flow requirements for fish. Going beyond the environmental effects analysis in this volume is critical to place the indicators in a regional context.

In reality, environmental effects are often cumulative. The analyses of forest water quality, water quantity, and biodiversity focus on the potential environmental responses associated with incremental biomass harvests, without considering effects of total harvests for conventional forest products, as well as residential development. Chapter 11 notes that for some forest species and locations, biomass removal may lower habitat quality such that it reduces local numbers of individuals, thereby increasing vulnerability to other factors affecting the population, such as competition or fragmentation effects.

Most results presented in *BT16* volume 2 represent environmental effects for biomass production and harvesting only (i.e., they do not consider feedstock transportation logistics, biomass conversion, or biofuel combustion). The analyses of logistics in the GHG and air emissions chapters are exceptions; these analyses illustrate the importance of studying environmental effects of later stages of the supply chain for relevant indicators.

A few illustrative cases have been completed to estimate displacement of fossil-derived GHG emissions and energy. Life-cycle GHG intensities for both biomass- and fossil fuel–derived fuel and energy products were applied to specific scenarios based on potential growth in energy, power, and chemical production between now and 2030. These cases illustrate that GHG-emissions reductions (between 4%–9%) and fossil energy consumption reductions could be expected, as compared to a scenario in which all U.S. energy and conventional products are produced from fossil fuels in that year. Results depend on these GHG intensities, the biomass supply, and how the biomass supply is allocated to different end uses.

Other than the illustrative cases showing the potential reductions in GHG emissions and fossil energy consumption, *BT16* volume 2 does not investigate other environmental or socioeconomic effects of displacing fossil feedstock–derived fuel and products. However, determining the net effects of displacing fossil energy and products with biomass-derived energy and products is a critical area for further analysis. Some of the environmental effects of gasoline supply chains are described in Parish et al. (2013) and Dale et al. (2015). For example, environmental effects of gasoline pathways include a shift of carbon from prehistoric times to today's atmosphere, a subterranean dimension of disturbances, and extraction locations in remote and fragile ecosystems that could negatively affect biodiversity.

14.1.2 Uncertainties and Limitations

As stated above and throughout the report, results are limited to particular scenarios, as in all environmental modeling studies. The results must be interpreted in light of the uncertainties in the models used to simulate biomass in *BT16* volume 1 (i.e., POLYSYS and ForSEAM) and models used to simulate environmental indicators in *BT16* volume 2. Volume 2 discusses sources of uncertainty in these analyses, including limited input data for model parameterization and questions about extending models to regions, feedstocks, and time periods for which they have not been calibrated or validated. Some of the uncertainties, such as how fast yields could increase and what conservation practices might be implemented by farmers, are handled through the use of multiple scenarios or cases.

A major assumption in *BT16* is that the agricultural land base and the forest land base do not change between the present and 2040. This assumption has implications for all of the environmental effects analyses, and modifying scenarios to allow transitions between these major land classes could result in environmental changes of different types, magnitudes, or directions than the comparisons presented here.

Model inputs, such as land-cover and land-management classes, are also uncertain, and chapter 3 focuses on those uncertainties. Large uncertainties in basic land-cover classifications are well documented (e.g., Congalton et al. 2014; Kline et al. 2011; Feddema et al. 2005). The classification uncertainties increase when land "use" is inferred from land-cover classes (Lambin et al. 2003), and uncertainties are inherently greater when an analysis attempts to quantify "change" (O'Hare et al. 2010; Dale and Kline 2013). Moreover, crop rotations have not been investigated in this study, even though they are a common land-management strategy.

Uncertainties in environmental models include presumed mechanisms or processes by which environmental indicators respond to changes in land management, as well as uncertainties in the drivers of change on which empirical models are based. Chapter 6 develops empirical relationships between forest harvest area and water quality but notes that if sufficient data and process-based platforms for silvicultural activities were available, a modeling approach that considers soil type, topography, climate, vegetation, and harvest systems involved in estimating water-quality response to biomass harvests could lead to more accurate results. Drivers of environmental change may be different in different regions. For example, the agricultural biodiversity analysis assumes that bird populations change in response to habitat, as reflected in empirical estimates from local studies. However, in a different location, the response may differ; e.g., if major changes in predator populations occurred in one region but not another.

Similarly, decisions about allocation methods can lead to uncertainties in environmental effects results. For example, allocating GHG emissions or irrigation water to corn grain and not to corn residues could affect conclusions about the effects of harvesting those residues on indicators. The importance of allocation method has frequently been identified as an issue that has a major effect on results in life-cycle analyses.

The county-level resolution is an important aspect of *BT16*. Analyses of environmental effects of terrestrial feedstocks require assumptions about how biomass production—estimated at the county level in *BT16* volume 1—is distributed within a county, especially when watershed-level effects are modeled. For example, the water-yield analysis (chapter 7) finds that increased water yields from biomass harvesting in forests would have little additional effect, relative to a 10-year reference. However, if harvest outputs of ForSEAM had been available for particular locations within the county, the effects of increased water yield could have been more important in some locations. Furthermore, biodiversity results depend on the arrangement of feedstocks across the county landscape.

The potential global impacts of an expansion of biomass production in the United States depend on many factors not analyzed under *BT16* scenarios. Reasonable assumptions about increasing biomass production could generate estimates that not only vary widely in terms of magnitude, but also in terms of direction of the effects, particularly with respect to whether forestland is expected to expand or contract in response to policies associated with biomass production (Kline et al. 2009). Potential international effects of future U.S. biomass production scenarios are not considered, including potential indirect LUC.

14.2 Enhancing Environmental Outcomes: Strategies Identified in this Report

Actual environmental effects of future biomass production depend on production practices that will be used. Strategies that can help move toward environmentally sustainable biomass production are described below. As with conventional agricultural and forestry resources, future potential supplies can be estimated, but the environmental effects and sustainability of these future supplies is wholly contingent upon how those supplies are actually produced in the future. Here, environmentally relevant supply constraints are introduced along with other approaches to realize improved environmental outcomes for biomass production.

14.2.1 Supply Constraints in Biomass Resource Assessments

As described in chapters 1 and 2, various supply constraints were assumed in *BT16*, some of which reflected sustainability principles. Though future biomass production practices are not known with certainty, these supply constraints reflect considerations that can be implemented or assumed at large scales. Environmental considerations that may affect biomass resource potential estimates can be implemented in models by:

- Restricting areas on which bioenergy crops may be grown or residues may be collected. For example, some areas in *BT16* were restricted from production to protect biodiversity. Fragile, reserved, protected, and environmentally sensitive forestland was not eligible for biomass harvests. Algae were not produced on agricultural, forest, or other sensitive lands.

- Restricting energy crop choices or forest biomass harvests to particular locations. For example, the Biomass Research and Development Board recommends selecting perennial crops based in part on water requirements and available water (BRDB 2012). Copeland et al. (2012) assert that species selection should consider effects of different crop choices on regional air quality. This type of restriction was not implemented in *BT16*. Instead, energy crops were allocated along with conventional crops at the county level in a way that maximizes profit from the landowners' perspective.
- Implementing management practices that maintain or enhance environmental outcomes (e.g., tillage type, production intensity, harvest frequency, harvest area, residue removal percentage). Many of the supply constraints in *BT16* relate to management practices. Agricultural residue removal coefficients were employed and constrained not to exceed the tolerable soil loss limit of the USDA Natural Resources Conservation Service (NRCS 2016a; 2016b), and not to allow long-term reduction of SOC. Moreover, energy crops were not irrigated. At least 30% of logging residues were left onsite to protect soil, provide habitat, and maintain soil carbon. The use of some BMPs was assumed and included in cost estimates for forests and agriculture. Harvest levels were restricted to ensure that timber growth always exceeds harvest at the state level.
- Implementing targets for environmental indicators (e.g., regulatory levels or thresholds) that can be linked to productivity estimates. Such targets are quantitative goals for an indicator, usually to be achieved at a particular place and time. (The German Advisory Council on Global Change terms these "guard rails," WBGU 2009). An example of the use of environmental targets was the restriction of algae water consumption to no more than 5% of mean annual basin flow.
- Altering farmer or forester choices (in agent-based models) based on incentives, preferences, and established culture. For example, environmental effects of energy crops, such as improved water quality and wildlife habitat have been shown to influence some farmers' motivations for adopting perennial energy crops (Hipple and Duffy 2002). While these feedbacks from environmental effects to feedstock production could be used to constrain supply, such feedbacks were not implemented in *BT16* volume 2.

14.2.2 Mitigation Strategies

While this report was not intended to be prescriptive, some strategies were identified that may be used to enhance the environmental outcomes from biomass production. Strategies to mitigate effects of the *BT16* volume 2 scenarios were identified.

Mitigation strategies were based on environmental analyses that identified drivers of environmental effects in the scenarios. For example, the GHG analysis found that in some counties logistics contributed more than 50% to GHG emissions (excluding soil-carbon change-related emissions). Consumption of fertilizer and agricultural chemicals, as well as nitrous oxide emissions stemming from fertilizer use, were also significant contributors to GHG emissions. Therefore, the energy efficiency of logistics operations and fertilizer efficiency should be improved. Counties with higher yields generally experienced lower GHG emissions intensities. Therefore, increasing yields would be an effective mitigation strategy for GHG emissions. The analysis also found that crop-residue removal (e.g., corn stover or barley straw) can reduce soil carbon levels, but practices such as manure application and cover crop adoption could counteract soil carbon losses and therefore should be pursued as a mitigation strategy (Qin et al. 2015). Planting of deep-rooted species like miscanthus and biomass sorghum could contribute to soil carbon storage.

The agricultural water quality chapter (chapter 5) focused on conservation practices that could reduce loadings of pollutants to surface waters. Large improvements in water quality indicators, on a percentage basis, were achieved without sacrificing production. This was true for landscapes dominated by annual residues and landscapes dominated by a mixture of perennial and annual crops. Results for the Iowa River Basin suggested that four practices (riparian buffer, cover crop, slow-release nitrogen (N) fertilizer, and tile-drain control) could reduce N loading substantially for watersheds planted in corn. In the Arkansas White-Red River Basin, filter strips provided water quality benefits from short-rotation woody crops (SRWCs). Results from the water quality analysis can be used to identify location-specific management practices that can achieve water quality goals and biomass production goals simultaneously. In addition, by choosing perennial feedstocks and implementing conservation practices, biomass production could reduce downstream nutrient loadings to the Gulf of Mexico.

With respect to forests, silvicultural activities have minimal effects on water quality, and potential effects from harvest operations are largely mitigated by the widespread adoption of BMPs, as is discussed below. Furthermore, where forest removals could increase stormflow volume in local areas, forest BMPs such as implementing forest riparian buffers may be effective to mitigate negative harvesting effects on stream hydrodynamics.

The water footprint chapter noted that the National Resources Conservation Service Ogallala Aquifer Initiative aims to reduce water withdrawals and extend the life of the aquifer by implementing multiple conservation measures. One of the strategies is converting operations to dryland farming, which is defined as the non-irrigated cultivation of crops. This strategy is consistent with one of the guiding principles in *BT16*: produce non-irrigated biomass.

The air emissions chapter noted that variability in county-level emissions estimates suggests that certain practices and production locations would result in much lower emissions than others. Higher yields, lower tillage requirements, and lower fertilizer and chemical inputs contribute to lower air emissions intensity. The use of either more efficient equipment or fewer passes would reduce emissions from fuel use and fugitive dust from soil disturbance. The application of emission reduction strategies (e.g., higher yielding seed varieties, energy crops with high nutrient use efficiency, more efficient farm engines, and wider adoption of less intensive tillage practices) could mitigate the potential increase in emissions from *BT16* scenario activities. This analysis illustrates that the long-term feedstock supply logistics system itself could reduce emissions per mile traveled through feedstock densification. In addition, using biomass more locally or using more fuel-efficient long-distance transportation methods (e.g., rail) could potentially decrease emissions from long-distance truck transport.

The agricultural biodiversity chapter echoes suggestions that benefits to birds (and other wildlife) can be attained by implementing wildlife-friendly practices, e.g., timing farm operations prior to avoid nesting periods, using a flushing bar and raising the height of mowing equipment to avoid nests and animals during farm operations; and, simply harvesting from the inside of a field toward the edges, instead of trapping wildlife in the center of a field. Mitigation strategies to protect wildlife biodiversity in forests are more uncertain because of the lack of data relating biomass harvest variables to habitat suitability for various taxa. However, optimal mitigation strategies are expected to be site-specific, for example, protecting species that rely on moist forest floors in lowland hardwood forests or forest systems in temperate rainforests.

As discussed in the chapter on climate sensitivity to feedstock productivity, climate change adaptation is important. Adaptation can be aided by greater focus on the implications of climate change on the long-term strategic selection and production of energy crops across the U.S. landscape.

14.3 Enhancing Environmental Outcomes: Going Beyond Analyses in this Report

The context of land management is a major determinant of environmental effects. Regardless of whether land cover is classified as pasture or energy crop, management that incorporates native species, avoids disturbance during key nesting periods, and increases productivity while reducing the use of pesticides and herbicide applications is likely to improve many environmental indicators compared to management where disturbances are not planned to conserve species, or with higher use of inputs, or minimal control of grazing, or where exotic and invasive species are not controlled. Furthermore, the implications of management practices for additional biomass production in forestlands may result in better control of pests, fires, and invasive species with benefits that extend beyond the managed forest to neighboring parks and reserves (Dale et al. 2015). Thus, real impacts will depend on the prior conditions and actual management practices, which are highly heterogeneous, while impacts estimated through modeling will depend on the assumptions and specifications broadly applied to represent those conditions and management practices. Here, a number of approaches are described that are currently being used or are under development to enhance environmental outcomes for biomass production.

14.3.1 Best Management Practices

BMPs can improve environmental outcomes for realized biomass and future biomass. BMPs are approaches, processes, activities, incentives, or rewards deemed effective at delivering a more favorable outcome than other techniques when applied to particular circumstances. These recommended practices "transform knowledge about local conditions and practices into prescriptions for low-impact operations by specifying methods that reduce negative impacts" (Lattimore et al. 2010). Additional descriptors of BMPs are "useful," "proven," "cost-effective," and "generally accepted" (Texas State Soil and Water Conservation Board 2005). For example, forestry BMPs help to ensure that adequate woody debris remains on site to protect soil and water quality (Evans et al. 2013; Fritts et al. 2014; Cristan et al. 2016). BMPs are sometimes called "conservation" practices, especially in the context of agriculture, as they may be intended to conserve water quality, water quantity, air quality, or other objectives (NRCS 2016). Most BMPs are focused on water quality, and some definitions of BMPs refer exclusively to water quality impacts (Ice 2004). The most useful BMPs are quantitative, reflect targets for environmental indicators, and are associated with detailed advice regarding implementation. As an example BMP, winter cover crops like winter rye (which was modeled in chapter 5) can provide synergistic benefits of soil conservation, water quality, and biomass production with no increased demand for agricultural land (Feyereisen et al. 2013). Chapter 5 and additional studies have shown that the use of cover crops can reduce negative water-quality effects from farming operations (Graham et al. 2007; Mann et al. 2002), while decreasing soil erosion, maintaining land productivity (Kaspar et al. 2001; Snapp et al. 2005; Wyland et al. 1996), and reducing nutrient loadings.

A review of BMPs shows that they are commonly implemented in forestry (Ice et al. 2010), and some BMPs have are commonly employed in agriculture as well. However, additional BMPs could be developed to maintain or improve environmental indicators. Existing BMPs, which often emphasize soil quality and water quality, could be tailored for the purposes of biomass production and harvesting, and additional BMPs could be developed for air quality, biodiversity, and GHG emissions. Moreover, BMPs could be developed for algae biomass production. Adaptive management is an important framework for developing BMPs because it integrates research, planning, management, monitoring, and learning into evolving and improving practices (Lattimore et al. 2010; McAfee et al. 2006; Holling 1978). McAfee et al. (2006) note that the efficacy of recommended management practices in achieving sustainable operations can be limited if monitoring and assessment are not carried out within an adaptive management framework.

14.3.2 Landscape Design

Important improvements in environmental effects can be achieved by within-county spatial allocation of land management for biomass and other purposes, land management to mitigate potential adverse effects, and production area restrictions. The county-level resolution used in *BT16* does not enable environmentally favorable strategies at the field or sub-field scale to be modeled. As some of the chapters in this report illustrate through caveats and sensitivity analyses, county-level biomass estimates lead to substantial uncertainty in environmental indicators if the specific location of the biomass is not defined.

Landscape design principles offer a means to integrate biomass production with other uses of the land while meeting simultaneous environmental, social, and economic goals. A landscape design framework suggested by Dale et al. (2016) involves six steps: (1) establish goals by engaging stakeholders in an open and participatory process that, ideally, facilitates common understanding and agreement on context-specific targets for environmental or other indicators; (2) identify constraints and opportunities, such as impacts to water, soil, or air, as well as the enabling factors that assist in meeting stakeholder goals; (3) identify optimal options for feedstock types, locations, management strategies, and logistics systems; (4) evaluate alternatives and define solutions that are spatially and temporally explicit; (5) monitor and evaluate outcomes over time using mechanisms that are cost-effective, doable, and transparent; and (6) adjust plans based on current information for "continual improvement" and alignment with desired outcomes. By involving diverse stakeholders who are part of the bioenergy supply chain as well as those affected by its development, landscape design can help those involved define appropriate goals, understand tradeoffs, and achieve benefits that would not necessarily be attained through conventional land management approaches (Dale et al. 2016).

Field studies are underway to test landscape design approaches that leverage the ecosystem services provided by second-generation perennial lignocellulosic energy crops. Perennial crops such as SRWCs, switchgrass, miscanthus, and other perennial grasses share traits that differentiate them from annuals like corn and soybeans: a deeper root system, a generally better ability to thrive on less productive soils, a lower dependence on fertilizer inputs, and management options that can be more beneficial to wildlife. When deployed on the landscape in specific locations based on soil and land characteristics and their potential to perform specific functions, perennial bioenergy crops may provide water quality services and patchiness patterns that improve ecological habitats. By working with local producers and stakeholders, bioenergy landscapes can be designed that balance productivity and environmental performance.

A case study being conducted by Argonne National Laboratory (ANL) and centered in the Agricultural Midwest illustrates a promising opportunity to enhance aspects of the environmental outcomes from producing biomass for energy. Ongoing field research shows that a willow contour buffer on a sub-productive portion of a field can intercept nitrate from subsurface soil and provide important reductions in nitrate losses through plant uptake (fig. 14.1), confirming modeled results (Ssegane et al. 2015). To scale this concept up to a 50,000-acre tile-drained agricultural watershed in Illinois in the Mississippi River basin (as described in (Hamada et al. 2015), researchers are targeting production of bioenergy crops in "marginal agricultural subfield areas" identified using seven soil-based environmental criteria (susceptibility to nitrate and pesticide leaching, soil drainage, frequency of surface water ponding, frequency of flooding, soil erosion, and crop productivity).

Figure 14.1 | A field site in Fairbury, Illinois, is providing primary data on the effectiveness of a willow contour buffer in reusing the nitrate lost by the adjacent corn. In the picture, willow plots in the foreground and background are shown after corn harvest in their 2nd year of growth.

Using a calibrated SWAT model, ANL simulated the effects of growing switchgrass, willow, and big bluestem in these targeted subfield areas on annual yields of both energy crops and predominant corn and soybeans, nitrate-N and sediment exports, and water yields (Ssegane and Negri 2016). Results show that water quality benefits can be obtained by converting underproductive and environmentally marginal portions of fields to energy crops, with the production of biomass more than compensating for the loss in output of commodity crops from the landscape. The introduction of perennial energy crops using the same water-quality-focused watershed design may help create additional ecosystem services in terms of pollinator habitat, based on bioenergy crop type, landscape configuration, and energy crop area (fig. 14.2) (Graham, Nassauer, W. S. Currie, et al. 2016). ANL compared the cost of this practice per unit of N removed, including production and logistics costs to delivery at a depot, to other conservation practices (fig. 14.3) and found, for example, that a willow buffer would be close in cost to the adoption of practices such as wetlands or denitrifying bioreactors, and cheaper than a cover crop (Ssegane et al. 2016).

Finally, ANL calculated a comprehensive value for the water-quality based ecosystem services provided for potential future trading markets, including services derived from improvements to reservoir, navigation, recreation, irrigation, fisheries and other categories. The values obtained show the potential for supporting the production of perennial energy crops, should these markets be established. Through targeted workshops, conversations with farmer stakeholders jointly reviewed the proposed landscape designs and discussed solutions that advance societal goals while being feasible and acceptable by those who will implement them (Graham, Nassauer, and, et al. 2016).

Figure 14.2 | Modeling of the same watershed has shown that increasing the amount of land transitioning from corn/soybean rotations to perennial energy crops has the potential to increase pollinator nesting indices, with differences attributable to type of crop, area extent, and landscape configuration. The figure shows percent change in wild bee abundance when comparing current land use with two willow cropping scenarios: (a) 11% of the land in willow, and (b) 22% of the land in willow. (Graham, Nassauer, W. S. Currie, et al. 2016).

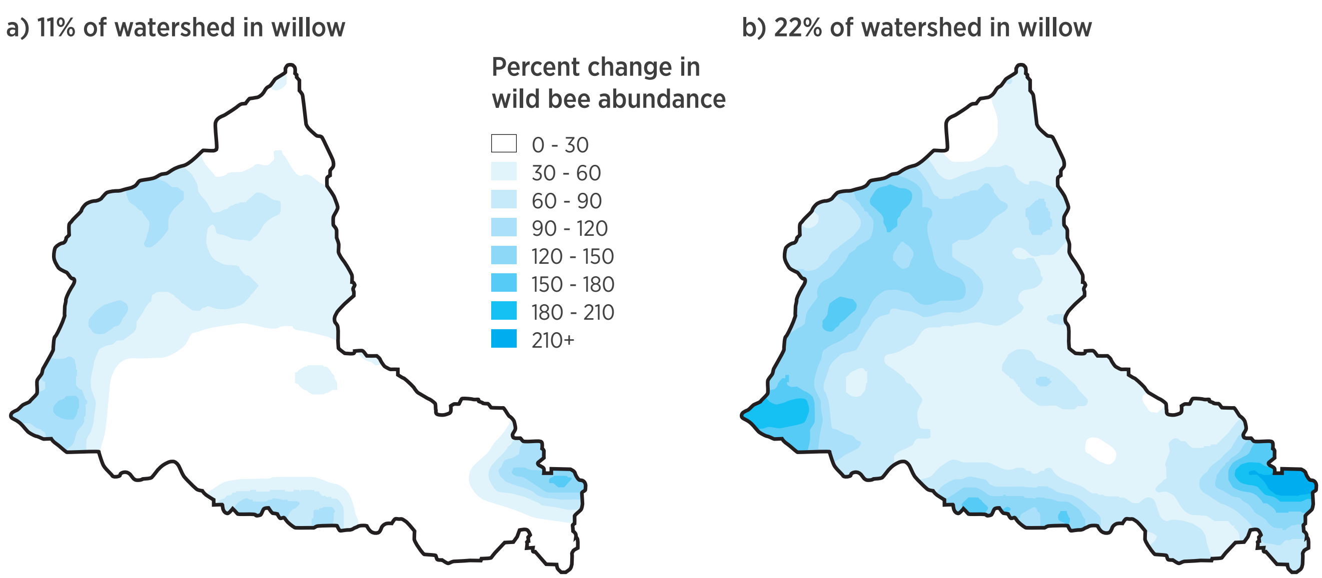
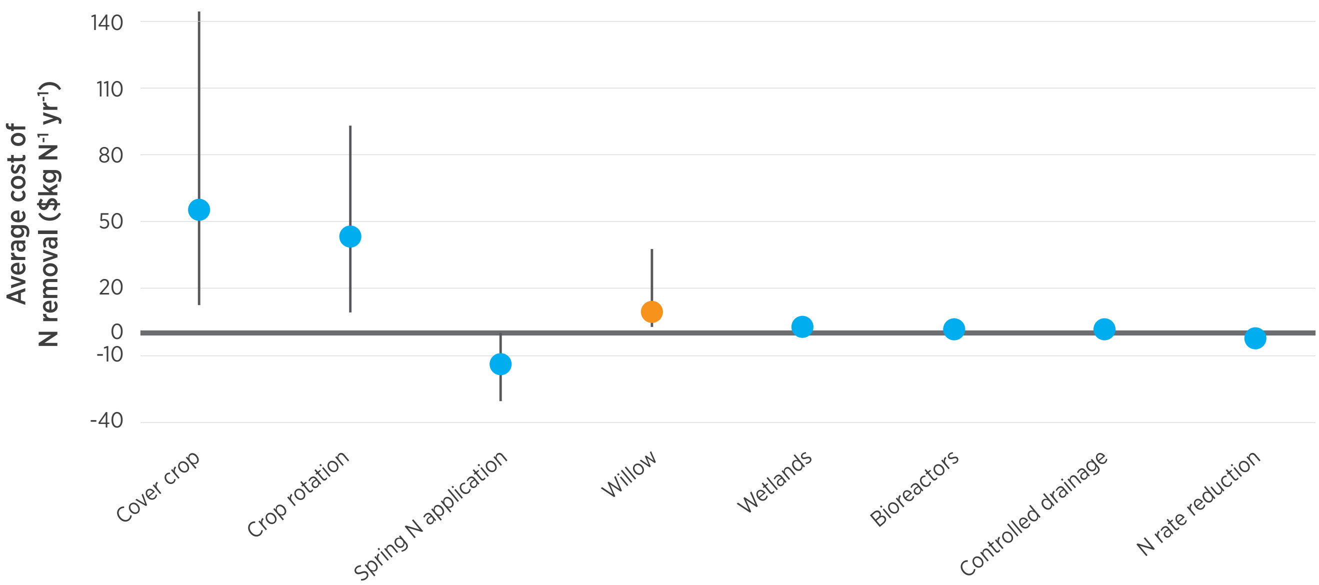


Figure 14.3 | Comparison between the calculated costs (per unit of nitrogen removed) of a willow buffer (orange dot) to intercept nitrate from a corn field and other conservation practices (blue dots). (Data from other conservation practices are from Christianson et al. 2013)



14.3.3 Precision Agriculture

Technological innovations and precision agriculture can also help enhance environmental outcomes (Muth et al. 2012). The biomass feedstocks from *BT16* volume 1 and those evaluated in volume 2 are quantified at the county level. However, sub-county and even subfield variability challenges the farmer's ability to sustainably produce and collect cellulosic biomass. *BT16* volume 1 includes assumptions regarding the operational availability of crop residues and how that availability may increase over time, given the potential of precision agronomics to enhance biomass availability in the future. Innovations in advanced logistics systems and precision agriculture enhance environmental outcomes by increasing biomass availability, practicality, and profitability while improving water quality through subfield stover removal decisions and associated variable harvesting technology.

Using the Landscape Environmental Assessment Framework (LEAF), a simulated supply shed (i.e., an area supplying feedstock to a biorefinery) in central Iowa was assessed for corn stover availability (fig. 14.4). To ensure that residues were being collected in a practical manner that also protected soil quality, water quality, and profitability, the analysis assumed that the entirety of individual fields must be managed to permit residue collection that meets environmental objectives. In other words, if a portion of a field could not support residue collection that met soil quality and water quality targets, the entire field was ineligible and contributed no biomass to the supply area total. This constraint results in a significant reduction in biomass availability compared with the future potential biomass supplies estimated an earlier Billion-Ton report (U.S. Department of Energy 2011), but it fairly represents the challenges and limitations of conventional field-level residue management (fig. 14.6). This ongoing research is described in Bonner, Cafferty, et al. (2014) and Bonner et al. (2016).

If the full potential of a billion-ton bioeconomy is to be realized, alternative management practices must be implemented to overcome the challenges of practicality and maintaining soil and water quality targets.

Figure 14.4 | Modeled feedstock supply shed relative to two pioneer lignocellulosic facilities

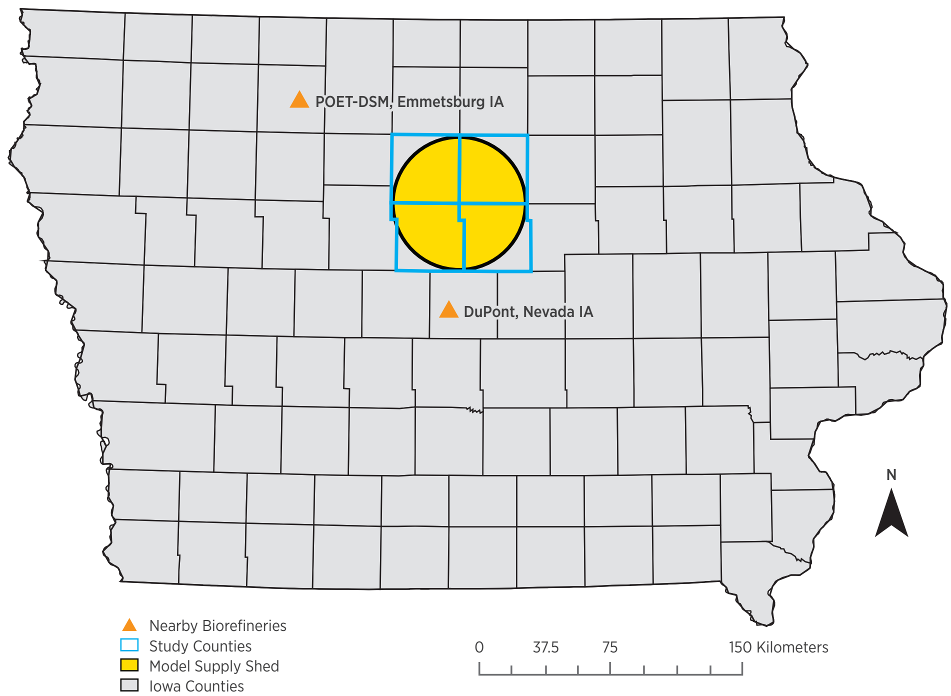
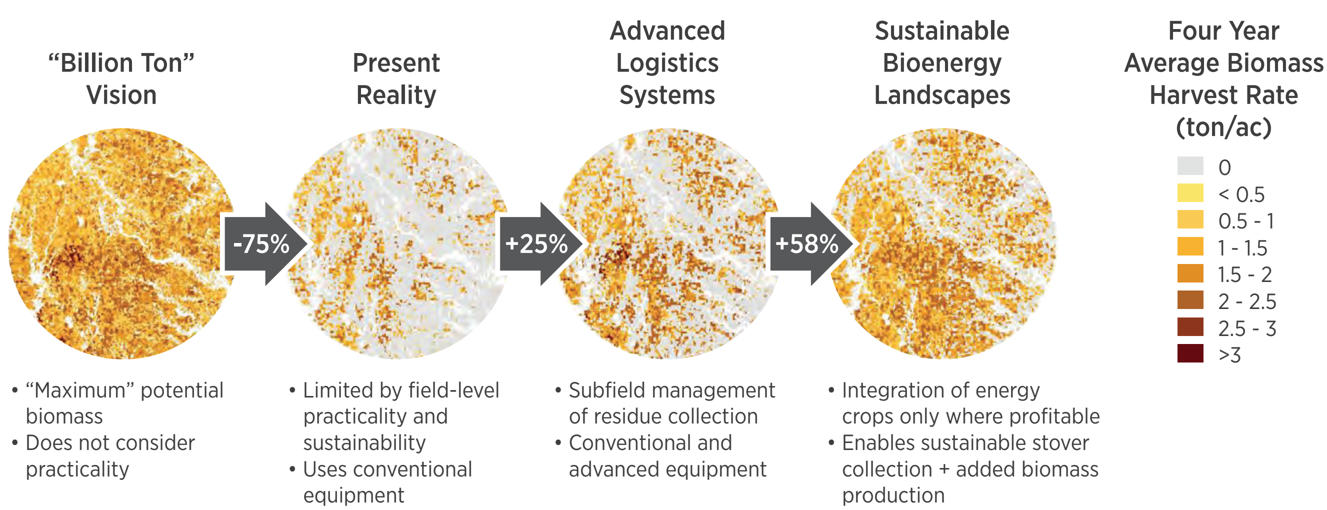


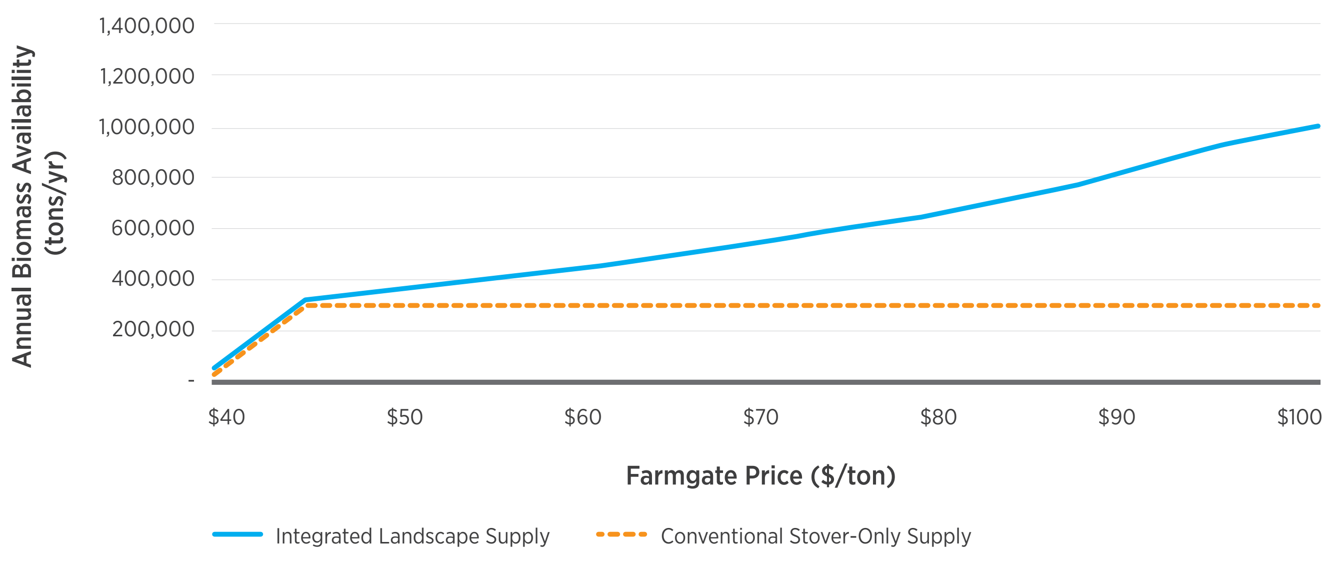
Figure 14.5 | Depiction of the reduction in biomass availability when practicality constraints are applied at the field level, and how biomass resources are mobilized and increased as a result of advanced logistics and conservation of soil carbon. Sustainability refers to soil quality and water quality.



Advanced logistics systems offer great potential, with one such alternative being a simple "binary" harvest where precision agronomics are used to avoid residue collection on sensitive portions of fields. Precision management plans constructed at the subfield level can then be used in conjunction with conventional equipment and Global Positioning System (GPS) guidance technology to direct harvesting equipment operation to where residue collection is permitted. This concept may be further expanded into variable-rate collection techniques whereby specialized equipment is used to apply real-time calculation of residue removal constraints during grain harvest and residue management (Karkee et al. 2012; Muth and Bryden 2012). For the case study supply shed modeled here, such advancements in logistics would permit nearly half of the fields to participate in sustainable residue collection, so that 50% of the available biomass that meets soil and water quality targets becomes practically available (fig. 14.5).

Although these results show a promising alternative for achieving greater biomass yields while maintaining soil quality, further improvements in land management will be required if all biomass from subfields that meet the soil and water quality targets is to be accessed in a practical manner. Such methods could be simple alterations of existing practices, such as reducing tillage intensity, or adoption of conservation practices like cover crops or vegetative barriers (Bonner, Muth, et al. 2014). Alternatively, the incorporation of dedicated bioenergy crops into the supply shed presents a valuable opportunity to increase biomass resources, sustainability, and profitability for growers. By better utilizing under-performing portions of row-crop-producing fields, energy crops can be introduced in a manner that is cost-competitive and beneficial for the environment (Bonner, Cafferty, et al. 2014; Bonner et al. 2016). For example, the integration of switchgrass into subfield locations within the modeled supply shed can be done in such a manner that field-level revenue is increased, additional biomass is produced, and the collection of corn stover is enabled on over 90% of fields (fig. 14.6). Through the combination of advanced logistics and subfield precision agriculture, a pathway to achieving the agricultural residues and biomass crop supplies discussed in *BT16* volume 1, while maintaining or improving environmental outcomes, becomes tangible.

Figure 14.6 | Comparison of feedstock availability between a conventional corn stover system limited by practicality and an integrated landscape in which switchgrass is used to increase biomass production and enable additional corn stover collection



14.3.4 Multipurpose Biomass Production and Removal

Strategies to produce and use biomass resources in ways that enhance environmental outcomes and provide multiple environmental benefits are being evaluated. These strategies include waste or "opportunistic resources" that, if used, provide benefits beyond biomass products. For example, biomass production can offer environmental benefits through phytoremediation, mineland reclamation, and wastewater remediation. Bioenergy can be a coproduct in many of these applications. Another example is the production of algae using waste CO_2 in flue gas that would otherwise be emitted directly to air.

Waste biomass is a category of biomass that is estimated in *BT16* volume 1, and potential benefits of waste use are discussed here. The multiple benefits to utilizing waste products for energy vary depending on the waste resource and include: displacing fossil fuels (thus reducing GHG emissions and reducing imports), reducing demand on disposal facilities (e.g., landfills, waste treatment facilities), odor control (from manure), protection of water quality, improved air quality (reduction in field burning of residues, reduced emissions from raw manure), conservation of natural resources by producing useful products from wastes, and a reduction in forest fire risk (from thinnings and use of standing dead wood).

Waste biomass is the most diverse category of feedstocks estimated in *BT16* volume 1. That volume includes twenty-four waste resources: agriculture (cotton gin trash, cotton field residues, grain dust and chaff, orchard and vineyard prunings, rice hulls, rice straw, sugar cane bagasse, sugar cane trash, soybean hulls, animal fats, yellow grease, animal manure, and the garbage fraction of municipal solid wastes (MSW)), forestry (other residue removals, treatment thinnings from other forestland, unused primary and secondary mill residues, urban wood wastes – construction and demolition, and urban wood wastes from MSW), and other resources (biosolids, brown trap grease, food wastes (industrial, institutional, and commercial), landfill gas, and utility tree trimmings). In the aggregate, waste resources in 2040 total 155 million dry ton at $60 per dry ton and 229 billion ft3 of additional landfill gas.

The use of waste resources for energy represents a substantial opportunity if the economics are favorable. The three largest categories of waste resources are the garbage fraction of MSW (i.e., paper and paperboard, plastics, rubber and leather, textiles, food waste, yard trimmings, and other, but excluding wood wastes) (55 million dry tons in 2040 at $60 per dry ton in *BT16* volume 1), animal manures (18 million dry tons in 2040 at $40 per dry ton, about half of agricultural waste resources), and construction and demolition (C&D) wastes (25 million dry tons in 2040 at $60 per dry ton). Major issues with MSW include (1) finding landfill space for disposal and (2) methane emissions from landfills. (If methane is captured and burned in a controlled situation it produces CO_2 and water, while methane has approximately 21 times the greenhouse warming potential of CO_2. A reduction in GHG emissions is a major benefit of capturing and combusting methane.) Utilization of MSW for energy purposes reduces the need for landfill space and capturing methane generated by existing landfills reduces GHG emissions and displaces other forms of energy.

The utilization of manures via anaerobic digestion to produce biogas has a number of environmental benefits that include protecting water quality by destroying potentially pathogenic bacteria and reducing biological oxygen demand, which can improve water quality and protect aquatic biodiversity; and reducing GHGs from methane produced from manure by capturing and utilizing the methane. In addition to these environmental benefits, the utilization of captured methane also displaces fossil fuels. EPA estimates that for dairy and swine (hog) farms with more than 500 and 2000 head, respectively, anaerobic digester systems to capture biogas could be economically feasible (U.S. Environmental Protection Agency 2011).

Another example of opportunistic resources is residues from forest thinnings, for which harvesting may reduce fire risk. At the time of this writing, forest fires are inflicting as-yet uncalculated damage in the Great Smoky Mountains National Park and nearby Gatlinburg, Tennessee, as well as other areas in the southeastern and western United States. Wildfires cost lives and over a billion dollars annually in the United States (Mosley et al. 2013). Biomass harvests in the wildland-urban interface, though likely not the cheapest source of biomass, can provide critical value in the form of fuel load removal and wildfire risk reduction (Staudhammer et al. 2011). These fuel load reduction treatments can provide biomass beyond the supplies reported in *BT16* volume 1.

Similarly, biomass removals for control of invasive species can provide win-win biomass use benefits. Ecosystem restoration efforts may benefit from the removal of kudzu, melaleuca, cogongrass, leuceana, castor bean, and other species. Powerline right-of-ways and other areas that require maintenance can also be used to produce biomass while providing co-benefits of vegetation-control.

Mineland reclamation and phytoremediation represent synergistic opportunities to produce biomass while providing other environmental benefits. These strategies may be preferred to conventional remediation technologies, which can be expensive and environmentally harsh. Through its "Re-Powering America's land" initiative, the USEPA encourages the development of renewable energy on potentially contaminated land, landfills and mine sites (U.S. Environmental Protection Agency 2016).

Phytoremediation uses green plants to remove contaminants from soil or water (Negri and Hinchman 1996) presenting dual-purpose opportunities for phytoremediation and biomass production (e.g., Rockwood et al. 2004). Phytoremediation can involve extraction of contaminants, stimulation of biological degradation, and sequestration in situ through the establishment of a functional ground cover. Phytoremediation has been proposed as a viable alternative to costlier remediation solutions in cases of large expanses of land contaminated with low levels of pollutants, or as a "gentle" remediation technique with more favorable lifecycle environmental impacts where harsher interventions would compromise other important ecological functions. In these cases, the potential to defray costs through the production of biomass is considered an attractive opportunity. A number of the same crops that are proposed for bioenergy have been used in phytoremediation, which typically share required traits of fast growth, deep root systems and the ability to grow in suboptimal conditions. In the US, 1,200 contaminated sites are listed in the National Priority List for remediation (U.S. Environmental Protection Agency 2011), with an estimated 2,600,000 hectares contaminated with trace elements alone.

Sites such as mine land, landfills and brownfields could be used to produce biomass while under a long-term reclamation/remediation process. Biomass production for mineland reclamation has been evaluated for applications in the Appalachian coal mines (Burger 2011; Akala and Lal 2000), phosphate and titanium mined lands in Florida, (Brown et al. 1992; Segal et al. 2001; Tamang et al. 2005; Rockwood et al. 2006; Langholtz et al. 2007; Proctor 2002), and elsewhere. Production of biomass from low opportunity cost lands can provide multiple environmental benefits while being publicly favorable and contributing to regional mine land reclamation goals.

Algae can be co-located with CO_2 or other waste nutrients. Co-location of algae biomass production facilities with CO_2 to produce energy or food (see chapter 7 in *BT16* volume 1) is a beneficial use of waste. Wastewater has nutrients that can be used by algae or, if reclaimed, taken up by irrigated crops. Algae biomass production can be co-located with wastewater treatment facilities that provide nutrients. The relative economic benefits of treating wastewater and producing algal biofuel as a coproduct versus producing algal biofuel as the main product and treating wastewater as a coproduct are discussed in Lundquist et al. (2010). Fast-growing terrestrial feedstocks can also absorb nutrients from reclaimed wastewater, providing a tertiary treatment while producing biomass ((Alker et al. 2002; Langholtz et al. 2005).

The benefits of phytoremediation, mineland reclamation, and wastewater biomass production strategies should be considered within the context of environmental economics.

14.3.5 Monetary Strategies

Environmental economics evaluates the value of environmental effects, both positive and negative, and potential solutions to reduce market failure[1] (e.g., Iftekhar et al. 2016; Hanley and JF White 2002). One example of the application of environmental economics is emissions-trading amendments to the Clean Air Act of 1990 (Clean Air Act Amendments of 1990), which reduced SO_2 emissions and acid rain. As suggested in this volume and from other researchers (e.g., Werling et al. 2014), the production of perennial native grasses can enhance benefits of a range of ecosystem services. A biofuels industry that creates a market for these feedstocks may increase the provision of positive externalities[2]. Environmental economics can use markets to reduce environmental costs or improve environmental benefits associated with increased biomass production and use. Such an approach could foster environmental efficiency of an expanded bioeconomy.

Ecosystem services offer a useful framework from which to consider associated trade-offs among effects of biofuel production and use (Gasparatos et al. 2013). Developing agreement on values and indicators among stakeholders is a prerequisite to building community and policy support for programs that enhance ecosystem services. Ongoing modeling and field projects are evaluating how biomass production can provide and improve ecosystem services such as soil quality, water quality, and wildlife habitat (Ssegane et al. 2016; Dale et al. n.d.).

14.4 Looking Forward and Future Research Needs

Research, science-based monitoring, and adaptive management can be used to further enhance environmental benefits of biomass production while mitigating potential negative effects. *BT16* volume 2 has identified potential environmental considerations that are relevant and important as biomass production industries develop in the United States. Analyses of environmental effects for the scenarios considered in this volume can help the research community, industry, and other decision makers prioritize research efforts and data collection, as well as move toward identification of priority locations for biomass production and location-specific BMPs.

14.4.1 Summary of Key Research Needs Identified in *BT16* Volume 2

Research gaps and needs are identified in the chapters of *BT16* volume 2, ranging from local monitoring of environmental indicators to national modeling studies and global indirect LUC (ILUC). Some of the research recommendations relate to how biofuels (or biomass in this case) can be "done right" (Kline et al. 2009) to improve environmental effects; other research relates to how the modeling of biomass can be improved with respect to accuracy and precision (e.g., through improved data collection and broader validation of models). Implications of environmental effects measured in *BT16* volume 2 (e.g., effects of

[1] Market failure is a situation where markets are not efficient, i.e., a different market situation could improve benefits to society as a whole without negative impacts to market participants.

[2] Externalities are unintended impacts, positive or negative, of a commercial activity that affect stakeholders not involved in the economic transaction.

changes in stream flows on fish, effects of air pollutant emissions on local air quality) are also recommended. Additional research that could follow this study is described below.

The establishment of consistent definitions for land cover and land management are required to support a consistent analysis of change over time. Consistent and transparent use of terms and definitions for land cover classes, crop types and rotations, and characterization of land management are essential elements for improved LUC analysis. Some of the research needs related to GHG emissions include exploring the sensitivity of SOC changes to model assumptions, including the treatment of tillage and effects of rotation, crop yield, land-use history, and land transition matrices. Techniques could be explored to mitigate factors that lead to hotspots of SOC change. The relative contribution of aboveground carbon changes is an additional research gap. Temporal emissions accounting could be added to the treatment of forest-derived feedstocks in GHG modeling.

Research is needed to model biomass removal at finer spatial scales, such as within a watershed rather than a county, which is too coarse for the assessment of some water yield effects of forest biomass production. Future studies should examine the cumulative effects of forest biomass removal in specific watersheds where harvesting activities are expected to occur, and should focus on ecologically relevant indicators of streamflow. In addition, future studies should link water quantity and quality to allow for a comprehensive assessment of water resources at watershed-to-county levels.

The context of environmental effects may require regionally specific monitoring. For example, while current forestry BMPs are likely adequate to maintain stream water quality for intensive pine silviculture in the Southeastern Coastal Plain, dominant groundwater flow paths suggest that groundwater quality and transit times should be monitored and evaluated (pers. comm. Natalie Griffiths to Matthew Langholtz, December 2016).

Further research is needed on fugitive dust emissions from forestry management activities and biogenic emissions from agricultural and whole-tree biomass feedstocks. The emission estimates provided in this study could be coupled with air-quality screening tools to evaluate potential changes in emissions concentrations, to assess potential human health impacts, and to develop sustainability constraints (i.e., excluded lands) for future scenarios related to biomass production.

Regarding vertebrate biodiversity in agricultural systems, research is required to (1) measure and model responses of additional combinations of wildlife taxa andnonnative feedstocks such as miscanthus; (2) increase the feasibility of production systems that employ more diverse communities of plants as feedstocks; (3) understand logistic, social, and economic barriers that could prevent farmers from adopting practices that benefit wildlife; (4) quantify relative effects of pesticide use for bioenergy feedstocks and for other managed lands; and (5) identify geographic hotspots where attention to wildlife-friendly practices is needed.

To further study the effects of forest biomass harvests on biodiversity, more manipulative studies need to be conducted (1) that vary amounts of coarse and fine woody debris retained across gradients in forest cover and forest types and (2) that measure the response of multiple species across trophic levels. Manipulative studies can also help determine whether responses are due to the forest-harvest treatment itself or the additive effect of removing dead and downed wood. Also, established studies should continue over longer time periods so that the effects of removing coarse woody debris and fine woody debris during second- and third-harvest rotations can be better understood. General relationships observed in this volume should be viewed as the basis for establishing testable hypotheses regarding biodiversity response to biomass harvest.

Research needs for algae production include quantifying the environmental effects that are only described in qualitative terms in this report. Quantitative estimates of the GHG emissions of biomass production alone are not possible for an algal biomass system that is highly integrated, so a life-cycle analysis would need to evaluate the whole supply chain for co-location of production facilities with various sources of CO_2. Water consumption must be understood in the context of regional competitive use. In addition, research is needed to evaluate potential biodiversity, air quality, water quality, and primary productivity effects of growing diverse species of algae at the commercial scale. As algae-produced food (protein) and feed become commercially viable, understanding the interactions between the profitability, food security, energy security, and water quantity will become paramount.

To advance climate change adaptation, research is needed on the development and genetic improvement of energy crops for climate-related stress; management practices to reflect climate change implications for plant establishment, maturation, and harvesting; and the implications of shifting energy crop yields and economic competitiveness for the biomass supply chain, including transportation and refining. The development of a more process-based understanding of biomass feedstock responses to changing climatic conditions that includes factors such as climate variability and extremes, the effects of CO_2 fertilization, and different management practices and economic constraints would assist in reducing uncertainties associated with purely empirical methods.

An additional research need is to model watersheds with multiple land uses so that silviculture, agriculture, urban, and other land uses can all be integrated in models of cumulative effects while assessing their individual effects as well. For example, long-term watershed-scale research should continue to measure the effects of traditional and emerging silvicultural practices on water quality. Moreover, tradeoffs could be studied between the environmental effects associated with increased residential development and those associated with the biomass harvesting that could generate income to slow development.

Determining the drivers and effects of land-cover and land-management changes attributable to biomass production—or to any specific intervention—requires monitoring both the effects over time and the human behaviors driving those effects. Models are not useful without monitoring to provide parameters or a measure of their validity. Most models used in *BT16* volume 2 are validated or verified under many conditions, and models that were created for this study (biodiversity and forest water quality models) are developed from empirical data. Yet, none of the model results have been validated with commercial-scale data for biomass across all of the regions where the models are employed. As data from operational systems become available, this validation will be critical for reducing uncertainties and increasing accuracy of modeled results.

14.4.2 Integrated Consideration of Environmental Indicators

BT16 volume 2 is a collection of analyses that consider categories of indicators independently. To help decision makers consider a suite of environmental effects in a region, tradeoffs among indicators, as well as aggregation functions, could be investigated. The joint consideration of environmental indicators could reveal locations of potential concern among indicators. The GHG, water quality, and biodiversity analyses, for example, show locations where biomass production could lead to environmental indicators that are less favorable than particular reference conditions. Further analyses with uniform assumptions would be needed to examine the analyses together.

Similarly, the integrated consideration of indicators could reveal tradeoffs. The water quality analysis for agriculture (chapter 5) was an initial step toward investigating tradeoffs among indicators, i.e., water quality and productivity indicators. This analysis found complementarities between increasing biomass yield and reducing total suspended sediment and total phosphorus, and tradeoffs between biomass yield and nitrate for perennial grasses and SRWCs. In addition, the analysis revealed water-quality benefits of coppiced willow, which minimized trade-offs between nutrient and sediment reduction and biomass yield in the scenario. Biodiversity studies (chapters 10 and 11) revealed potential tradeoffs among species, i.e., benefits of land transitions and biomass harvesting for some species (e.g., forest species that prefer young forests and grassland birds such as ring-necked pheasant) and decreases in range for some grassland species and potential reductions in species that require moist forest floors. Additional integration of indicator analyses with evaluation of a broad range of tradeoffs is needed.

Large quantities of data about diverse aspects of environmental (as well as social and economic) sustainability are difficult to visualize without some sort of reduction in dimensionality (Pollesch and Dale 2015). Aggregation functions are used to simplify data and clarify communication. Aggregation theory is an area of mathematics that explores the form and properties of such aggregation functions. In their book, Aggregation Functions, Grabisch et al. (2009) present a comprehensive mathematical treatment of aggregation functions that Pollesch and Dale (2015, 2016) have adopted for bioenergy assessment. Pollesch and Dale (2016) use methods that allow for inclusion of context-specific baselines and target values.

Parish et al. (2016) developed one example of aggregation applied to switchgrass in east Tennessee. A suite of 35 environmental and socioeconomic indicators in 12 categories was considered in a holistic assessment of a 5-year switchgrass-to-ethanol production experiment centered on a demonstration-scale biorefinery in Vonore, Tennessee. Three alternative scenarios were compared within a qualitative sustainability evaluation framework built for the case study using freely available DEXi 4.0 software that was designed to solve complex decision problems that involve 15 or more attributes, inaccurate and/or missing data, group decision-making, and expert judgment (Bohanec et al. 2013). Within this east Tennessee context, switchgrass production can improve environmental and social trajectories without adverse economic impacts, which can lead to enhanced sustainability overall (Parish et al. 2016). Future research could apply aggregation theory to biomass production in other contexts.

14.4.3 Integration across Environmental, Social, and Economic Effects

BT16 volume 2 focuses on environmental effects, but it is important that future studies investigate environmental, social, and economic effects in a more integrated manner to provide a broader view of sustainability of expanding biomass production in the United States. Integrating environmental, social, and economic analyses should give a geographic picture of locations and regions that could benefit most from biomass production and those which might experience adverse effects or tradeoffs among effects.

Socioeconomic indicators have been proposed to measure and model sustainability of bioenergy systems (Dale et al. 2013; Efroymson et al. 2016). These indicators represent social well-being, energy security, external trade, profitability, resource conservation, and social acceptability. Social and economic effects of biomass production have been investigated elsewhere and suggest a range of potential benefits. For example, a substantial increase in rural jobs has been associated with biomass production over the past decade (Golden et al. 2015; Golden et al. 2016), and one would expect this to continue with an expanding biomass industry. Agricultural systems designed to integrate energy crops are more diversified and resilient, factors that improve market stability and food security (Kline et al. 2016), as well as economic stability in communities. An important aspect of enhancing sustainability is building markets that provide economic incentives for sustainable land-use practices. Markets that improve the economic viability of working forests can help keep forests as forests and mitigate conversion of forestland to residential and commercial development.

Despite these potential social and economic benefits, more research is needed (1) to quantify and validate these effects as biomass production expands and (2) to evaluate how growth in the bioeconomy sector causes beneficial or adverse effects to other sectors. Future research on the application of aggregation applied to bioenergy, as discussed in section 14.4.2, would assist in quantifying these complex relationships between environmental, social, and economic effects. Visualization tools would help researchers and decision makers understand the relationships between multi-dimensional effects.

Integrating social and economic research with the environmental analyses in *BT16* volume 2 could lead to modifications of economic assumptions in volume 1. Environmental effects of energy crops, such as improved water quality and wildlife habitat, have been shown to influence some farmers' motivations for adopting perennial energy crops (Hipple and Duffy 2002). Caldas et al. (2014) note many economic and cultural factors that may affect Kansas farmers' willingness to grow cellulosic energy crops or to harvest residues, and Song et al. (2011) have found that farmers need large incentives to compensate for risk and potential reversibility of transitioning to perennial energy crops.

14.4.4 Concluding Thoughts

Integrating resource analysis and sustainability concepts should continue to be a broad goal for future research on potential biomass supply in the United States. *BT16* volume 2 is a first effort to consider potential biomass supply and environmental effects in a more integrated manner. This study can assist stakeholders in identifying beneficial biomass production opportunities while considering their local conditions and specific environmental goals. For example, the DOE Bioenergy Knowledge Discovery Framework (www.bioenergykdf.net) provides data sets from both *BT16* volume 1 and volume 2 as well as interactive tools that can be used to investigate relationships between biomass production and environmental effects and explore how different assumptions can influence outcomes. Furthermore, *BT16* volume 2 provides an extensive resource for informing future research and development efforts to enhance environmental benefits and mitigate negative effects associated with a growing bioeconomy.

As identified in the *BT16* volume 1, a wide range of feedstocks and suitable lands are potentially available to realize a future bioeconomy vision. BT16 volume 2 begins to examine the factors that are needed to make that vision more environmentally sustainable. As with existing agricultural and forest production, environmental outcomes of biomass production are contingent on local decisions and practices. This report suggests that with continued diligence and innovation, biomass can be produced and harvested in ways that avoid or mitigate adverse environmental effects while providing tangible environmental benefits.

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Glossary of Key Terms

Glossary of Key Terms

Agricultural baseline – Building on both the USDA 2015 baseline and the agricultural census data (USDA NASS 2014), this baseline scenario was developed using POLYSYS (see chapter 2 and volume 1, appendix C). This is the reference case for the *BT16* volume 1 agricultural scenarios. The agricultural baseline runs from 2015 through 2040.

Agricultural residues – Aboveground biomass produced as byproducts of conventional crops. Crop residues modeled in *BT16* volume 1 include: barley straw, corn stover, oat straw, sorghum stubble, wheat straw.

Agriculture scenarios – Also called exogenous price simulations or "specified -price" simulations - For the purpose of *BT16* volume 2, these include a base-case scenario with a 1% annual yield increase for energy crops, (referred to as "BC1") and a high-yield scenario with 3% annual yield increase for energy crops (referred to as "HH3"), among other scenario-specific assumptions (see volume 1 chapter 4 and appendix C, as well as volume 2 chapter 2).

Algae co-location scenarios – For the purposes of *BT16* volume 2, these scenarios include open-pond algal biomass production that may be associated with select resource co-location opportunities to utilize carbon dioxide (CO_2) from ethanol plants, coal power plants, and natural gas plants. Biomass, and price ranges for that biomass, are estimated for *Chlorella sorokiniana* (a freshwater strain) and *Nannochloropsis salina* (a saline strain).

ANPP – aboveground net primary productivity – Linked to the energy flow in ecosystems and important for global carbon estimates, ANPP is sometimes referred to as plant yield and is the rate of storage of organic matter in plant tissues in excess of the respiratory utilization by plants. (Definition adapted from Odum E., 1971. Fundamentals of Ecology, Saunders: Philadelphia).

Biodiversity – biological diversity – Variability among living organisms from all sources and the ecological complexes of which they are part; this includes diversity within species, between species and of an ecosystem. (Definition adapted from the United Nations Convention on Biological Diversity as cited in ISO, 2015. 13065:2015 - Sustainability Criteria for Bioenergy).

Bioeconomy – From a broad economic perspective, the bioeconomy refers to the set of economic activities relating to the invention, development, production and use of biological products and processes. (Definition adapted from OECD, 2016. The Bioeconomy to 2030: designing a policy agenda).

Bioeconomy AGE – Bioeconomy Air and Greenhouse Gas Emissions – This model estimates the energy, air quality, and GHG impacts of the Billion-Ton Bioeconomy cases compared with an all fossil baseline.

Bioenergy – Energy derived from biomass.

Biofuels – Fuels made from biomass resources, or their processing and conversion derivatives. Biofuels include ethanol, biodiesel, and methanol among others.

Bioindicator – Species or group of species whose function, population, or other characteristics can reveal environmental conditions such as pollutants.

Biopower – The use of biomass feedstock to produce electric power or heat through direct combustion of the feedstock, through gasification and then combustion of the resultant gas, or through other thermal conversion processes. Power is generated with engines, turbines, fuel cells, or other equipment.

Biomass sorghum – An annual herbaceous crop, currently grown in rotation throughout the Southeast and Great Plains for grains and forage. Biomass sorghum exhibits non-photoperiod sensitivity and drought tolerance. For the purposes of *BT16* analyses, this term depicts any variety of sorghum developed for high biomass yields, and neither for grain nor sugar content. Budgets for biomass sorghum under *BT16* volume 1 can represent biomass sorghum, forage sorghum, or sweet sorghum. Modeled yields represent either biomass or forage sorghum; the variety with the highest productivity in a certain region was used in the agriculture scenarios.

Biomass supply scenarios – For the purpose of *BT16* volume 2 analyses, this term denotes an empirically modeled scenario with combined agriculture forestry resources.

BMP – Best Management Practice –The practice, or combination of practices, that is determined to be an effective and practicable (including technological, economic, and institutional considerations) means of achieving a given goal (Definition adapted from North Carolina Forest Service, Best Management Practices: What are BMPs). Often the goal associated with BMPs is conservation of resources, and so this term is often used within the context of environmental management.

BTB – Billion-Ton Bioeconomy – The evaluation of potential benefits of using biomass to produce fuel, power, and chemicals as compared to fossil-derived feedstocks.

Bulk density – An indicator of soil compaction, bulk density describes the weight of a material (e.g., soil) divided by its volume. Grams per cubic centimeter are generally used as units of measurement. (Definition adapted from the U.S. Department of Agriculture's Natural Resources Conservation Service).

CDL – Cropland Data Layers – A product of the United States Department of Agriculture, National Agricultural Statistics Service, Research and Development Division, Geospatial Information Branch, Spatial Analysis Research Section with the scope to "use satellite imagery to provide acreage estimates to the Agricultural Statistics Board for the state's major commodities and to produce digital, crop-specific, categorized geo-referenced output products." (Definition from USDA NASS, 2016. FAQ CropScape and Cropland Data Layers).

Clear-cut /clearcut – General term used to denote a type of forest harvest in which every tree has been cut down during a logging operation, but which in practice means that a few trees of non-commercial value are left standing.

CLU – Common Land Unit – The smallest unit of land that has a permanent, contiguous boundary, a common land cover and land management, a common owner and a common producer in agricultural land associated with USDA farm programs. CLU boundaries are delineated from relatively permanent features such as fence lines, roads, and/or waterways. (Definition adapted from USDA Farm Service Agency, Aerial Photography, Imagery Products).

CO_2 – carbon dioxide – A colorless, odorless noncombustible gas with the formula CO2 that is present in the atmosphere. It is formed by the combustion of carbon and carbon compounds (such as fossil fuels and biomass), by respiration, which is a slow combustion in animals and plants, and by the gradual oxidation of organic matter in the soil. (Definition adapted from DOE EERE, 2016. Glossary of Energy-Related Terms).

CO_2 equivalent emissions (CO_2 e) – CO_2 and nitrous oxide [N_2O].

CO_2 fertilization – This is theoretical principle associated with climate change science describing the fertilization of plants that use CO_2 in photosynthesis from increased CO_2 in the atmosphere.

CO – carbon monoxide – A toxic, colorless, odorless, and tasteless gas, CO is produced in the incomplete combustion of carbon and carbon compounds such as fossil fuels (i.e. coal, petroleum) and their products (e.g. liquefied petroleum gas, gasoline), and biomass. (Definition adapted from DOE EERE, 2016. Glossary of Energy-Related Terms).

Conservation – To reduce or avoid the consumption of a resource or commodity. (Definition adapted from DOE EERE, 2016. Glossary of Energy-Related Terms).

Consumptive water use – A key parameter for water depletion (see ISO, 2015. 13065:2015 - Sustainability Criteria for Bioenergy), this term refers to the use of water for a process (e.g., industrial or agricultural irrigation).

Conventional crop – Under the *BT16* analyses, conventional crops are known as the primary U.S. commodity crops, such as barley, corn, cotton, hay, oats, rice, sorghum, soybeans, and wheat are considered conventional crops.

Conventional tillage – System using soil tillage to turn the soil (e.g., with a chisel plow and an offset disk) and prepare a field for planting. This “traditional” practice helps to control for weeds and pests and traditionally involves burying crop residues.

Conventional wood – Roundwood, whole-tree chips, or wood residues that are used for the production of wood pulp (also referred to as pulpwood), or dimension lumber, or construction products.

Conversion – A fundamental change in form, character, or function. Can refer to chemical conversions (e.g., biomass feedstocks to biofuel and bio-based chemical products), or land conversion (e.g., the process of first-time conversion of a high-canopy forest to a human-managed landscape for agriculture and settlement).

CORRIM – Consortium for Research on Renewable Industrial Materials – This organization provides data for LCA analyses with a focus on the “environmental impact of the production, use, and disposal of wood and other bio-based materials.” CORRIM regions include Northeast, North Central, South, Inland West, and Pacific Northwest. (Definition adapted from CORRIM, 2016. Our Mission. Accessed from http://www.corrim.org/).

Cropland – Similar to the 2012 USDA Census of Agriculture definition of “total cropland,” this land category includes planted and harvested acres of corn, wheat, grain sorghum, barley, soybeans, rice, cotton, and hay (see Natural Resources Conservation Service definition of cropland and *BT16* volume 1 appendix C for more details). Under some definitions, cropland can also include subcategories cropland pasture (cropland used as pasture), as well as idle cropland (as defined by USDA for the Census of Agriculture) and Conservation Reserve Program lands, but for the purposes of *BT16* analyses, these lands are excluded from the cropland base. Note: County-level distribution is determined by a multi-year average of production from 2010-2013 USDA National Agricultural Statistics Service surveys of agricultural production. It is assumed to be a total 312.6 million acres in the initial simulation year of agricultural production in 2015. (See *BT16* volume 1, appendix C for more details).

Cropland pasture, or cropland used for pasture or grazing – Defined in the 2012 USDA Census of Agriculture appendix B as “land used only for pasture or grazing that could have been used for crops without additional improvement. Also included are acres of crops hogged or grazed but not harvested prior to grazing” (Adapted from the U.S. Department of Agriculture). Note: It is assumed to be a total 11.2 million acres across the projection period. (See *BT16* volume 1, appendix C for more details).

DBH – diameter at breast height – The common measure of wood volume approximated by the diameter of trees measured at approximately breast height from the ground.

CRP – Conservation Reserve Program – A land conservation program administered by the Farm Service Agency (FSA) that pays a yearly rental payment in exchange for farmers removing environmentally sensitive land from agricultural production and planting species that will improve environmental quality. (Definition from U.S. Department of Agriculture FSA Conservation Programs).

Dedicated energy crops – Poised to complement the process to further commercialize biofuels, biopower, and bioproducts, these crops can improve supply security and help control feedstock quality characteristics. Under the *BT16* analyses, these include energy cane, biomass sorghum, switchgrass, miscanthus, and short-rotation woody crops (eucalyptus, pine, poplar, and willow).

EF – emission factor – A measure of the average amount of a specified pollutant or material emitted for a specific type of fuel or process. (Definition adapted from DOE EERE, 2016. Glossary of Energy-Related Terms).

Elasticity of demand – The ratio of the percentage change in the quantity of a good or service demanded to the percentage change in the price. (Definition adapted from DOE EERE, 2016. Glossary of Energy-Related Terms).

Energy cane – A perennial tropical grass with high yield potential across the Gulf South. Low-sugar, high-cellulose varieties (a hybrid of commercial and wild sugar cane species) can be established, managed, and harvested using existing sugar-cane industry equipment.

Enterprise budgets – Financial management tools used by farmers to estimate costs and returns from farm operations.

Environmental indicator – Quantitative variable that can be measured and which provides information about potential or realized environmental effects of human activities on phenomena of concern. For the purposes of *BT16*, environmental indicators, combined with social and economic indicators measure sustainability of bioenergy.

ET – Evapotranspiration – The simultaneous process of evaporation of water from the earth's surface to the atmosphere and transpiration from plants.

Eucalyptus – A short-rotation woody crop ideal for Gulf States as well as Georgia and South Carolina.

Extractable phosphorus (P) – Indicates the amount of available P for plants. Phosphorus is considered one of the most important soil nutrients in typical productive land management systems. (Definition adapted from McBride et al., 2011. Indicators to support environmental sustainability of bioenergy systems. doi:10.1016/j.ecolind.2011.01.010).

Farmgate – Denoting sector of the bioenergy or bioproduct supply chain for agricultural products: after harvest, ready for delivery to a processing facility.

Farm Resource Regions - FRRs – 13 regions used by the POLYSYS model and based on the nine USDA regions carrying the same name.

Feedstock – Raw material for a process (e.g., industrial). Biomass feedstocks are the plant and algal materials used to derive fuels like ethanol, butanol, biodiesel, and other hydrocarbon fuels. (Definition adapted from DOE

EERE, 2016. Glossary of Energy-Related Terms).

Filter strip – A strip or material, usually vegetation, used to separate material (e.g., organic matter, vegetation, or other pollutants) from water. (Adapted from USEPA, 2010. Guidance for Federal Land Management in the Chesapeake Bay Watershed, Glossary).

FIA – Forest Inventory and Analysis – A program of the U.S. Forest Service of the U.S. Department of Agriculture that collects, analyzes, and reports information on the status and trends of America's forests: how much forest exists, where it exists, who owns it, and how it is changing. It has been in continuous operations since 1928. The latest technologies are used to acquire a consistent core set of ecological data about forests through remote sensing and field measurements. The data in this report are summarized from more than 100,000 permanent field plots in the United States.

Forest land – Land at least 10% stocked by forest trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated. (Definition adapted from the U.S. Forest Service of the U.S. Department of Agriculture).

Forestry scenarios – Also called specified-biomass demand levels or "specified-demand" simulations – For the purpose of *BT16* volume 2, these include a baseline scenario with moderate housing -- low wood energy demand (referred to as "ML") and a scenario with high housing-high wood energy demand ("HH"), among other scenario-specific assumptions.

ForSEAM – Forest Sustainable and Economic Analysis Model – a linear programming model used to estimate potential forestry feedstocks in *BT16*.

Fuelwood – Wood used for conversion to some form of energy, primarily for residential use but also has industrial applications (e.g., steel and alloy production in developing countries).

Grasslands – Defined by land cover and also by land use. Under land cover, grasses are dominant in grasslands but plants may also include legumes, forbs, shrubs and, in some locations, sparse tree cover (up to 10% canopy before classification is changed). Thus, grassland cover can include woody plants, grasses and forbs - herbs or non-woody flowering plants that are not grass species. In terms of land use, grasslands can be defined by grazing, haying, and other forms of forage harvest. By these definitions, grassland encompasses a wide variety of grassland types from minimally managed or nearly "natural" grasslands to grassland that was seeded or improved and actively managed for forage production to feed livestock.

GHG – greenhouse gas – Natural or anthropogenic gas that can absorb and emit radiation at specific wavelengths within the spectrum of infrared radiation emitted by the earth's surface, the atmosphere, and the clouds. Water vapor (H_2O), carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), and ozone (O_3) are the primary greenhouse gases in the Earth's atmosphere. (Adapted from the Intergovernmental Panel on Climate Change, 2007. Working Group I: The Physical Science Basis).

Grower payment – Payment to the landowner for the value of standing biomass and rights to harvest biomass, equivalent to "stumpage price" as used in the forest industry (grower payment plus the cost of harvest equals the farmgate price).

Growing stock – A classification of timber inventory that includes live trees of commercial species meeting specified standards of quality or vigor. Cull trees are excluded. When associated with volume, growing stock includes only trees 5.0 inches dbh and larger.

GDP – gross domestic product – A primary indicator of the state of an economy, it is a measure of the value of all goods and services within a given timeframe.

Herbicide concentration (in streams and export) – A measure of the amount (g) of herbicides in a given volume (L) of water extracted from a waterbody (e.g., stream or surface water when considering export or runoff). This is considered an important test to assess the exposure of aquatic life to chemicals, and the potential toxic effects on aquatic life. (Definition adapted from McBride et al., 2011. Indicators to support environmental sustainability of bioenergy systems. doi:10.1016/j.ecolind.2011.01.010).

Housing starts – Used as an indicator of economic conditions, this term represents the number of privately owned new housing construction projects begun in a given year.

ILUC -- Indirect land-use change – Roundabout course or path that modifies human actions of using land, or human purpose(s) of land, or management of natural resources, or benefits derived from natural resources.

Indirect – Not in a direct course or path; roundabout; resulting otherwise than directly or immediately, as "indirect effects or consequences."

Infiltration – Process by which surface level water enters the soil.

Land class – A descriptor for a predominant vegetation feature in a defined land area. LUC models commonly consider a simplified global map with major land classes such as forest, grassland, cultivated cropland, and other (including developed).

Land cover – The physical appearance of the land surface based on a classification system (e.g., forests, grasslands) (Definition from Turner and Meyer, 1994. Global Land-Use/Land-Cover Change: Towards an Integrated Study). Note: change in land cover reflects a shift based on a defined cover classes, regardless of land use. Changes in land-cover classification can result from how data are interpreted or aggregated, the resolution of analysis, threshold definitions for predominant cover, the ontology applied, the scale and order of land cover class analysis, as well as from actual physical changes that cross the threshold values that define a given land-cover class.

Land management –The process of dealing with or controlling the use and development of land resources. Tillage is an example of agricultural land management.

Land use – Human management of terrestrial resources, designated purpose of those resources, or benefits derived from those resources (land use may involve vegetation, animals, soil, groundwater, streams, wetlands, minerals, air flow, and other resources).

LUC – land-use change – Modification of the human actions of using land, or human purposes of land (e.g. zoning), or human management of natural resources, or benefits derived from natural resources.
Note: Almost anything humans do, or dictate, or refrain from doing, that impacts land and related natural resources, could be considered LUC.

LCC – land capability class – A USDA classification system based on soil productivity for common agricultural crops.

Land rent – A payment to the land owner for production (usually agricultural or forestry) on that land.

LAI – leaf area index – The total one-sided area (m^2) of photosynthetic tissue per unit ground surface area (m^2). (Definition from Gobron, 2016. Leaf Area Index, FAO).

Load (or loading) – Quantity delivered to a water body. Synonymous with yield (nutrient yield, water yield). Term is usually used for sediment or nutrients.

Logging residues – The unused portions of growing-stock and non-growing-stock trees cut or killed by logging and left in the woods.

Marginal cost –The added cost of producing one additional unit of an item.

Marginal land (see also degraded-, idle-, underutilized-, etc.) – Relative term that varies widely by country, institution and local conditions. In traditional economics, land is marginal if the combination of yields and prices barely covers cost of production. In practice, the term is generally used more broadly to describe any lands that are not in productive use, presumably due to their low potential productivity, in contrast to other lands yielding rents from services. Depending on time and place, marginal land may also refer to idle, underutilized, barren, inaccessible, degraded, rocky, excess and abandoned lands, or to lands occupied by politically and economically marginalized populations, or land with characteristics that make a particular use unsustainable or inappropriate. Furthermore, the classification of marginal lands by remote means (using satellite imagery or land-cover data sets) involves multiple sources of large uncertainty. For example, what may be seen from above as idle lands may actually be fallow land between cropping regimes, recently harvested lands, or areas that are being mismanaged. (Definition from CBES, 2009. Land-Use Change and Bioenergy: Report from the 2009 workshop, ORNL/CBES-001 USDOE ORNL).

Minimum base flow – A minimum portion of stream water that results from non-runoff sources (e.g., seepage of groundwater). This is a constant, regardless of stream-height. (Adapted from Santhi, 2008. doi:10.1016/j.jhydrol.2007.12.018).

Miscanthus – A sterile triploid with low nutrient requirements and wide adaptability across cropland.

NEI -- National Emissions Inventory – The official United States' air pollutant emissions inventory.

Net change in total area planted in a given crop – The net change in the total areas managed for a specific crop type (e.g., corn) in a defined simulation; calculated by comparing results from two simulations. Note: aggregate or average values were used in assessing net change; a multitude of crop transitions could occur involving crops even if they do not reflect a net change in area.

NH_3 – ammonia – A colorless, pungent, gas (NH3) that is extremely soluble in water, may be used as a refrigerant; a fixed nitrogen form suitable as fertilizer. (Definition adapted from DOE EERE, 2016. Glossary of Energy-Related Terms).

No-till – Describes the process of planting without using conventional tilling processes (e.g., substituting a drill for plow or disk tillage) and is considered an important conservation practice. (Definition adapted from USDA NRCS).

Nonattainment area – Any area that does not meet (or that contributes to ambient air quality in a nearby area that does not meet) the national primary or secondary ambient air quality standard for the pollutant. (Definition adapted from EPA, 2016. Air Quality Planning & Standards, Air Quality).

Non-stationarity – The probability distributions of stochastic processes change with shifts in time and in effect, the process of interest is not in a state of statistical equilibrium.

Nutrient loading – A measure of pollution, this term describes the nutrients in a system (e.g., water body) at a given time.

Ozone precursors – Describes a set of gasses (NOx, CO, and non-methane VOCs - volatile organic compounds) that participate in the chemical reaction to produce ozone in the lower atmosphere (O_3 or tropospheric ozone), a harmful type of pollution. (Definition adapted from USDA NRCS, 2012. Resource Concerns: Air, Ozone Precursors).

Pastureland, all – A category not explicitly defined in the 2012 USDA Census of Agriculture, but estimated as the reported composite category of cropland used as pasture, permanent pasture, woodland pasture, rangeland, irrigated pastureland, and wasteland in the 2012 UDSA Census of Agriculture. It is assumed to be a total 446.3 million acres across the projection period across the *BT16* simulations (i.e. BC1 and HH3).. (See *BT16* volume 1, appendix C).

Perennial grasses – A grass that lives for more than two years.

Permanent pasture – Grazable land that does not qualify as woodland pasture or cropland pasture (per USDA census definition; see volume 1 appendix C and volume 2 chapter 3 for discussion of how pasture was classified for *BT16* projections). Note: This land class may be irrigated or dry land. In some areas, it can be a high quality pasture that could not be cropped without improvements. In other areas, it is barely able to be grazed and is only marginally better than wasteland. It is assumed to be a total 402 million acres across the projection period.

Pine – Also referred to as "southern pine" – A tree representing the major commercial tree crop in the South, with 32 million acres of plantations (Fox, Jokela, and Allen 2007). This crop can be adapted to grow in high density on agricultural land assuming 8-year rotations.

PM – particulate matter – Considered pollution, PM is a mixture of extremely small particles and liquid droplets suspended in the air. (Definition adapted from USEPA, 2016. Particulate Matter (PM) Pollution).

$PM_{2.5}$ - Total particulate matter less than 2.5µm diameter.

PM_{10} - Total particulate matter less than 10µm diameter. **POLYSYS** – Policy Analysis System – An agricultural policy modeling system of U.S. agriculture, including both crops and livestock. It is based at the University of Tennessee Institute of Agriculture, Agricultural Policy Analysis Center.

Poplar – A short-rotation woody crop with great potential in the Lake States, the Northwest, the Mississippi Delta, and other regions.

Price elasticities – Responsiveness of supply or demand to changes in price.

NOx - nitrogen oxides – Nitrogen dioxide and nitric oxide.

reduced-till – Managing the amount, orientation, and distribution of crop and other plant residue on the soil surface year-round while limiting soil-disturbing activities used to grow and harvest crops in systems where the field surface is tilled prior to planting. (Source USDA NRCS, 2016. Conservation Practice Standard, Code 345).

Residue – The portion of a crop (agriculture or forestry) remaining after the primary product is harvested.

RFS – Renewable Fuel Standard – The RFS was established by the Energy Policy Act of 2005. It required 7.5 billion gallons of renewable-based fuel (which was primarily ethanol) to be blended into gasoline by 2012. This original RFS (referred to sometimes as RFS1) was expanded upon (RFS2) by the Energy Independence and Security Act of 2007 (EISA) to include diesel in addition to gasoline as well as to increase the volume of renewable fuel to be blended into fossil-based fuel to 9 billion and ultimately 36 billion gallons by 2022. RFS2 established life-cycle greenhouse gas requirements (less than fossil fuels they replace) for renewable fuels.

Riparian buffer – A section of ecosystem (traditionally terrestrial and sometimes aquatic) along a water body that is used to protect the aquatic ecosystem from impacts of adjacent land uses. Benefits of riparian buffers can include bank stabilization and reduction of non-point source pollution (e.g., nutrient loading) and can depend on their extent and composition. Inclusion of riparian buffers is considered a best management practice and single component of a comprehensive watershed management plans. (Definition adapted from Mayer 2006, EPA/600/R-05/118).

RISI – Pulp and Paper Industry Intelligence – Organization that produces the international wood fiber report, an annual report that "examines the markets for globally traded pulpwood fiber and the pulpwood resources for domestic and export supply in more than 35 countries." (Definition adapted from RISI, 2016. www.risiinfo.com).

Roadside – Denoting sector of the bioenergy or bioproduct supply chain for forestry products: after harvest, ready for delivery to a processing facility.

RUSLE2 – Revised Universal Soil Loss Equation – A computer program that estimates erosion and sediment delivery for conservation planning in crop production.

SCSOC – Surrogate CENTURY Soil Organic Carbon model – Uses calculations and parameters from CENTURY, a soil organic matter model, to estimate SOC changes based on local conditions like crop yield, soil type, and weather data.

Sediment yield – Term is usually used for water and describes the quantity of sediment delivered to a water body. Synonymous with sediment load or loading.

SOC – soil organic carbon – One of the most important soil quality indicators, this term refers to the carbon (C) stored in soils. (Adapted from: Reeves 1997 The role of soil organic matter in maintaining soil quality in continuous cropping systems.; McBride et al., 2011. Indicators to support environmental sustainability of bioenergy systems. doi:10.1016/j.ecolind.2011.01.010).

Soil pH – A measure of soil acidity or alkalinity and a common indicator of soil health due to impact on crop yields, plant nutrient availability, and microorganism activity. (Definition adapted from USDA NRCS, 2011. Soil Quality Indicators).

SO_x – sulfur oxides – Compounds of sulfur and oxygen. Sulfer Dioxide (SO_2) is considered the greatest concern for human health and the environment. (Definition adapted from USEPA, 2016. Sulfur Dioxide (SO_2) Pollution).

SRWC – short-rotation woody crop – Intensively managed, fast-growing species that are purpose-grown in a plantation system with the primary intent to use the wood for bioenergy, biofuels, bioproducts, pulpwood, or, in some limited cases, for lumber. These energy crops produce large amounts of biomass over a short period of time, usually less than 10 years. Depending on the species and the production method, harvest frequency can occur in as little as three years per coppice cycle. For the purpose of *BT16* analyses, these include non-coppice (loblolly pine and poplar) and coppice (eucalyptus and willow) species. (USDOE 2011, Billion Ton Update).

Storm flow – Runoff or flow of water due to a rainfall or storm event.

Strip-harvest – Alternating cut and uncut sections when harvesting a field or forest. **Stumpage price** – The price paid for the right to harvest standing timber ("on the stump") on a given piece of land.

Sustainability – Aspirational concept denoting the capacity to meet current needs while maintaining options for future generations to meet their needs. To make the concept of sustainability operational, consistent approaches are required that facilitate comparable, science-based assessments using measurable indicators of environmental, economic, and social processes (Hecht et al. 2009 .Good policy follows good science: using criteria and indicators for assessing sustainable biofuels production.; McBride et al. 2011 doi:10.1016/j.ecolind.2011.01.010; Dale et al. 2015 DOI: 10.1002/bbb.1562). Notes: Conceptual sustainability and sustainable development goals are described in the Brundtland Report (1987) and the National Environmental Policy Act (U.S. Government 1969), the latter of which committed "to create and maintain conditions under which humans and nature can exist in productive harmony, that permit fulfilling the social, economic and other requirements of present and future generations." Sustainability does not imply a steady state or an absolute value, but instead is a relative and comparative term that must have a defined context, based on clear objectives (Efroymson et al. 2013. doi: 10.1007/s00267-012-9907-5).

Suspended sediment concentration (in streams and export) – A measure of the amount (g) of sediment in a given volume (L) of water extracted from a waterbody (e.g., stream or surface water when considering export or runoff).

Switchgrass – A model perennial native grass, with wide range and potential distribution.

Taxon of special concern – A taxonomic category or group (e.g., species, genus), varying in identity and number by site and region, with value (intrinsic or other) in which there is special concern related to the continuation of that taxon. Term relates to the Species of Special Concern designation, which has varying legal definitions, but which generally means an extremely uncommon species (but not legally designated as endangered or threatened) with unique or highly specific habitat requirements. An indicator of biodiversity, taxa of concern can include rare native species, but also species of commercial value, cultural importance, or recreational value. Taxa of special concern can also be keystone species or those whose impact on the ecosystem is disproportionately large relative to its abundance. (Definition adapted from McBride et al., 2011. Indicators to support environmental sustainability of bioenergy systems. doi:10.1016/j.ecolind.2011.01.010).

Thinning from above - The removal of trees from the dominant and codominant crown classes in order to favor the best trees of those same crown classes. Interchangeable terms: crown thinning, high thinning. (Definition adapted from Adams, et al. 1994. Silviculture Terminology, SAF).

Tillage flexibility index – A flexibility constraint is included in POLYSYS to control switching between tillage classes (tillage production distribution (CTIC 2007. National Crop Residue Management Survey) categories of management: no-till production, reduced tillage, conventional tillage) among each individual crop.

Timberlands – Forest land that is producing or is capable of producing crops of industrial wood, and that is not withdrawn from timber utilization by statute or administrative regulation. Areas qualifying as timberland are capable of producing more than 20 cubic feet per acre per year of industrial wood in natural stands. Currently inaccessible and inoperable areas are included. (See also *BT16* volume 1 glossary).

Tolerable soil loss – An erosion factor important to the soil loss equation for conservation planning, this is the maximum amount of soil loss (tons per acre per year) tolerated while still permitting a high level of crop productivity to be sustained economically and indefinitely. (Definition adapted from RUSLE, 2002. T Value, Institute of Water Research, Michigan State University).

Total factor productivity – A measurement of the contribution of all inputs in production: total agricultural output per unit of total agricultural inputs.

Total nitrogen (N) – An essential nutrient for plants and animals, this compound (sum of ammonia, organic and reduced nitrogen, and nitrate-nitrite) is considered a contaminant when found in excess amounts. In waterways, excessive total N can lead to low levels of dissolved oxygen and negatively alter plant and aquatic life. (Definition adapted from USEPA, 2013. Total Nitrogen).

Total phosphorus (P) concentration in streams (and export) – A measure of the amount (g) of Phosphorus in a given volume (L) of water extracted from a waterbody (e.g., stream or surface water when considering export or runoff).

Traditional perennial crops – Pasture, hay, and cropland pasture that persists for two or more years.

USDA 2015 (agricultural) baseline – Agricultural projections to 2024 developed by the US Department of Agriculture. Commonly referred to as "USDA Baseline" in Billion Ton studies. (Definition from USDA Agricultural Projections No. (OCE-2016-1) 99 pp, February 2016; National Agricultural Statistics Service Agricultural Statistics 2015).

USFPM – The U.S. Forest Products Module – A partial market equilibrium model of the U.S. forest sector that operates within the Global Forest Products Model (GFPM) to provide long-range timber market projections in relation to global economic scenarios. (Definition adapted from USDA Forest Service, 2011. U.S. forest products module: a technical document supporting the Forest Service 2010 RPA Assessment).

VWC – virtual water content – The volume of water used in production of a good or service. (Adapted from Allan, 1993. Fortunately there are substitutes for water otherwise our hydro-political futures would be impossible).

VOC – volatile organic compound – A variety of organic chemicals that are emitted as gases from solids or liquids. (Definition adapted from EPA . 2016. Volatile Organic Compounds' Impact on Indoor Air Quality.).

Water footprint analysis – Method to quantify the consumptive water use during the production of a product (e.g., biomass).

Water intensity – Gallons of water consumed to produce a unit of feedstock or the amount of water consumed from an acre of land for feedstock production annually.

Water yield – The total outflow from all or part of a drainage basin through either surface channels or subsurface aquifers within a given time (e.g., one year).

Whole-tree harvest – There are four combinations of harvest methods and intensity for whole trees: 1) full-tree clear cut, 2) full-tree thinning, 3) cut-to-length clear cut, and 4) cut-to-length thinning. The full-tree method can use the entire tree, including branches and tops. The cut-to-length method harvests logs only, leaving logging residue behind. For both methods, the intensity can be either clear cut or thinning. Clear cutting removes all of the standing trees in a selected area. Thinning removes part of the standing trees in a selected area.

Willow – A short-rotation woody crop assumed to be managed on a 20-year cycle and harvested at 4-year growth stages. It is being commercialized widely in the Northeast.

Wood pellets – Generally made from compacted forestry residues (e.g., sawdust and industrial waste), these are the most common type of pellet fuel and are included in the BT16 advanced supply system analyses.

List of Acronyms

BU -- Bushels

DOE – United States Department of Energy

EISA – The Energy Independence and Security Act of 2007

EPA – United States Environmental Protection Agency

FPEAM – Feedstock Production Emissions to Air Model

GFPM – Global Forest Products Model

GREET® - Greenhouse gases, Regulated Emissions, and Energy use in Transportation model

HRU – Hydrologic Response Unit

ISO – International Organization for Standardization

MODIS – Moderate Resolution Imaging Spectroradiometer

NASS – National Agricultural Statistics Service

NAAQS – National Ambient Air Quality Standards

NOAA - National Oceanic and Atmospheric Administration

NRCS – Natural Resource Conservation Service

OAI – Ogallala Aquifer Initiative

SO_x - oxides of sulfur

PBR – photobioreactors

PRISM-EM – Parameter-elevation Relationships on Independent Slopes Model (PRISM) Environmental-Model

SCI – the Soil Conditioning Index

SCM – Supply Characterization Model

SSURGO – Soil Survey Geographic Database

Sun Grant RFP – Sun Grant Regional Feedstock Partnership

SWAT - Soil Water Analysis Tool

USDOE – U.S. Department of Energy

USGS – U.S. Geological Survey

WaSSI – Water Supply Stress Index Ecosystem Services Model

WATER – Water Analysis Tool for Energy Resources

WEPS – Wind Erosion Prediction System

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