UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF COLUMBIA

SIERRA CLUB Plaintiff, v. ANDREW WHEELER, in his official capacity as Acting Administrator, U.S. Environmental Protection Agency,[1] Defendant.	Case No. 1:17-cv-02174-APM

EPA's Cross-Motion for Summary Judgment and Opposition to Sierra Club's Motion for Summary Judgment

[1] Andrew Wheeler has been substituted for Scott Pruitt under Fed. R. Civ. P. 25(d).

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INTRODUCTION

Sierra Club has brought a lawsuit that is not wholly within this Court's jurisdiction. It first argues that EPA missed deadlines to report to Congress on the environmental impact of the agency's renewable-fuel program. But this first claim is not justiciable now that EPA has issued its report. The Court should thus deny Sierra Club's motion for summary judgment and grant summary judgment for EPA on that claim.

Sierra Club also alleges that EPA missed deadlines to complete a study on renewable fuel's impact on air quality and to take certain follow-up action based on the study. On this point, the agency concedes that it is liable. The only question, then, is remedy—the deadline this Court should set for EPA to perform those tasks. The most expeditious deadline for completing the study is 14 months from the date of the Court's ruling on summary judgment. Because the study's results will inevitably affect the time needed for the follow-up action, at this point the agency cannot meaningfully propose—and this Court cannot meaningfully evaluate—a deadline for that action. EPA thus asks the Court to (1) set a deadline of 14 months from the date of its ruling for completion of the study, but (2) hold this matter in abeyance as to the follow-up action. Once EPA completes the study, the parties can submit briefs on remedy for the follow-up action.

BACKGROUND

I. The Clean Air Act

Under the Clean Air Act, 42 U.S.C. §§ 7401-7671q, transportation fuel sold or introduced into commerce must contain certain volumes of renewable fuel, which is made from renewable biomass like corn.[2] *See* 42 U.S.C. § 7545(o)(1)(I) & (J), (o)(2); *see generally id.* § 7545(o).

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[2] In response to paragraph 11 of Sierra Club's statement of undisputed facts, EPA agrees that the Administrator has set annual standards under 42 U.S.C. § 7545(o), but notes that that provision does not establish a statutory maximum for conventional biofuels. *See* Memorandum of Points

Cont.

Sierra Club's lawsuit involves two statutory provisions that direct EPA to study renewable fuel's environmental impacts. First, EPA has to "assess and report to Congress on the [renewable-fuel program's] impacts . . . and likely future impacts" on environmental and resource-conservation issues. *Id.* § 7545 note ("Environmental and Resource Conservation Impacts"). That assessment is set forth in triennial reports, so called because they must be submitted to Congress every three years, starting in December 2010. *Id.* EPA released its first triennial report in December 2011 after notice and comment. *See* Biofuels and the Environment: First Triennial Report to Congress, at ix.[3] EPA released its second triennial report[4] on June 29, 2018, shortly after Sierra Club filed its summary-judgment motion.[5] *See* Attach. 1.

Second, EPA has to complete what is called the anti-backsliding study. *See* 42 U.S.C. § 7545(v)(1)(A). In this study, due "18 months after December 19, 2007" (that is, June 19, 2009), EPA must determine "whether the renewable fuel volumes required by [Section 7545] will adversely impact air quality as a result of changes in vehicle and engine emissions" of certain air pollutants. *Id.* Then, "3 years after December 19, 2007" (that is, December 19, 2010), EPA must either (1) considering the study's results, promulgate regulations to "implement appropriate measures to mitigate" any adverse impacts on air quality; or (2) determine that no mitigation measures are necessary. *Id.* § 7545(v)(2). For simplicity we refer to these two alternative

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and Authorities in Support of Plaintiff's Motion for Summary Judgment (Br.), ECF No. 25-1, at 16 & n.10.

[3] *Available at* https://cfpub.epa.gov/si/si_public_file_download.cfm?p_download_id=506091&Lab=NCEA (last visited Aug. 8, 2018).

[4] Sierra Club's summary-judgment motion overlooks the 2011 triennial report. *See* Br. at 2 (requesting that EPA prepare a "first" triennial report); *id.* at 28 n.15 (referring to report allegedly due in December 2013 as the "First Triennial Report"). In contrast, the complaint correctly refers to the two allegedly overdue reports as the second and third triennial reports, a practice we follow here. *See, e.g.*, Compl., ECF No. 1, ¶ 54.

[5] EPA's counsel sent Sierra Club's counsel a copy of the second triennial report later that day.

duties—promulgating regulations or determining that none is needed—collectively as the "follow-up action."

Finally, the Clean Air Act authorizes citizen suits against the EPA Administrator for failure to perform a mandatory duty. *See* 42 U.S.C. § 7604(a)(2). A plaintiff bringing such a lawsuit must first give the Administrator 60 days' notice of the alleged violation. *Id.* § 7604(b). The citizen-suit provision waives a limited part of the United States' sovereign immunity by giving federal district courts jurisdiction to "order the Administrator to perform [a statutorily required] act or duty[.]" *Id.* § 7604(a).

II. The complaint

In February 2017 Sierra Club notified EPA of its intent to sue the agency for failure to issue the second and third triennial reports, complete the anti-backsliding study, and take follow-up action. *See* Addendum to Memorandum of Points and Authorities in Support of Plaintiff's Motion for Summary Judgment, Ex. B, ECF No. 25-2. Eight months later Sierra Club filed its complaint. *See* ECF No. 1. After EPA timely answered, for some months the parties tried unsuccessfully to settle the dispute. *See* ECF Nos. 12, 14, 21, 23.

The complaint states two claims. First, it alleges that EPA violated the Clean Air Act and the Energy Independence and Security Act[6] by failing to complete the second and third triennial reports required by 42 U.S.C. § 7545 note. *See* Compl., ECF No. 1, ¶¶ 50-56. Second, the complaint alleges that EPA violated the two acts by failing to complete the anti-backsliding study and to take the follow-up action required by 42 U.S.C. § 7545(v). *See id.* ¶¶ 57-64.

[6] The Energy Independence and Security Act of 2007, Pub. L. No. 110-140, 121 Stat. 1492, amended the Clean Air Act's renewable-fuel provisions.

STANDARD OF REVIEW

Under Fed. R. Civ. P. 56(a), courts "shall grant summary judgment if the movant shows that there is no genuine dispute as to any material fact and the movant is entitled to judgment as a matter of law." Claims of failure to perform mandatory duties may be resolved on summary judgment. *See, e.g.*, *Sierra Club v. Pruitt*, 238 F. Supp. 3d 87 (D.D.C. 2017). Of course, before a court can consider such claims, it must determine whether it has jurisdiction. *See, e.g.*, *Lance v. Coffman*, 549 U.S. 437, 439 (2007) (per curiam).

ARGUMENT

I. This Court lacks jurisdiction over Sierra Club's first claim.

Federal courts have only such jurisdiction as granted to them by the Constitution or statute. *See Kokkonen v. Guardian Life Ins. Co. of Am.*, 511 U.S. 375, 377 (1994). The only source of jurisdiction here is the Clean Air Act's citizen-suit provision. *See* 42 U.S.C. § 7604. That provision confers no jurisdiction on this Court to consider Sierra Club's first claim as to the triennial reports.

To begin, the citizen-suit provision limits the judicial relief available to Sierra Club to an "order [to] the Administrator to perform [his mandatory] duty . . . ." 42 U.S.C. § 7604(a). But now that the second triennial report has been released, "the Court is unable to grant any relief beyond requiring steps that EPA has already taken . . . ." *Sierra Club v. Browner*, 130 F. Supp. 2d 78, 82 (D.D.C. 2001). This portion of the claim is thus moot. *See id.*; *Theodore Roosevelt Conservation P'ship v. Salazar*, 661 F.3d 66, 79 (D.C. Cir. 2011) (explaining that a claim is moot if it becomes "impossible for the court to grant any effectual relief whatever" (internal quotation marks omitted)); *Daimler Trucks N. Am. LLC v. EPA*, 745 F.3d 1212, 1216 (D.C. Cir. 2013) ("the case must remain live at all stages of review, not merely at the time the complaint is filed." (internal quotation marks omitted)).

As for the third triennial report, it is not due until 2021, three years from now. *See* 42 U.S.C. § 7545 note (requiring reports "every 3 years").[7] And EPA has not missed that deadline. Because a district court's "limited statutory authority under [the citizen-suit provision] vests only *after* EPA has failed to undertake some mandatory action prior to a certain deadline," this Court currently lacks statutory authority to grant relief. *Sierra Club v. Browner*, 130 F. Supp. 2d at 93 ("in advance of a deadline's expiration, the agency has not yet failed to undertake its duty"); 42 U.S.C. § 7604(a). This Court has no jurisdiction to review the first claim, and so it should deny that portion of Sierra Club's summary-judgment motion and enter summary judgment for EPA.

II. The Court should grant EPA's proposed remedies as to Sierra Club's second claim.

EPA does not dispute that it failed to complete the anti-backsliding study or to take follow-up action as required by Section 7545(v).[8] The only issue, then, is remedy. The most expeditious deadline for the agency to reasonably complete the study is 14 months from the date of the Court's ruling. And because EPA must consider the study's results before it knows how much time it needs to take follow-up action, the Court should hold that portion of Sierra Club's motion in abeyance until after the study is complete, at which point the parties can submit briefs on the proper deadline for that action.

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[7] More precisely, the statute provides that the first report is due "3 years after the enactment of this section [on] Dec. 19, 2007," and that later reports are due "every 3 years thereafter." 42 U.S.C. § 7545 note (internal brackets omitted). A very literal reading of this language would deem December 19, 2016 as the third triennial report's deadline. But the point of the "every 3 years" requirement is to build in a meaningful interval between triennial reports so each can reflect any changes since the last one. So a better reading is that the third report is not due for another three years, in 2021, and EPA objects to Sierra Club's contrary statement of undisputed facts. *See* Br. at 14.

[8] As discussed *supra* in Argument Section I, EPA's deadline for completing the third triennial report has not passed, so the agency is not liable on the first claim.

A. The most expeditious deadline for completing the anti-backsliding study is 14 months from the date of the Court's ruling.

A district court has broad discretion to fashion equitable remedies such as a deadline for EPA to perform a mandatory duty. *See Natural Res. Def. Council, Inc. v. Train*, 510 F.2d 692, 713 (D.C. Cir. 1974); *Sierra Club v. Pruitt*, 238 F. Supp. 3d at 89. The court is not, as Sierra Club suggests, limited by the applicable statutory timetable, *see* Memorandum of Points and Authorities in Support of Plaintiff's Motion for Summary Judgment (Br.), ECF No. 25-1, at 31; it may give the agency additional time for compliance "where it is convinced by the official involved that he has in good faith employed the utmost diligence in discharging his statutory responsibilities." *Train*, 510 F.2d at 713; *see Envtl. Def. Fund v. Thomas*, 627 F. Supp. 566, 569-70 (D.D.C. 1986) (finding EPA's proposed compliance schedule, one that exceeds the statutory timeframe, "reasonable" and adopting it where agency had failed to comply with a nondiscretionary duty). Indeed, in a similar case decided recently, Judge Huvelle accepted EPA's proposed deadline when EPA's declaration was "sufficiently detailed to support [its] proposed deadline and to satisfy the Court" that plaintiff's proposed deadline "would be unattainable." *Sierra Club v. Pruitt*, 238 F. Supp. 3d at 89; *see Train*, 510 F.2d at 713 (prohibiting courts from ordering EPA "to do an impossibility" (internal quotation marks omitted)). Relevant factors in assessing what is "attainable" include "manpower demands" and "methodological constraints" on the agency's ability to complete the necessary tasks. *Train*, 510 F.2d at 712-13.

Here, EPA's current plan to complete the anti-backsliding study requires 14 months from the date of the Court's order.[9] *See* Attach. 2, Declaration of Christopher Grundler (Grundler Decl.), ¶ 9. That is the most expeditious deadline that EPA can meet. *See id.* Sierra Club's proposed deadline of the end of this year—only four months away—ignores the analytical complexities and resource constraints that EPA faces. *See* Br. at 2.

As presently conceived, the study, which would examine renewable fuel's impact on air quality, is a complex endeavor. *See* Grundler Decl. ¶ 8. It involves estimating emissions from motor vehicles and their engines under different conditions, and predicting how those emissions would behave in the atmosphere in the presence of emissions from other sources (like power plants). *See id.* ¶¶ 4-6, 8, 18. Many factors come into play, such as vehicle and engine properties, fuel properties, emissions from other sources, weather, and so on. *See id.* ¶¶ 6, 14-15, 18; 42 U.S.C. § 7545(v)(1)(B). To further complicate matters, because the study has to consider both a world with the required renewable-fuel volumes and a world without, EPA has to model two different scenarios. *See* 42 U.S.C. § 7545(v)(1)(A); Grundler Decl. ¶¶ 9, 14(a), 15, 18. It takes time to address these complexities and to assure the quality of EPA's work product. *See* Grundler Decl. ¶¶ 8, 9, 11.

Importantly, EPA accelerated its timeline by planning to execute different steps of the study concurrently when possible. *See id.* ¶ 13, 18.[10] Many steps, however, must be performed sequentially because the bulk of the air-quality analysis entails generating information from one

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[9] As EPA explains in its declaration, because 14 months is the most expeditious deadline for the agency's current plan to complete the study, a shorter deadline ordered by the Court would require the agency to reevaluate its plans. For that reason, EPA intends to initiate the study once the Court orders a deadline for it. *See* Grundler Decl. ¶ 9.

[10] EPA also plans to run information from representative counties, instead of all counties, through one of its models, thus significantly reducing the computation time on this step. *See* Grundler Decl. ¶ 14.

model and then using it as the input in another model. *See id.* ¶¶ 7, 13-15, 17-18 (describing the process of modeling emission factors that are used to model vehicle-emission rates, which are then used to model air quality). So EPA cannot save time by running all models simultaneously. Likewise, the agency cannot interpret results and develop conclusions before finishing the modeling work. *See id.* ¶ 21. Nor can EPA proceed to the next step of its analysis without completing quality assurance of the previous one. *See id.* ¶¶ 11, 13. In fact, step-by-step quality assurance is critical to EPA's plan to expeditiously complete the study because if earlier errors do not come into light until very late in the process, the agency would have to redo all previous steps. *See id.* ¶ 11.

In proposing its deadline, EPA assumed that its relevant subject-matter experts will be fully involved in the study. *See id.* Because air-quality analysis calls for specialized skills, adding EPA employees without the relevant expertise would not speed things up. *See id.* EPA's proposal also accounts for the use of contractors when they can shorten the schedule. *See id.* ¶¶ 10, 20. But expanding the contractor workforce beyond what EPA proposes is unlikely to hasten the study's completion, for contractors lack the necessary expertise to do the work assigned to EPA employees. *See id.* ¶ 16. And though contractors can be trained for a particular task, it can take EPA more time to train them and review their work product than if agency employees did the work themselves. *See id.*

Thus, the proposed deadline of 14 months from the date of the Court's order is the most expeditious one for the agency to complete the anti-backsliding study. That is the deadline the Court should order EPA to comply with, and the Court should deny Sierra Club's request for a four-month deadline.

B. The Court should hold in abeyance Sierra Club's motion for a deadline on the follow-up action until after the anti-backsliding study is completed.

EPA currently lacks sufficient information to rationally propose a deadline for the follow-up action. So the Court should hold that aspect of Sierra Club's motion in abeyance and delay its decision until after completion of the anti-backsliding study, which will supply the missing information.

Without seeing the study, EPA cannot know what type of follow-up action—promulgate regulations to implement mitigation measures, or determine that none is needed—it will take, let alone how long it needs for that action. Indeed, Congress specifically required EPA to consider the study's conclusions in taking follow-up action and gave the agency time to do so. *See* 42 U.S.C. § 7545(v)(2) (setting deadline for follow-up action at 3 years from December 19, 2007—which is 18 months after the deadline for the anti-backsliding study). Having that time makes sense because what the study says about the kind of impacts that renewable fuel has on air quality could greatly affect whether EPA requires mitigation measures as well as the types of measures it considers in a rulemaking. Grundler Decl. ¶¶ 22-23, 42. And the measures considered could, in turn, determine the kind of supporting analyses that EPA will conduct. *See id.* ¶¶ 22-23. Some of the analyses may be straightforward and take relatively little time. *See id.* ¶ 30, 34. Others may be extremely time-consuming, whether because they are complicated, require modeling or data that do not currently exist, or both. *See, e.g.*, *id.* ¶¶ 28-30, 34.

Even if EPA determines that no mitigation measures (and thus no rulemaking) are necessary, it would still need to know the study's conclusions to propose a deadline for making a determination. If the agency finds no adverse air-quality impacts, then it may need only about three months to complete its determination. *See id.* ¶ 42. But if the study concludes that some adverse impacts exist, EPA may need to undertake extensive analysis—the same as it would in a proposed rulemaking—to determine whether control measures are necessary. *See id.*

Put another way, EPA may need to perform extensive analysis whatever follow-up action it takes, because—depending on the anti-backsliding study—it may need to consider the results of that analysis before deciding whether to propose a mitigation rule or determine that none is needed. Sierra Club's request that EPA make the determination immediately upon the study's completion would thus force the agency to act without adequate information. *See* Br. at 2-3.

So at this point, it is impossible to know precisely how much time is needed for the follow-up action. *See* Grundler Decl. ¶¶ 24, 42. EPA's best estimate of the most expeditious deadline for a proposed mitigation rulemaking is 23 to 44 months; for a final rule, 13 to 26 months; and for a determination that no mitigation measures are necessary, 3 to 44 months. *See id.* ¶ 44. Those are big ranges, driven by uncertainty and contingencies at nearly every turn. Without the anti-backsliding study, EPA cannot rationally propose a deadline and this Court lacks sufficient information to decide whether a particular deadline is attainable. *See Sierra Club v. Pruitt*, 238 F. Supp. 3d at 89. We thus ask the Court to not rule on this portion of Sierra Club's summary-judgment motion at this time. Once EPA completes the anti-backsliding study, the parties can submit briefs on the appropriate deadline for the follow-up action based on the study's conclusions and the Court can decide the issue. If the Court nevertheless wishes to set a deadline now, the deadline should account for not only the 18 months that Congress provides, but also EPA's estimated timeframe based on the complex work involved in the follow-up action.[11] *See* 42 U.S.C. § 7545(v)(2) (providing 18 months after study's completion for EPA to

[11] Sierra Club asks that the Court set the comment period for any proposed regulation at 30 days. *See* Br. at 3. But the length of any comment period is for EPA to decide, subject to the

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take follow-up action); *Train*, 510 F.2d at 712 (recognizing "the need for some leeway for modification of [a statutory] deadline when circumstances preclude [the relevant action's completion] by that date.").

CONCLUSION

The Court should (1) deny Sierra Club's summary-judgment motion as to the triennial-report claim and grant summary judgment for EPA; (2) direct EPA to complete the anti-backsliding study by 14 months from the date of the order; and (3) hold the portion of Sierra Club's motion as to the follow-up action in abeyance and order the parties to brief the matter after EPA completes the anti-backsliding study.

Respectfully submitted,

Dated: August 13, 2018

Jeffrey H. Wood
Acting Assistant Attorney General

/s/ Sue Chen
Sue Chen
Trial Attorney
U.S. Department of Justice
Environment & Natural Resources Division
Environmental Defense Section
P.O. Box 7611
Washington, D.C. 20044
202.305.0283

Of Counsel:
David Orlin
Seth Buchsbaum
U.S. Environmental Protection Agency
Office of General Counsel
Washington, D.C.

requirements of the Clean Air Act. *See Sierra Club v. Browner*, 130 F. Supp. 2d at 89 (observing that the court's "power [under the Act's citizen-suit provision] is limited to requiring EPA to undertake nondiscretionary actions required by the statute."); Grundler Decl. ¶ 38.

CERTIFICATE OF SERVICE

I certify that on August 13, 2018, I filed the foregoing and its attachments using the Court's CM/ECF system, which will electronically serve all counsel of record registered to use the CM/ECF system.

/s/ Sue Chen
SUE CHEN

Attachment 1

EPA/600/R-18/195
June 2018

Biofuels and the Environment

Second Triennial Report to Congress

U.S. Environmental Protection Agency
Office of Research and Development
Washington, DC

June 29, 2018

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Contributors

Report Leads

Paul Argyropoulos[1]	Anne Grambsch[1]
Julia Burch	C. Andrew Miller

Lead Authors

Randy Bruins[1]	Stephen D. LeDuc
Dallas Burkholder	Joe McDonald
Christopher Clark	Briana Niblick
Rich Cook	C. Andrew Miller
Rebecca Dodder	Scott Unger[2]
Anne Grambsch[1]	Dilip Venugopal[3]

Contributing Authors

Ellen Cooter	Mike Madden
Kathleen Fahey	Robert Sabo
Mark Johnson	Jay R. Reichman
Aaron Levy	Joseph Schubauer-Berigan

[1] Retired from EPA.
[2] Now at Maricopa County, Arizona
[3] Now at the University of Maryland

Executive Summary

Background

This report is the second of the U.S. Environmental Protection Agency's (EPA's) triennial reports to Congress required under the 2007 Energy Independence and Security Act (EISA). EISA Section 204 calls for EPA to report to Congress on the environmental and resource conservation impacts of the Renewable Fuel Standard (RFS) program, specifically air and water quality, water quantity, ecosystem health and biodiversity, soil quality, invasive species, and international environmental impacts (hereafter referred to as the Section 204 statutory impacts).

Consistent with how EPA conducted the first Section 204 report, EPA has chosen in this assessment to focus on the Section 204 statutory impacts and not to expand the scope of the report beyond the factors explicitly enumerated in the law. As a result, some environmental impacts are not discussed in this report. Lifecycle greenhouse gas emissions impacts of biofuel use, for example, are addressed as part of the RFS program and are not included in this report. Furthermore, this report does not include a comparative assessment of the impact of biofuels on the environment relative to the impacts of other transportation fuels or energy sources, including fossil fuels, for every environmental endpoint. For example, the environmental impacts of growing corn, refining ethanol from that corn, and burning that ethanol in an internal combustion engine has a different environmental impact than drilling for oil, refining gasoline, and burning that in an internal combustion engine. EPA recognizes that a fully comprehensive assessment of the benefits and impacts of biofuel production and use would be broader than what is represented by this report, but conducting such an evaluation is beyond the scope of this study.

This report updates the findings of the first Report to Congress, published in 2011, with respect to environmental and resource conservation impacts, and, together, the two reports are intended to address the Section 204 statutory impacts since the passage of the EISA. The primary conclusions of the 2011 Report included the following two findings: (1) the environmental and resource conservation impacts of biofuel production and use as delineated in Section 204 of EISA were, on balance, negative; and (2) EISA's goals could be achieved with relatively minimal adverse environmental impacts if existing conservation and best management practices were widely employed, concurrent with advances in technologies that facilitate the use of second-generation feedstocks. The 2018 Report reaffirms the findings of the 2011 Report and reflects the current understanding about biofuel production using data gathered through May 2017. The 2018 Report also reviews data on U.S. land use and the scientific literature through April 2017.

Major Findings

- Data from observations made since the 2011 Report indicate that the biofuel production and use conditions that led to the conclusions of that report have not materially changed.
- Substantial volumes of cellulosic and advanced biofuels have not been produced as anticipated by EISA. The Section 204 statutory impacts anticipated as a consequence of large-scale use of feedstocks other than corn and soybeans have therefore not occurred.
- Corn grain and soybeans remain by far the dominant feedstocks for biofuel production. Biofuel production associated with large-scale cultivation of corn and soybeans contributes to the adverse environmental and resource conservation impacts of the type listed in EISA Section 204, though we caution that this report does not evaluate the net effects of displacing petroleum or other energy sources with biofuels.
- There has been an observed increase in acreage planted with soybeans and corn between the decade leading up to enactment of EISA and the decade following enactment. Evidence from observations of land use change suggests that some of this increase in acreage and crop use is a consequence of increased biofuel production mandates.
- It is likely that the Section 204 impacts associated with land use change are, at least in part, due to increased biofuel production and use associated with the RFS. However, at this time we cannot quantify with precision the amount of land with increased intensity of cultivation nor confidently estimate the portion of crop land expansion that is due to the market for biofuels.

Likely Future Impacts

Section 204 of EISA also requires that the triennial report identify likely future impacts. We interpret the requirement to address “likely futures” as encompassing near-term future impacts presuming current technologies and rates of market penetration, and current policy and market dynamics. Thus, where this report discusses likely future impacts, it is addressing anticipated changes over the next three to five years. This report finds that there are some indications of growth in cellulosic ethanol production, primarily from corn stover, but that large-scale production at levels approaching the original EISA targets is not likely to be reached in the next few years. Available data suggest that current trends using corn starch and soybeans as primary biofuel feedstocks, with associated environmental and resource conservation impacts, will continue in the near term.

Overall Conclusions

Reports and data published since the first Report to Congress have increased the confidence in the conclusions of that report (Table ES-1).

Table ES-1. Comparison of overarching conclusions from first and second reports to Congress.

Conclusions from the first Report to Congress	Conclusions from second Report to Congress
Evidence to date from the scientific literature suggests that *current* environmental impacts from increased biofuels production and use associated with EISA 2007 are negative but limited in magnitude.	Disregarding any effects that biofuels have on displacing other sources of transportation energy, evidence since 2011 indicates the specific environmental impacts listed in EISA Section 204 are negative. The environmental and resource conservation impacts, whether positive or negative, related to displacement of other transportation energy sources by biofuels were not assessed.
Published scientific literature suggests a potential for both positive and negative environmental effects in the future.	Literature published since 2011 supports the conclusion of the potential for positive and negative effects. Available information suggests, without accounting for the environmental effects of displacing other sources of transportation energy, the specific environmental impacts listed in EISA Section 204 are negative in comparison to the period prior to enactment of EISA.
EISA goals for biofuels production can be achieved with minimal environmental impacts if existing conservation and best management practices are widely employed, concurrent with advances in technologies that facilitate the use of second-generation feedstocks.	Evidence continues to support the conclusion that biofuel production and use could be achieved with reduced environmental impacts. The majority of biofuels continue to be produced from corn grain and soybeans, with associated impacts that are well understood. Cellulosic and other feedstocks remain a minimal contributor to total biofuel production.

Specific Conclusions

Land use change

- **Evidence since enactment of EISA suggests an increase in acreage planted with soybeans and corn, with strong indications from observed changes in land use that some of this increase is a consequence of increased biofuel production.**

Since the first Report to Congress there have been several advances in our understanding of land use change in the United States. Land use change has been identified as one of the primary drivers of environmental impacts from an expanding biofuels industry. However, the connections between land use change due to biofuels and environmental effects have not been evaluated sufficiently to allow quantification specifically attributable to biofuel production. There are strong indications that biofuel feedstock production is responsible for some of the observed changes in land used for agriculture since enactment of EISA. However, we cannot quantify with precision the amount of land with increased intensity of cultivation nor confidently estimate the portion of crop land expansion associated with the market for biofuels.

Air Quality

- **The emission impacts of biofuel production and distribution, and offsetting indirect impacts on petroleum fuel production and distribution, are important to consider along with end-use impacts for volatile organic compounds (VOCs), particulate matter (PM), and oxides of nitrogen (NO_x); emission and air quality impacts associated with feedstock production and conversion of feedstock to biofuels are highly localized.**

Emissions of NO_x, SO_x, CO, VOCs, NH_3, and particulate matter can be impacted at each stage of biofuel production, distribution, and usage. These impacts depend on feedstock type, land use change, and feedstock production practices. Ethanol from corn grain has higher emissions across the life-cycle than ethanol from other feedstocks, and ethanol facilities relying on coal have higher air pollutant emissions than facilities relying on natural gas and other energy sources. The magnitude, timing, and location of emissions changes can have complex effects on the atmospheric concentrations of criteria pollutants (e.g., O_3 and $PM_{2.5}$) and air toxics, the deposition of these compounds, and subsequent impacts on human and ecosystem health. Only limited data exist on the impacts of biofuels on the tailpipe and evaporative emissions of Tier 3 light-duty vehicles and light-duty vehicles using advanced gasoline engine technologies to meet GHG emissions standards. Comprehensive studies of the impacts of biofuels on the emissions from advanced light-duty vehicle technologies, similar in scope to previous studies of such impacts on Tier 2 vehicles, would improve understanding.

Water quality

- **Demand for biofuel feedstocks may contribute to harmful algal blooms, as recently observed in western Lake Erie, and to hypoxia, as observed in the northern Gulf of Mexico. Changes to future nitrogen and phosphorus loadings will depend on feedstock mix and crop management practices.**

The increased intensity of corn production on land already under cultivation and the expansion of corn and soybean cultivation onto grasslands negatively impact water quality but have not been consistently quantified to date. Differences in nutrient application, management practices, and runoff characteristics make direct connections between increased feedstock production and water quality impacts difficult to quantify and assess. Empirical studies suggest water quality impacts but the magnitude of these changes is variable across the landscape and may be detectable only in some regions. Recent modeling studies conclude that row crop agriculture plays an important role in driving downstream impacts such as harmful algal blooms, particularly in fresh waters, and hypoxia, particularly in coastal waters, and suggest that biofuel feedstock production is a contributing factor. Continued adoption and expansion of sustainable conservation practices are expected to decrease nutrient loadings and associated adverse impacts.

Water quantity

- **There are some indications of increased water use due to increases in irrigated areas for corn and elevated land conversion rates in more arid Western states. Adverse water availability impacts will most likely arise in already-stressed aquifers and surface watersheds. Irrigation practices are dependent on a number of economic and agronomic factors that drive land management practices making attribution of increased irrigation and water quantity to biofuels difficult.**

Quantitative evaluations are needed to understand increases in water use through changes in land use and/or land management change, to understand whether those changes can be directly or indirectly attributed to feedstock production for biofuels, and to determine whether increases in water demands and water stress have occurred or are occurring along water-stressed areas or “hot spots” (e.g., Ogallala aquifer) where high water demands and water stress are coinciding.

Ecosystem health and biodiversity

- **The conversion of environmentally-sensitive land to cropland consistent with increased production of current biofuel feedstocks is associated with negative impacts to ecosystem health and biodiversity**

Loss of grasslands and wetlands are occurring in ecologically sensitive areas, including the Prairie Pothole Region. Row crop expansion is resulting in the loss of habitat and landscape simplification. Increasing pesticide use for feedstock production is associated with negative impacts to pollinators, birds, soil-dwelling organisms, and other ecosystem services both in terrestrial and aquatic habitats. Increased fertilizer applications of N for corn and of P for corn and soybean have known negative effects on aquatic biodiversity. Opportunities exist for continued adoption and expansion of practices and technologies that will enhance ecosystem services and sustainable feedstock production.

Soil quality

- **Conversion of grasslands to annual production of the dominant biofuel feedstocks typically adversely affects soil quality, with increases in erosion and the loss of soil nutrients and soil organic matter, including soil carbon.**

Impacts of this conversion can be partially mitigated – though not entirely – through the adoption of management practices such as conservation tillage. Corn stover, a cellulosic biofuel feedstock, is now being harvested at the commercial scale in Iowa, and the scientific literature suggests this must be done carefully to avoid negatively affecting soil quality and crop yields.

Invasive Species

- **Current biofuel feedstocks pose little risk of becoming invasive species. Cultivation of herbicide-tolerant feedstock crops (e.g., glyphosate-tolerant soybean) and concomitant application of the associated herbicide (e.g., glyphosate) has the potential to contribute to herbicide-resistant weed development, just as herbicide-tolerant crops grown for other purposes.**

International

- **U.S. ethanol imports have decreased while biodiesel and renewable diesel imports have increased, leading to potential land use change impacts in countries of origin. Exports of corn, DDGS, soybeans, and ethanol primarily increased.**

Reports indicate that demands for biofuel feedstocks have led to market-mediated land use impacts (both direct and indirect land use changes) in the past decade. Quantification and causal attribution of land use change and international environmental impacts due to biofuel production and renewable fuel standards remains uncertain. Comprehensive causal analysis frameworks and coordinated frameworks for evaluating land use changes across biofuel trading nations may help our understanding of international land use change and environmental impacts.

Recommendations

To promote actions to address impacts, EPA recommends the following:

- **Additional research in coordination with other organizations (e.g., federal agencies, states, trade organizations) is recommended to better characterize land use change due to changes in biofuel feedstock production.**
- **Efforts at the federal level, as described by the Biomass Research and Development Board, to improve efficiencies and sustainability of processes across the biofuel supply chain should be continued and strengthened where possible.**
- **An ecosystem approach is recommended to evaluate environmental and natural resource impacts of biofuel production. Such an approach provides an integrative perspective that accounts for complex interactions of multiple stressors across different locations.**
- **Incorporating local information and perspectives will improve understanding of changes at local scales, which will enhance opportunities for improved information and will enable targeted responses to prevent and mitigate adverse impacts of biofuel production and use.**
- **Best management practices should be encouraged, incentivized, and otherwise expanded to promote conservation and sustainability in agricultural systems.**

1 Introduction

In December 2007, Congress enacted Public Law 110-140, the Energy Independence and Security Act (EISA), with the stated goals of providing "greater energy independence and security [and] to increase the production of clean renewable fuels." In accordance with these goals, EISA revised the Renewable Fuel Standard (RFS) program, created under the 2005 Energy Policy Act[1] and managed by the U.S. Environmental Protection Agency (EPA), to increase the volume of renewable fuel required to be blended into transportation fuel from 9 billion gallons per year in 2008 to 36 billion gallons per year by 2022.

The revised statutory provisions and implementing regulations (commonly known as the RFS2 program) specify increasing applicable volumes of cellulosic biofuel, biomass-based diesel, advanced biofuel, and total renewable fuel that EPA is directed to use (unless it establishes lower volume requirements using specified waiver authorities) in establishing annual percentage standards for these renewable fuel categories in transportation fuel. The purpose of this report is to examine the environmental and resource conservation impacts of the RFS2 program, as required under EISA Section 204.

EISA Section 204 calls for EPA to report to Congress every three years on the environmental and resource conservation impacts of increased biofuel production and use as stated in the relevant text of the Act:

In General. Not later than 3 years after the enactment of this section and every 3 years thereafter, the Administrator of the Environmental Protection Agency, in consultation with the Secretary of Agriculture and the Secretary of Energy, shall assess and report to Congress on the impacts to date and likely future impacts of the requirements of Section 211(o) of the Clean Air Act[2]on the following:

1. *Environmental issues, including air quality, effects on hypoxia, pesticides[3], sediment, nutrient and pathogen levels in waters, acreage and function of waters, and soil environmental quality.*

[1] The 2005 Energy Policy Act amended the Clean Air Act and established the first national renewable fuel standards. The statute specifies the total volume of renewable fuel that is to be used based on the volume of gasoline sold in the United States.

[2] EISA 2007 amended Section 211(o) of the Clean Air Act to include the definitions and requirements of RFS2.

[3] Pesticides include antimicrobials, fungicides, herbicides, nematicides, insecticides, and rodenticides.

2. *Resource conservation issues, including soil conservation, water availability, and ecosystem health and biodiversity, including impacts on forests, grasslands, and wetlands.*
3. *The growth and use of cultivated invasive or noxious plants and their impacts on the environment and agriculture.*
4. *The report shall include the annual volume of imported renewable fuels and feedstocks for renewable fuels, and the environmental impacts outside the United States of producing such fuels and feedstocks. The report required by this subsection shall include recommendations for actions to address any adverse impacts found.*

The first report to Congress was completed in 2011 (hereafter the 2011 Report) and provided an assessment of the environmental and resource conservation impacts associated with increased biofuel production and use (EPA 2011). Although many impacts had been speculated or anticipated by the July 2010 publication cutoff date for inclusion in the 2011 Report, few had been actually observed in the peer reviewed literature. Thus, the first report was largely forward-looking and evaluated the potential impacts of several assumed future scenarios that were common in the literature. The overarching conclusions of the 2011 Report were: (1) the environmental impacts of increased biofuel production and use were likely negative but limited in impact; (2) there was a potential for both positive and negative impacts in the future; and (3) EISA goals for biofuels production could be achieved with minimal environmental impacts if best practices were used and if technologies advanced to facilitate the use of second-generation biofuel feedstocks (corn stover, perennial grasses, woody biomass, algae, and waste).

This is the second report on the current and potential future environmental impacts associated with the requirements of Section 211(o) of the Clean Air Act. This report updates the findings of the 2011 Report with respect to EISA Section 204 statutory impacts, provides recommendations to address adverse impacts, and reflects the current understanding concerning biofuel production using data gathered through the RFS program and other federal databases through May 2017. We also reviewed U.S. data on land use and peer-reviewed scientific literature through April 2017, focusing on observed changes as opposed to projected changes in impacts associated with changes in feedstocks, fuel types, and volumes. This report focuses on the Section 204 statutory impacts since passage of the EISA in 2007. Where appropriate, the report provides additional information over longer time frames to provide context for the discussion of impacts.

EPA identified a number of studies during the review process that were published after the April 2017 cut-off date or that were not peer-reviewed. Where their findings were sufficient to require changes to this report's conclusions or recommendations, such changes were made, if they demonstrated the required data quality. We did not conduct original quantitative analyses for this report. A qualitative review was considered necessary to meet the publication schedule for this report, and we anticipate that the recommendations from this report will guide research, including possible quantitative analyses, in preparation for the next triennial report.

In establishing the scope of this report, EPA chose to adhere closely to the enumerated requirements of EISA Section 204. Therefore, this report addresses only the Section 204 statutory impacts associated with implementation of the RFS2 program. This report does not attempt to quantitatively evaluate either the current or potential future benefits associated with the program. It does, however, point to possible opportunities for future improvements related to the Section 204 statutory impacts. It is important to note the distinction between a "likely future" as prescribed in Section 204 of EISA and a "potential future." We interpret likely futures to encompass near-term future impacts presuming current technologies and rates of market penetration, and current policy and market dynamics. Thus, where this report discusses likely future impacts, it is addressing anticipated changes over the next three to five years. Where this report discusses potential future impacts, it is recognizing the possibilities for changes over the longer term that can affect the environmental and natural resource impacts associated with biofuels.

An exploration of the potential longer-term future benefits associated with use of biomass for energy and other products is found in the 2016 Billion-Ton Report developed by the U.S. Department of Energy (DOE) (DOE 2016). Operating with greater analytical freedom than indicated by EISA's Section 204, DOE developed scenarios of biomass production and use that far exceed current levels. The DOE report provides a potentially useful complement to this report's focus on observed changes in recent years. In addition, the U.S. Department of Agriculture (USDA) supports the sustainable production of high-quality, nonfood feedstocks for conversion into bioproducts, bioenergy, bioheat, and biopower. These efforts support the broader federal Bioeconomy Initiative "…to develop and coordinate innovative approaches to expanding the sustainable use of America's abundant biomass resources, while maximizing economic, social, and environmental benefits."[4]

Consistent with the 2011 Report and EISA Section 204, this report does not evaluate emissions of carbon dioxide (CO_2) or other greenhouse gases (GHGs), nor does it review and assess studies that analyze GHG impacts in its conclusions (see Box 1, "Greenhouse Gas Emissions and Impacts" for

[4] The Billion Ton Bioeconomy Initiative: Challenges and Opportunities; Biomass Research & Development Board, Washington, DC, 2016. Available at: https://biomassboard.gov/pdfs/the_bioeconomy_initiative.pdf

details). Emissions of GHGs over the life cycle of biofuel production, conversion, and use are addressed under the RFS program. This report focuses on the Section 204 statutory impacts and therefore does not attempt to make detailed comparisons to the estimated impacts associated with use of other transportation fuels or energy sources. EPA acknowledges that a lack of comparative assessments for every environmental endpoint is a limitation on the ability of this report to draw conclusions regarding the comprehensive environmental impacts of biofuels, but we believe that the information provided nonetheless provides value by reviewing observed impacts specifically from biofuel production and use.

This report emphasizes U.S. impacts; however, the substantial market created for biofuels by the U.S., Brazil, and other countries has important global implications. For example, countries that produce

Box 1. Greenhouse Gas Emissions and Impacts

A key feature of EISA is the establishment of mandatory life cycle GHG reduction thresholds for the renewable fuels that are intended by the law to displace petroleum based fuels. EPA used state-of-the-art models, data, and other information to project the GHG impacts of biofuels, as described in the RFS2 Final Regulatory Impact Analysis (RIA) (EPA 2010). EPA conducted a formal, independent peer review of key components of the analysis. The modeling of GHG emissions in the RFS2 RIA provides a reasonable and scientifically sound basis for making threshold determinations and estimating GHG impacts. As EPA conducts lifecycle assessments for new fuel pathways, the most recent science and data are incorporated wherever possible (see https://www.epa.gov/renewable-fuel-standard-program/fuel-pathways-under-renewable-fuel-standard). For example, EPA has updated its analyses to reflect new data on fuel conversion efficiencies, forest carbon stocks, projected crop yields, and agricultural inputs. The GHG impacts associated with biofuel production and use remains an area of active research, and EPA continues to evaluate the relevant science to inform consideration of the need for any reevaluation of previous determinations of life cycle GHG emissions. Other agencies and institutions also evaluate life cycle GHG emissions, providing information for comparative purposes. The Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) spreadsheet analysis tool developed by Argonne National Laboratories (Burnham et al. 2006) is an example of such a tool. As discussed above, this report does not attempt an evaluation of emissions of carbon dioxide or other GHGs, nor does it attempt to encompass GHG impacts in its conclusions. Instead, this report provides information on other impacts that is complementary to the GHG impacts described in the RIA (EPA 2010), which should be consulted for more information on this topic.

(or will produce) feedstocks that are converted to biofuels that qualify for use in the U.S. will experience direct impacts; other countries (including the U.S.) will have to adapt to changing agricultural commodity distributions that result from diversion of food exports to biofuel production. While there may be economic or other benefits to such market changes, this report focuses on the environmental and natural resource impacts of increased feedstock and biofuel production in other countries as a result of U.S. policy, as required and defined under EISA Section 204.

The information included in both the first and second biofuels Reports to Congress is considered foundational for future efforts to quantitatively compare the potential environmental impacts of alternative scenarios for meeting the goals of the RFS2 program. They serve as a starting point for future assessments, especially for the next triennial assessment, and for taking action to achieve the goals of EISA. Future reports will reflect the evolving understanding of biofuel impacts in light of new research results and data as they become available.

Box 2. Second Generation Biofuels

The requirement in EISA Section 204 to assess "the likely future impacts" of the Renewable Fuel Standard means that second-generation biofuels and feedstocks must be considered in this report. It is clear from the fuel volumes specified in EISA that Congress anticipated the production of substantial volumes of cellulosic ethanol and other second-generation fuels. Although such levels of production have not yet been reached, the requirement to assess the likely future impacts remains.

Because these fuels have not yet reached large-scale commercial production when compared to other fuel types, there are limited observational data to illustrate the environmental and resource conservation impacts related to production of those fuels. Thus, this report must rely upon model-based projections to provide any assessment of impacts. There is potential that the projected impacts will be more serious than anticipated, given that large-scale commercial production will likely create incentives for increased use of production methods and chemical inputs that can reduce the anticipated environmental benefits of grasses and other second-generation feedstocks.

Although second-generation biofuels have not yet demonstrated an economic benefit over current feedstock-fuel combinations, the promise of improved environmental performance at economically acceptable costs continues to encourage development.

1.1 Organization of this Report

Chapter 2 provides information on the drivers of environmental issues, including biofuel volumes, feedstocks, conversion technologies, agricultural practices, and U.S. land use changes. Chapter 3 focuses on the implications of biofuel production for environmental and natural resource issues and includes a summary of impacts to date. Chapter 4 presents overarching conclusions and provides recommendations for improving scientific understanding, as well as practices for minimizing environmental impacts. Chapter 5 describes a path forward for future reports, including options for the scope of the next triennial report based on the findings of this report and reported advances in the science.

2 Drivers of Environmental Impacts

Numerous factors influence the markets for biofuels and the associated environmental impacts of their production and use. These factors, which are “drivers” of the Section 204 statutory impacts, include: regional considerations; scale and volume of commercial biofuel operations; development of biofuel conversion processes; changes in vehicle technologies; and changes in agricultural practices due to biofuel production and implications for environmental impacts. Each of these, whether individually or in combination, will affect the ultimate environmental impacts associated with biofuel production and use. Land use change is both a driver of environmental impacts and an environmental impact directly affected by other market and non-market drivers discussed above.

As noted in the 2011 Report, many potential Section 204 statutory impacts of biofuel production and use (e.g., water quality, water quantity, biodiversity) are a result of land use conversion and the subsequent management of that land. Management includes tillage practices, nutrient application, and other chemical inputs during feedstock production. Air quality impacts depend largely on the volume of biofuels used, their impact on vehicle air pollutant emission rates, and the emissions associated with their production and distribution.

As EPA does lifecycle assessments for new fuel pathways, the most recent science and data are incorporated where possible. For example, EPA has updated its analyses to reflect new data on fuel conversion efficiencies, forest carbon stocks, projected crop yields, and agricultural inputs. The GHG impacts associated with biofuel production and use remains an area of active research, and EPA continues to evaluate relevant science to inform consideration of the need for any reevaluation of previous GHG determinations.

This chapter presents information on these key drivers: biofuel production, feedstock production, vehicle type and use, conversion technologies, and land use change.

2.1 Biofuel Volumes

2.1.1 U.S. Biofuel Production

Since 2012 the production of biofuels in the U.S. has grown steadily, rising from 14.1 billion gallons in 2012 to 16.6 billion gallons in 2016 (see Figure 1). As in 2012, ethanol and biodiesel remain the types of biofuels produced and consumed in the largest quantities in the U.S. However, in recent years the production of other biofuels, such as renewable diesel and biogas used as transportation fuel, have increased.

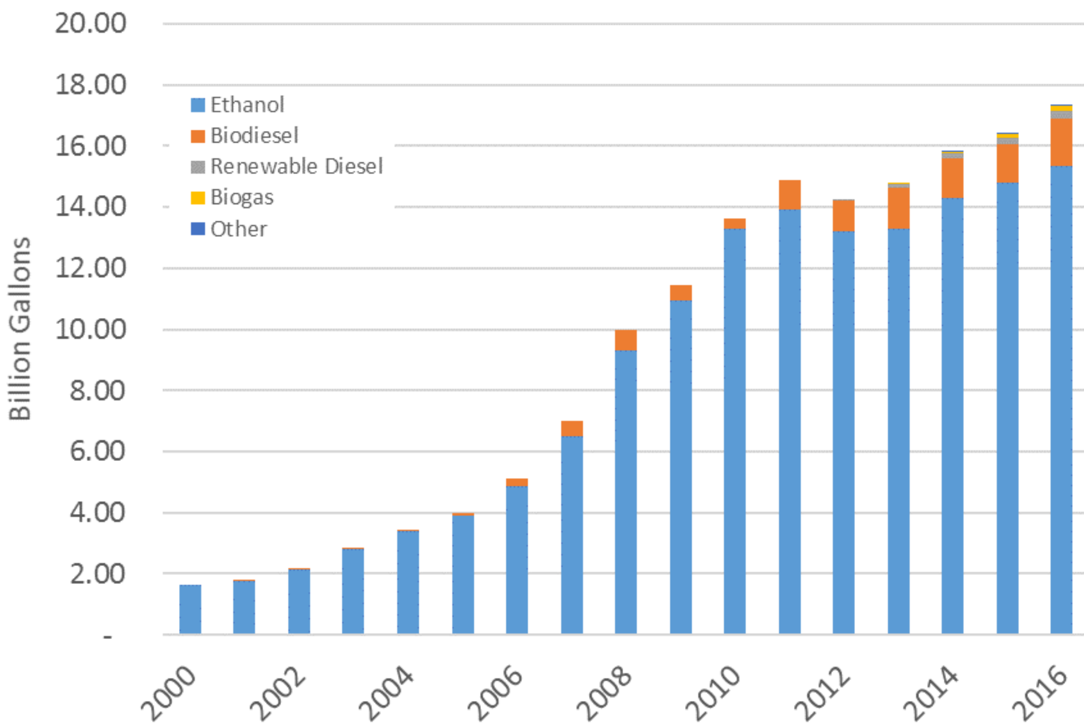

Figure 1 Annual U.S. biofuel production, 2000-2016. [5]

After a rapid rise in U.S. ethanol production from 2007 to 2011, more recent U.S. ethanol production has increased relatively slowly, from 13.22 billion gallons in 2012 to 15.33 billion gallons in

[5] Data for ethanol and biodiesel from USDA ERS US Bioenergy Statistics (https://www.ers.usda.gov/data-products/us-bioenergy-statistics/); ethanol data available in Table 2, and biodiesel data available in Table 4. Renewable diesel, biogas, and other data from EPA's public data for

2016. This slower rate of growth is likely due to challenges associated with the E10 blendwall.[6,7] Since 2013, nearly all gasoline sold in the U.S. has contained at least 10% ethanol. To further expand the ethanol market in the U.S. would require greater sales of fuel blends that contain higher levels of ethanol, such as E15 or E85. To date, sales of such fuels have been limited. If transportation fuel consumption in the U.S. declines in future years, as projected by the U.S. Energy Information Administration (EIA), demand for ethanol will likely also decline unless sales of E15 and/or E85 increase to offset the lower consumption of E10. In this section we have presented data from 2000-2016 where available, but have focused our discussion on the years since 2007.

U.S. production of biodiesel has increased fairly steadily since 2007, with temporary declines in 2009 and 2010. In 2016, biodiesel production in the U.S. reached a record high of 1.56 billion gallons. Demand for biodiesel has likely been driven by increasing volume requirements under the RFS as well as national and state-level incentives and requirements.

2.1.2 Biofuel Imports

Since 2012, the volume of biofuels imported into the U.S. has grown, rising from 666 million gallons in 2012 to 1.07 billion gallons in 2016 (see Figure 2). Over the same time period, the types of biofuels imported into the U.S. have changed significantly. Prior to 2012, ethanol was the predominant biofuel type imported into the U.S. However, ethanol imports decreased significantly starting in 2013. At the same time rising RFS standards, along with national and state-level incentives and requirements, have resulted in a significant increase in the volume of biodiesel and renewable diesel[8] imported into the U.S.

2.1.3 Biofuel Exports

After reaching a high of 1.27 billion gallons in 2011, biofuel exports decreased to 870 million gallons in 2012 before rising steadily to 1.18 billion gallons in 2016 (see Figure 3). Ethanol exports increased significantly in 2011 and have remained high since, as ethanol production capacity in the U.S.

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the Renewable Fuel Standard (https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-renewable-fuel-standard).

[6] E10 is a gasoline blend with 9% to 10% ethanol content; E15 is a gasoline blend with >10% to 15% ethanol content; and E85 is a gasoline blend with 51% to 83% ethanol content.

[7] The blendwall for E10 refers to the point at which all gasoline in the US is blended with 10 volume percent ethanol, at which point the ability to consume additional ethanol through blending in gasoline is challenged by limitations on the existing vehicle fleet and market to go to higher blend concentrations.

[8] Biodiesel is a renewable fuel produced through transesterification of organically derived oils and fats. Renewable diesel is derived from biomass, generally using a thermal depolymerization process.

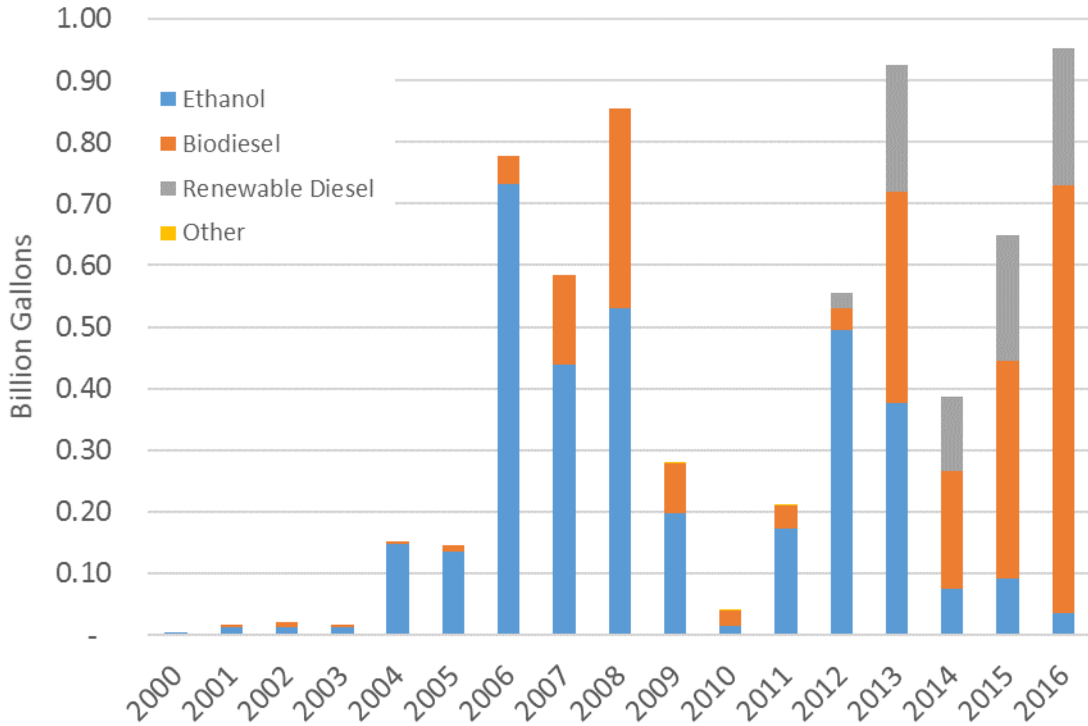

Figure 2 Annual biofuel volumes imported into the U.S., 2000-2016.[5]

has exceeded the ability to consume ethanol due to the E10 blendwall. Biodiesel exports have been low since 2010, as the RFS program has provided a significant incentive for the U.S. consumption of biofuels, especially non-ethanol biofuels that are not subject to the challenges associated with the E10 blendwall. From 2012-2016 ethanol exports ranged from a low of 620 million gallons in 2013 to a high of 1.05 billion gallons in 2016. From 2012-2016 biodiesel exports ranged from a low of 80 million gallons in 2014 to a high of 200 million gallons in 2013. A small volume of renewable diesel exports (40 million gallons) were reported for the first time in 2016.

2.2 Feedstocks

The primary planted crops used as biofuel feedstocks in the U.S. are corn and soybeans. This section will therefore focus on the planted acres, total production, end uses, and management practices for these two crops. In this section we again present data from 2000-2016 where available, but have focused our discussion on the years since enactment of the EISA.

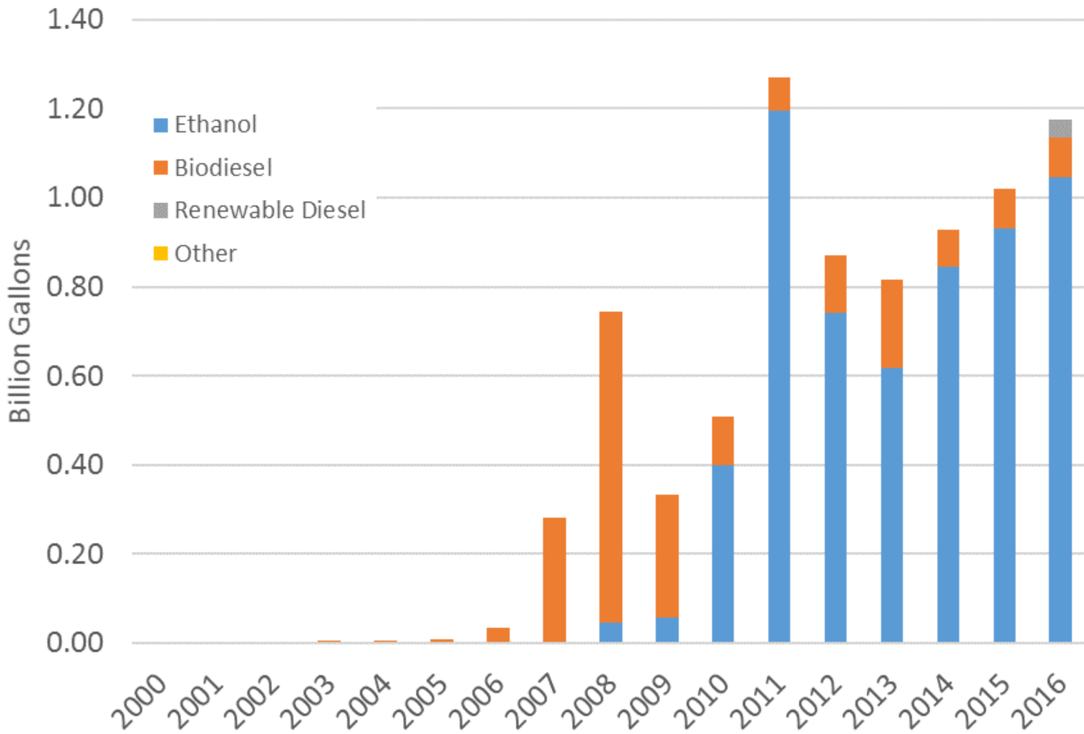

Figure 3 Annual biofuel volumes exported from the U.S., 2000-2016.[5]

2.2.1 Acreage

The number of planted corn acres fluctuated considerably between 2006 and 2016. After an average of roughly 80 million acres between 2000 and 2007, planted acres of corn increased to roughly 90 million acres between 2007-2016 (see Figure 4), with higher variability in the 2007-2016 period than before 2007. A modest general increase in soybean acreage is also evident over the period 2000-2016, averaging between 70-75 million acres between 2000 and 2006 and increasing to 82-83 million acres from 2014-2016. Total corn and soybean acres planted in 2007 were significantly different than in preceding years, with high corn and low soybean acreages planted because of U.S. and international market and climatic factors.

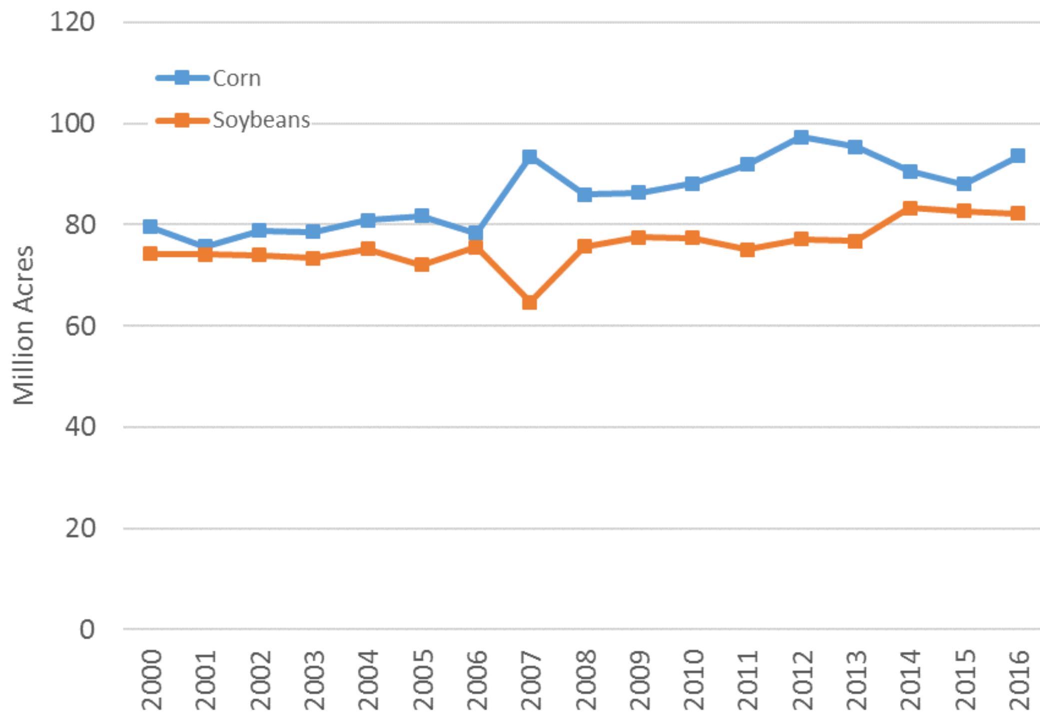

Figure 4 Total U.S. annual planted acres of corn and soybeans, 1996-2017[9]

2.2.2 Total Production of Biofuel Feedstocks

The total production of corn and soybeans has increased over time since enactment of EISA in 2007. From 2007-2016 corn production ranged from a low of 10.8 billion bushels in 2012 to a high of 15.1 billion bushels in 2016 (see Figure 5). From 2007-2016 soybean production ranged from a low of 2.7 billion bushels in 2007 to a high of 4.3 billion bushels in 2016. Productivity for both corn and soybeans was unusually low in 2012 due to drought conditions in many areas of the U.S.

2.2.3 End Use of Biofuel Feedstocks

Corn used for ethanol production has increased since enactment of EISA, but remained relatively steady from 2010 through 2016, accounting for a low of 4.64 billion bushels of corn in 2012 and a high of 5.21 billion bushels of corn in 2016 (see Figure 6). Corn used for ethanol production as a

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[9] https://quickstats.nass.usda.gov/

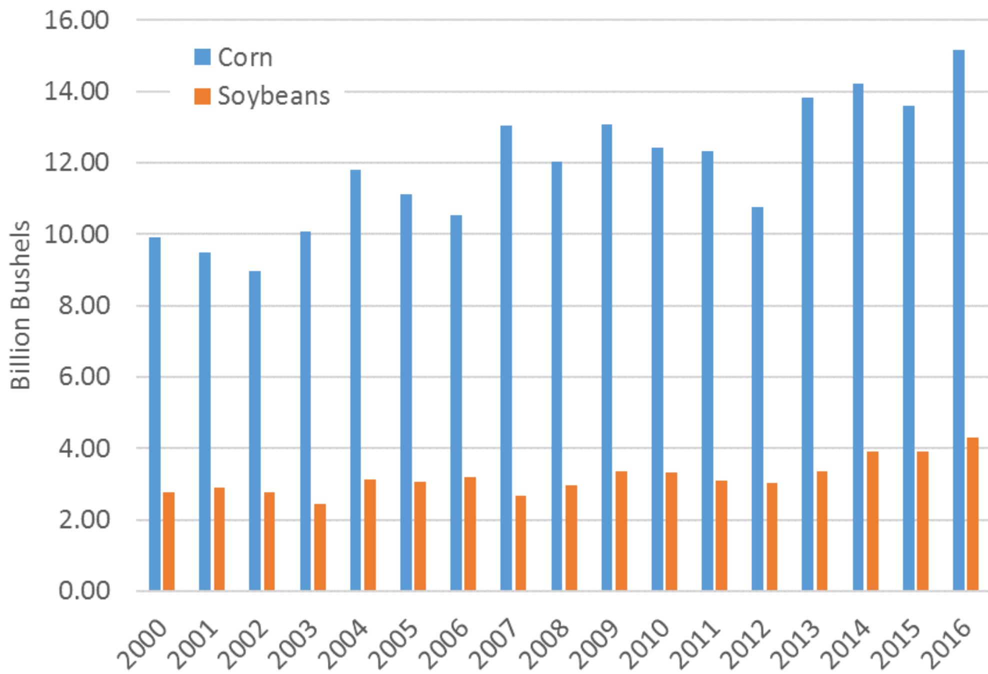

Figure 5 Total annual U.S. corn and soybean production volumes, 2000-2016.[9]

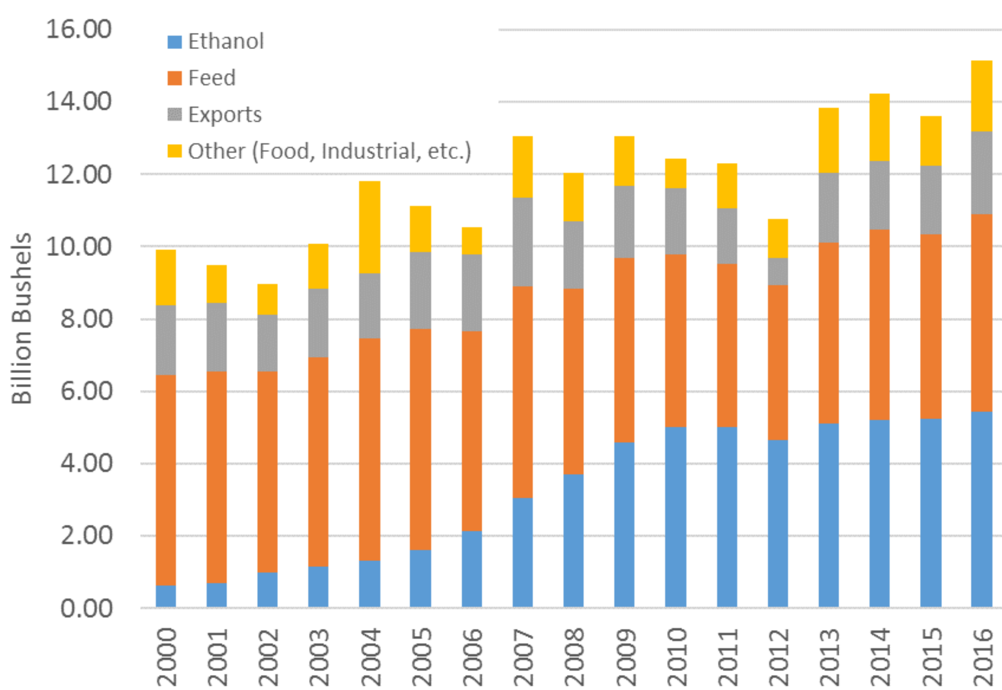

Figure 6 Annual volumes of U.S. corn used for fuel and other purposes, 2000-2016.[10]

[10] Data from USDA ERS US Bioenergy Statistics (https://www.ers.usda.gov/data-products/us-bioenergy-statistics/); U.S. corn use data available in Bioenergy Statistics Table 5.

percentage of overall corn production increased from 19% in 2007 (the first commodity market year prior to enactment of EISA in December 2007) to between 38% and 42% between 2011 and 2016. Corn used for ethanol production as a percentage of overall corn production was relatively stable from 2012-2016, with a high of 42% in 2013 and a low of 38% from 2014-2016. Corn used for feed generally decreased from 2007 through 2011 and then generally increased from 2012-2016, with a low of 4.52 billion bushels of corn used for feed in 2012 and a high of 5.28 billion bushels used for feed in 2015. These numbers do not account for feed sourced from distillers grains, an important co-product of corn ethanol production. Approximately 32% of each bushel of corn used for ethanol production (approximately 12% of the total corn production from 2014-2016) is returned to the feed market in the form of distillers grains.[11] Therefore, these numbers may overstate the quantity of corn used for ethanol production and under-represent the quantity of corn used for feed by approximately 32%. Corn exports generally decreased from 2007-2011 but increased from 2012-2016. From 2012-2016 corn exports ranged from 1.54 billion bushels in 2012 to 1.90 billion bushels in 2016. All uses of corn decreased in 2013 following the relatively low corn production in 2012 caused by a drought.

The use of soybeans in all sectors increased from 2007 to 2016. Soybeans used for biodiesel production increased from 0.40 billion bushels in 2012 to 0.53 billion bushels in 2016 (see Figure 7). Soybeans used for biodiesel production as a percentage of overall soybean production increased from 9% in 2007 to 13% in 2011. Soybeans used for biodiesel production as a percentage of overall soybean production was relatively stable from 2012-2016, with a high of 13% in 2012 and 2016, and a low of 11% in 2014. These percentages are estimated based on the total soybean crush and the percentage of U.S. soy oil used to produce biodiesel in each year.[12] These percentages likely overstate the percentage of the soybean crop used for biodiesel production by approximately 80%, as only the soybean oil (which is approximately 20% of the soybean by weight) is used for biodiesel production, while the non-oil components of the soybean are generally used in the feed market. Soybean crush (for non-biodiesel uses) increased from 1.69 billion bushels in 2012 to 1.94 billion bushels in 2016. Soybean exports increased from 1.33 billion bushels in 2012 to 2.03 billion bushels in 2016. Soybeans used for seed and feed increased from 0.09 billion bushels in 2012 to 0.13 billion bushels in 2016.

[11] A bushel of corn weighs approximately 56 pounds. On average, 18 pounds of distillers grains are produced for every bushel of corn used to produce ethanol.

[12] U.S. soybean use for biodiesel production is estimated by multiplying the total soybean crush by the percentage of U.S. soybean oil used for biodiesel production in each year.

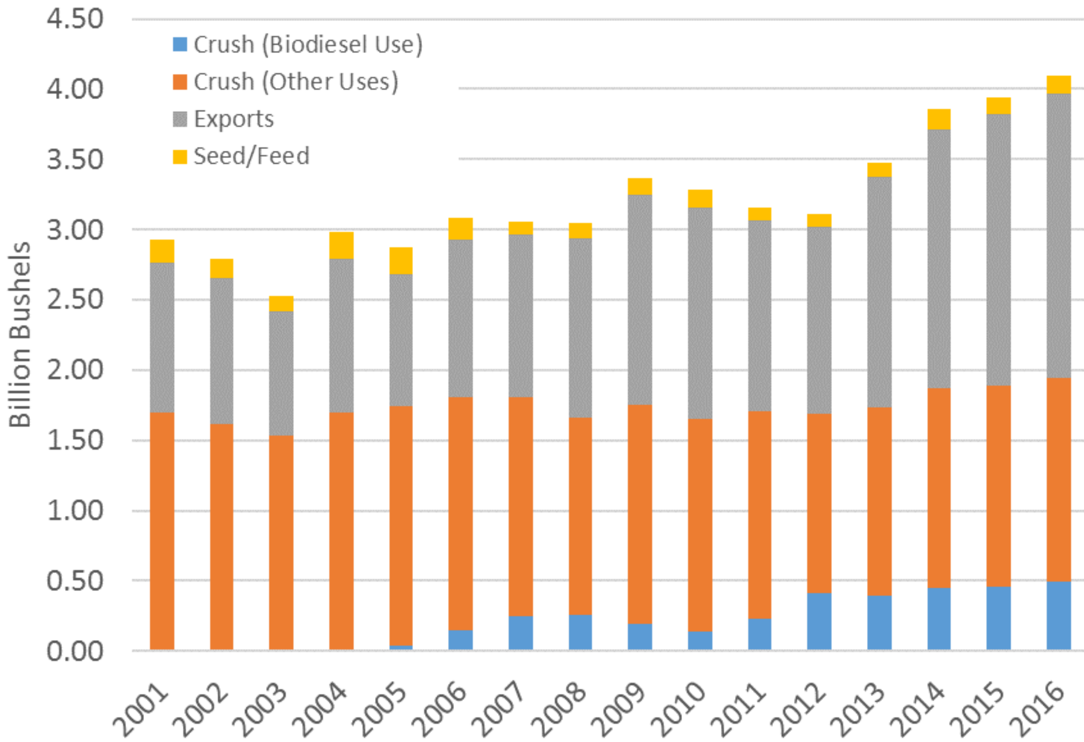

Figure 7 Annual volumes of U.S. soybeans used for fuel and other purposes, 2000-2016.[13]

2.2.4 Nutrients Applied

Nutrient usage for corn production has generally increased since 2000 in total amount applied per year and the rate of application per unit area, albeit with a notable decline in 2002 (see Figure 8). Total nutrients in the form of nitrogen applied increased from 9.75 billion pounds in 2000 to 12.2 billion pounds in 2016 (Figure 8A). This was mainly due to increased acreage and the increase in average rate applied per year from 136 pounds per acre to 145 pounds per acre over the same period (Figure 8B). Phosphate usage increased from 3.1 to 4.2 billion pounds, and potash usage increased from 3.8 to 4.5 billion pounds between 2000 and 2016 (Figure 8A). Sulfur use increased from 0.13 to 0.5 billion pounds during 2005-2016 (Figure 8A).[14]

Nutrient usage for soybean production has increased since 2000 in total amount applied per year but remained somewhat stable in the rate of application per unit area (see Figure 9). Total nitrogen supplied increased from 0.32 to 0.38 billion pounds from 2000 to 2015. Potash applied increased from 1.4 billion

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[13] Data for Soybean use from USDA ERS Oil Crops Yearbook (https://www.ers.usda.gov/data-products/oil-crops-yearbook/oil-crops-yearbook/). Soybean use for biodiesel was estimated using data on total soybean crush and the share of soybean oil seed for biodiesel from USDA ERS US Bioenergy Statistics Table 6 (https://www.ers.usda.gov/data-products/us-bioenergy-statistics/).

[14] USDA, National Agricultural Statistics Service Quick Stats, https://quickstats.nass.usda.gov/.

pounds in 2000 to 2.5 billion pounds in 2015 mainly due to increased acreage and the increase in average rate applied per year from 76 to 83 pounds per acre. Similarly, phosphate usage increased from 0.82 to 1.56 billion pounds from 2000 to 2015 along with the average applied per year from 48 to 51 pounds per acre.[14]

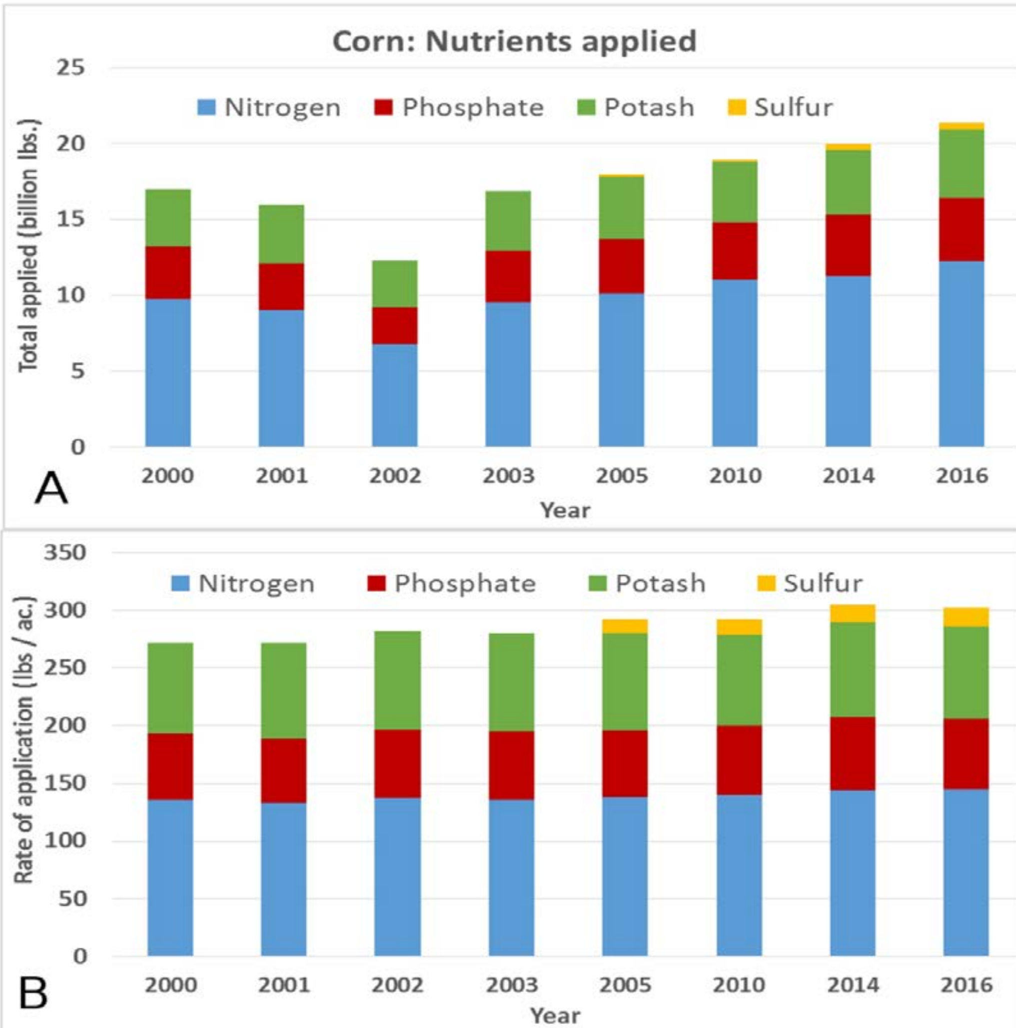

Figure 8 Total volumes of fertilizers used for U.S. corn production (A) and rates of application (B), 2000-2016.[14]

It is important to recognize the improvements in corn and soybean production per acre and the associated per bushel change in applied nutrients. Between 2000 and 2016, corn yield increased by approximately 25%, with a 15% reduction in pounds of nitrogen per bushel and a reduction of 8% in pounds of phosphate per bushel. Over the same time period, soybean yield increased about 39% and application of nitrogen fell by about 28% in terms of pounds per bushel. Application of phosphate and potash per bushel both increased, by 22% for phosphate and 15% for potash.[13]

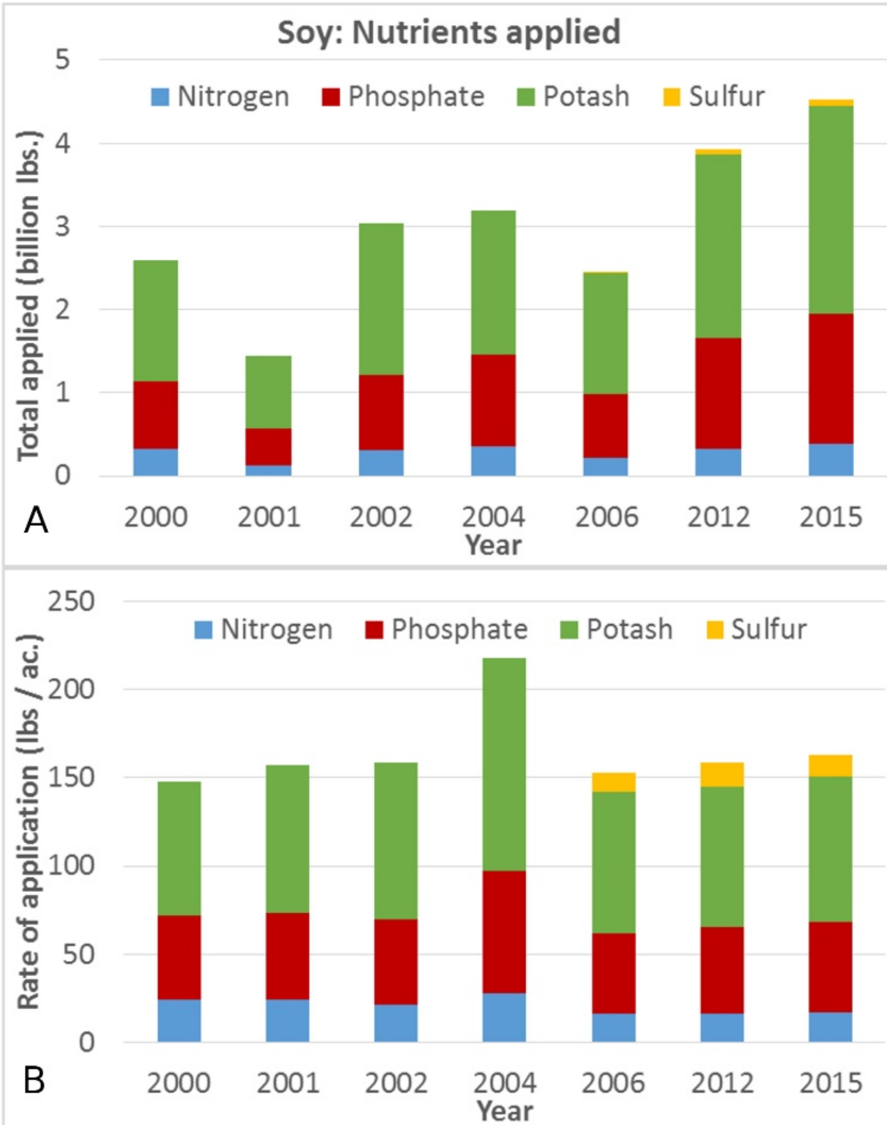

Figure 9 Total volumes of fertilizers used for U.S. soybean production (A) and rates of application (B), 2000-2015.

2.2.5 Pesticides Applied

Pesticide usage data is relevant for understanding potential risks of environmental impacts, including water quality and biodiversity, and to a lesser extent the estimation of water usage. Pesticides for biofuel feedstocks (corn and soybeans) include herbicides, fungicides, insecticides, and nematicides. The pesticides are applied as foliar sprays, direct applications to soil (pre-plant or post-plant), seed treatments, or in the case of Bt,[15] incorporated through genetic engineering. Pesticides are also used on stored grains, including fumigants and insecticides applied to grain bins; such applications are not likely to have the same potential for environmental impacts as other approaches.

Yearly data on pesticide usage in corn and soybeans since 2007 are not available for all pesticides. For those pesticides for which data are available, the estimated usage for production of corn and soybeans varies among different sources (see Table 1 for example). All these data sources, however, documented the increasing trend in usage of the herbicide glyphosate over the past 10 years (Benbrook 2016). Based on data from the USDA's Agricultural Chemical Use Program, a total of 82.3 million pounds of glyphosate (in different forms) was applied to corn during 2016, and 106.9 million pounds to soybeans during 2015. For insecticides, the usage of neonicotinoids, which are applied as seed treatments, is not captured by USDA (Douglas et al. 2015b). Approximately 90% of corn (Douglas et al. 2015b) and 30% of soybean fields planted during 2008-2012 contained neonicotinoid seed treatments.[16]

Table 1. Estimates of total applied glyphosate and atrazine (in million pounds) by different data sources.

Herbicide	Crop	USDA Agricultural Chemical Use Survey[a]	USGS Pesticide National Synthesis Project[b]	Benbrook (2016)
Glyphosate (different forms)	Corn (2014)	61.4	92.2	68.9
	Soybeans (2012)	109.3	115.9	113.9
Atrazine	Corn (2014)	45.2	61.3	NA

a. USDA National Agricultural Statistics Service: Agricultural Chemical Use – Corn 2016. https://quickstats.nass.usda.gov/.
b. USGS National Water Quality Assessment Program: Pesticide National Synthesis Project. State-level pesticide use estimates by major crop and crop groups. https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/.

[15] Bacillus thuringiensis (Bt) is a naturally occurring soil bacterium that produces proteins active against certain insects. Beginning in the mid-1990s, crop plants expressing Bt genes were commercialized in the United States.

[16] Benefits of Neonicotinoid Seed Treatments to Soybean Production (2014). U.S. EPA memorandum, Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/production/files/2014-10/documents/benefits_of_neonicotinoid_seed_treatments_to_soybean_production_2.pdf.

During 2007-2011 neonicotinoid usage increased from 1.2 million to 2.1 million pounds for corn, and 0.26 million to 1.3 million pounds for soybeans (Douglas et al. 2015b).

Through genetic engineering, herbicide-tolerant (HT) corn and soybeans were developed to survive application of specific herbicides targeting weeds. Similarly, insect-resistant Bt corn containing the gene from the soil bacterium *Bacillus thuringiensis* expresses insecticidal Cry proteins. During 2007-2016, the percentage of planted acres of genetically engineered corn steadily increased from 73% to 92% for Bt only, HT only, and "stacked" Bt/HT varieties ("stacked" varieties have both types of traits, and in some cases, multiple Bt and HT traits). The percentage of planted acres of corn with HT or stacked traits increased from 52% to 89%, and from 49% to 79% for Bt varieties, including stacked traits. The percentage of planted HT soybean varieties has remained around 94% since 2007.[17] Reports indicate that herbicide usage in HT corn and soybean increased relative to non-HT, whereas less insecticide (in kg/ha) was applied in Bt corn relative to non-Bt corn (NAS 2016; Perry et al. 2016).

2.2.6 Conservation Practices

Agricultural conservation practices can reduce the impacts of feedstock production and appear to be increasing in prevalence. USDA data show increased use of conservation buffers on 6.2% of planted corn acres in 2001 to 11% in 2010, although soil erosion controls remained roughly the same, with 17.8% of planted corn acres using such controls in 2001 and 18.0% in 2010. Precision agriculture[18] and variable rate technology (VRT), both of which can improve the efficiency of chemical treatments, increased considerably over the same time period. Precision agriculture was applied on 37% of planted corn acres in 2001 and on 72% in 2010. Likewise, VRT for fertilizer application increased from use on 8% of planted corn acres in 2001 to 19% in 2010.[19] The most recent USDA data on these practices is from 2010, so it is uncertain how the extent of these practices has changed since then.

[17] U.S. Department of Agriculture: Adoption of genetically engineered crops in the United States, by trait, 2000-2017 https://www.ers.usda.gov/webdocs/charts/55237/biotechcorn_d.html?v=42565

[18] Precision agriculture is a set of technologies, methods, and information that are applied at a local scale to improve production efficiencies related to outcomes including yield, application of chemical treatments, and irrigation. See USDA's 2007 publication, Precision Agriculture: NRCS Support for Emerging Technologies, https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1043474.pdf

[19] U.S. Department of Agriculture, Economic Research Service: Agricultural Resource Management Survey Farm Financial and Crop Production Practices, https://www.ers.usda.gov/data-products/arms-farm-financial-and-crop-production-practices/.

2.3 Technologies

2.3.1 Biofuel Conversion Technologies

The primary technologies used to produce biofuels remain much the same as in 2011: (1) the fermentation of corn starch to produce ethanol; and (2) the conversion of virgin vegetable and other biogenic oils (including waste fats, oils, and greases) to produce fatty acid methyl esters (biodiesel). Since 2011, the use of feedstocks other than virgin vegetable oils, such as waste oils and corn oil produced at ethanol production facilities, has increased significantly. The production of renewable diesel, produced by hydrotreating vegetable or other biogenic oils, has increased since 2011, with several new large-scale production facilities coming online. In 2014, EPA determined that compressed natural gas (CNG) and liquefied natural gas (LNG) derived from biogas qualified as a cellulosic biofuel.[20] Since this determination, the use of CNG/LNG derived from biogas as transportation fuel has increased significantly, and these fuels now account for the majority of the cellulosic biofuel in EPA's RFS program.[21]

Significant investments have been made by government, universities, and private parties to develop the technologies necessary to economically convert cellulosic biomass to transportation fuel at commercial scale, including both ethanol and hydrocarbon fuels. Several large-scale cellulosic ethanol plants have been constructed, although production from these facilities remains very limited.[22] Recently new technologies have been developed that enable the conversion of the cellulosic portions of the corn kernel (corn kernel fiber) to ethanol at existing corn ethanol production facilities. While the expected production volume of cellulosic ethanol from corn kernel fiber at any individual facility is relatively small (generally less than 5% of the volume of ethanol produced from starch), if widely adopted, this technology could be used to produce significant volumes of cellulosic ethanol. Other technologies being developed to convert cellulosic biomass to hydrocarbon fuels have faced, and continue to face, challenges associated with relatively high capital costs of these facilities coupled with other market and

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[20] 79 FR 42128

[21] Production of cellulosic biofuel RINs for CNG/LNG derived from biogas increased from approximately 32.6 million RINs in 2014 to 188.6 million RINs in 2016. In 2016 CNG/LNG derived from biogas accounted for 97.8% of all cellulosic RINs (D3 and D7) generated. All data are from EPA's public website: https://www.epa.gov/fuels-registration-reporting-and-compliance-help/public-data-renewable-fuel-standard

[22] While the production of cellulosic biofuel has increased significantly in recent years, the vast majority of this fuel has been CNG/LNG derived from biogas. According to EPA data, total production of cellulosic ethanol in 2017 was approximately 10 million gallons. See https://www.epa.gov/fuels-registration-reporting-and-compliance-help/2017-renewable-fuel-standard-data.

policy uncertainties. Several companies are focusing on the production of bio-crudes (a synthetic liquid produced from cellulosic biomass that generally has a high oxygen content relative to petroleum based crudes) that may be able to be processed in traditional refineries. Such technologies, if successfully commercialized, could require lower capital investment if they are able to be utilized in traditional refineries to upgrade these bio-crudes to finished drop-in transportation fuel. These drop-in fuels could lower the need and cost currently required to deploy new expensive supply and distribution infrastructure, and they also have the added benefit of being compatible with existing engines and vehicles.

2.4 U.S. Land Use Change

2.4.1 Overview

Land use change has been identified as one of the primary drivers of potential environmental impacts from an expanding biofuels industry (EPA 2011). Land use is commonly distinguished from land cover, in that land cover strictly describes the physical cover of the land surface (e.g., grassland), while land use involves human activity and reflects human decisions about how land will be used (e.g., grassland used for grazing livestock) (Campbell 1996; Nickerson et al. 2015). The term land use is often also used to describe *how* the land is used for a particular purpose (e.g., agriculture), which can include many land management practices (e.g., fertilizer application, irrigation). Here we use the term land use change generally to describe changes in any of these processes (i.e., land cover, land use, land management) that can affect how land is used and managed.[23]

Land use affects most environmental end points considered in this report, including runoff from agricultural lands, emissions of criteria air pollutants, and habitat acreage and quality for various plant and animal species. Increased agricultural production can come from two distinct processes,

[23] Several publications and organizations use slightly different terms and acronyms to describe generally similar processes (e.g., LULCC for land use/land cover-change in the Intergovernmental Panel on Climate Change Fifth Assessment Report (Ciais et al. 2013); LULUC for land-use-land-use-change in the EPA's Greenhouse Gas Inventory). The USDA National Resources Inventory defines land cover as "the vegetation or other kind of material that covers the land surface," and land use as "the purpose of human activity on the land; it is usually, but not always, related to land cover." DOE's Billion Ton Study defines land use change as "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" (DOE 2016). Our use of the term land use change is intended to be general to encompass all these processes of how the land is used and the physical cover is affected to meet that use. It is not the purpose of this report to reconcile varying definitions in the literature.

extensification (i.e., the expansion of agricultural land onto previously uncultivated land) and *intensification* (i.e., increased production from the land without an increase in acreage). Intensification often occurs from changes in land management and agronomic practices, including double cropping, irrigation, seed improvements, and changes in fertilizer or other chemical inputs. Although land use change is commonly associated with agricultural extensification and intensification, it can also lead to a reduction in total crop acreage (e.g., through cropland abandonment) or a decrease in management intensity (e.g., through replacement of chemical fertilizer with manure inputs).

A second common distinction of land use change types is direct and indirect land use change. In the context of biofuels, direct land use change is the land use change that occurs to support the cultivation of feedstocks specifically for biofuel production. Indirect land use change occurs from the diversion of crops to the biofuel market, which results in an unmet market demand for agricultural products that then induces land use change to meet that demand (Searchinger et al. 2008; EPA 2010). These terms are often used in the context of greenhouse gas emissions from biofuels production and use but are relevant for all environmental end points.

There are many methods used to assess or forecast changes in land use, including empirical observations (e.g., based on remote sensing or plot sampling), surveys (e.g., the USDA Census of Agriculture), and dynamic models (e.g., FASOM-GHG, FAPRI-CARD, GTAP, POLYSYS[24]). This section focuses on domestic U.S. land use change that has been assessed either through empirical observations or surveys of respondents. These are often conducted using comprehensive land use categories or representative statistical samples, rather than focusing on one particular economic sector or region. For example, the corn and soy trends from the USDA NASS data in section 2.2 describe increases in both of these crops, but without a comprehensive land classification assessment it is impossible to know whether these increases came from existing agricultural lands or new lands that were not recently in cultivation. A strength of this approach is greater confidence in the amounts and types of land use change actually occurring. However, there are still uncertainties and challenges with comparing different empirical observations, including differences in definitions, methods, and scope (Nickerson et al. 2015). A weakness of empirical approaches is the difficulty of confidently attributing the *causes* of land use change. There are many potentially contributing market (e.g., crop prices, transportation costs)

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[24] FASOM-GHG is the Forest and Agricultural Sector Optimization Model Greenhouse Gas Version developed by Texas A&M and Oregon State University. FAPRI-CARD is the agricultural model developed by the Food and Agricultural Policy Research Institute (FAPRI) and the Center for Agricultural and Rural Development (CARD). GTAP refers to the international trade model developed by the Global Trade Analysis Project at Purdue University. POLYSYS is the Policy Analysis System developed by the University of Tennessee and the USDA ERS to simulate the U.S. agricultural sector.

and nonmarket (e.g., climate, pests) factors that influence land use changes that could be coincident with the passage of EISA and therefore correlated in an empirical analysis. Attributing all of the observed trends to biofuels is not appropriate, and thus methods of causal analysis for biofuels has emerged as an active area of research [e.g., Efroymson et al. (2016)]. Some empirical studies have attempted to assess attribution to biofuels through proxy data, such as the proximity to a biorefinery (Brown et al. 2014; Motamed et al. 2016; Wright et al. 2017) or surveys of farmers (Wallander et al. 2011; Gray et al. 2013). Challenges with quantifying attribution are summarized in Box 3.

Dynamic agricultural models are simplified representations of complex agronomic, economic, social, and biophysical systems. These often include a reference scenario (e.g., without the RFS) and a focal scenario (e.g., with the RFS) to isolate the simulated effect of a given policy (Koponen et al. 2018). A strength of this approach is the ability to look at "what if scenarios" to isolate the effects from a given policy or scenario. Weaknesses of this approach include: (1) many models make different projections about the same subject; (2) it is difficult to objectively assess model skill at projecting future unobservable states; and (3) because the models are simplifications of real systems, they often lack many details known to influence the system. As an example for biofuels, in 2011 USDA summarized six major modeling efforts available at the time (Wallander et al. 2011) and found wide ranges in predicted increases in cropland (0.7-8.1 million acres) and corn acres (1.8-19.4 million acres) (see Table 2). These limitations are not unique to modeling biofuels or agricultural systems more broadly and are addressed in other areas of study (e.g., climate change research) by using averages or other statistics derived from multiple models, using historical data to assess and improve on performance, or other measures. These approaches are still in their relative infancy in the area of biofuel simulation modeling. Nevertheless, dynamic models are useful tools for assessing system behavior to better understand sensitivities to changes in key parameters and scenarios of interest.

Section 204 of EISA requires that the triennial report assess "impacts to date and likely future impacts." For impacts to date we rely on the empirical record. For likely future impacts, we assume that the trends to date are a reasonable estimate of trends over the short-term future (e.g., less than 3 to 5 years or the interval between Section 204 Reports). As discussed in the introduction, we minimize our discussion of likely impacts further into the future because of inherent uncertainties in such projections, but we include some examples for illustrations.

There is a large body of research that uses dynamic models mentioned above and other methods to assess potential future impacts from bioenergy and biofuel production [e.g., (Souza et al. (2015); Dale et al. (2016); Emery et al. (2016); DOE (2017))]. It is important to note the distinction between a "likely future" as prescribed in Section 204 of EISA, and a "potential future." We interpret likely futures to encompass future effects under conditions of current policy and market dynamics. Potential futures are much more broad and may implicitly assume potential changes in current policy, technological advances not yet observed, changes in land ownership and decision making processes, and/or market dynamics that do not reflect current conditions. Such research, although less relevant for assessing "likely futures" under the requirements of Section 204, are still valuable tools for understanding the complex agro-economic system and are helpful for decision makers who are designing public policy. One recent notable example is the 2016 DOE Billion Ton Study (DOE 2016; DOE 2017), which highlighted these limitations in the disclaimer[25], and used POLYSYS to generate several potential future scenarios of land

Table 2 Comparison of different simulation studies summarized in Wallander et al. 2011 (source material from Searchinger et al. 2008, Malcolm et al. 2009, and EPA 2010).

Study	Searchinger et al. (2008)	Searchinger et al. (2008)	Malcolm et al. (2009)	EPA (2010) RFS2 RIA (FASOM)	EPA (2010) RFS2 RIA (FAPRI-CARD)
Year modeled	2016/2017	2016/2017	2015	2022	2022
			Billion gallons		
Increase in ethanol	14.77 (from 14.75 to 29.52)	8.08 (from 14.75 to 22.84)	1.7 (from 13.30 to 15.00)	2.7 corn-based (from 12.3 to 15.00) plus 13.5 cellulosic	2.7 corn-based (from 12.3 to 15.00) plus small change in imported ethanol
			Predicted change in land-use/cropping selection Million acres		
Predicted increase in corn acres	19.4	10.0	3.2	3.6	1.8
Predicted increase in cropland	5.5	2.9	4.9	8.1	0.7
Other major predicted increases			Soybeans (1.9)	Switchgrass (12.5)	
Major predicted decreases	Soybeans (-9.6) Wheat (-4.8)	Soybeans (-4.1) Wheat (-3.3)	Rice and sorghum (each -0.1)	Wheat (-2.9) Soybeans (-1.4) Barley (-1.2) Rice and hay (each -0.8) Oats and cotton (each -0.2)	Soybeans (-0.7)

25 The disclaimer in volume 2 of the 2016 Billion-Ton Report states: "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 benefits, challenges, and research needs related to increasing biomass production. Users should refer to the chapters and associated information on the Bioenergy Knowledge Discovery Framework (bioenergykdf.net/billionton) to understand the assumptions and uncertainties of the analyses presented." (DOE 2017)

use change and biomass/biofuel production based on assumptions of minimal cropland extensification,[26] which were then run through a suite of models or other approaches to assess environmental effects to air, land, and water resources.

The following sections detail the land use change observed to date, briefly discuss future projections of land use change, and the conclusions of this work. We focus on extensification because bringing new lands into cultivation can have a larger environmental impact per unit area than intensification (EPA 2011). However, some forms of intensification can also have significant effects and will be discussed. Other land use changes focused on land management operations (e.g., tillage practices, tile drainage, fertilizer) are addressed elsewhere in this report. [27]

2.4.2 Observed Land Use Change to Date

The 2011 Report found that quantifiable land use change had not been reported as of the Report's publication and thus reviewed potential environmental impacts of different land use change scenarios common in the literature (EPA 2011). It concluded that land use change would likely drive most environmental effects aside from air quality and that the most plausible land use change scenario for corn and soybeans was for conventionally managed corn to replace no-till soybean or other row crops and soybeans to maintain a stable acreage (EPA 2011).

2.4.2.1 National trends in major land uses and cropland extensification

Since the 2011 Report there have been many important studies on land use change trends in the U.S. Five major national efforts have been published: (1) the USDA's Major Uses of Land in the United States, 2012 (termed "Major Land Uses" series, MLU) (Bigelow et al. 2017); (2) the USDA 2012 Census of Agriculture ("Census") (USDA 2014); (3) the USDA 2012 National Resources Inventory (NRI) (USDA 2015); (4) the USGS U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends (NWALT), 1974–2012 (Falcone 2015); and (5) a pair of studies from the University of Wisconsin (Lark et al. 2015) and the University of Minnesota (Wright et al. 2017). As mentioned above, these efforts vary

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[26] The DOE's 2016 Billion Ton Report "explicitly assumes no changes in the size for each major land class (forest, agriculture, etc.) and also keeps both plantation and natural commercial forest areas fixed; there are minimal changes in management on pasture and cropland within the agricultural land base, and other projected demands for goods and services are met in addition to biomass for energy to produce over one billion dry tons of biomass with minimum environmental effects by 2040" (DOE 2016).

[27] Trends in fertilizer and chemical inputs are discussed in section 2.2.4 and 2.2.5, respectively, while trends in tillage are discussed briefly in section 3.5.

in their definitions, scope, and approach, influencing their comparability. We clarify these differences below and provide key definitions of terms in Appendix B. There have also been several regional studies documenting land use change in different parts of the country, including the Prairie Pothole Region (Johnston 2013; Johnston 2014; Reitsma et al. 2016), around the Great Lakes (Mladenoff et al. 2016), for the western corn belt (Shao et al. 2016), for lands in the Conservations Reserve Program (CRP) (Morefield et al. 2016), and for corn/soybean farms (Wallander et al. 2011). We focus primarily on the major national efforts covering many crops and land uses but mention the more specific studies where appropriate.

The USDA Major Land Uses (MLU) report (Bigelow et al. 2017) is produced by the USDA's Economic Research Service (ERS) and is one of the most comprehensive land use assessments available in the United States. The MLU is constructed using information from several sources, including USDA (Census, ERS, NASS, NRI), the US Census Bureau, and the U.S. Forest Service (USFS, Forest Inventory and Analysis, or FIA). The MLU has been produced since 1949 and reports on five-year intervals coincident with the Census. Total cropland as defined in the MLU has five components. The first three (harvested cropland, failed crops, and summer fallow crops) make up "cropland used for crops" and describe the acreage devoted to crop production. The last two (cropland pasture and idle cropland) are not directly used for crop production in a given year but may rotate into production (see Appendix B for full definitions). The MLU includes set-asides such as the Conservation Reserve Program, the Acreage Reduction Program, and other Federal acreage-reduction programs into the "Idle" category. The MLU was released in August of 2017 and was not available at the time of the External Review Draft of this Report (i.e., after the May 2017 cutoff), thus the ERD was updated with information from the MLU for the Final Report.

The MLU found that total cropland decreased by 16 million acres between 2007 and 2012, continuing a long decline in total cropland that began in the 1970s. We focus on changes since 2007 because of the intended focus of this Report, and we refer readers to the MLU and elsewhere for longer term discussions of land use in the U.S. The decrease in total cropland between 2007 and 2012 reported in the MLU came primarily from a 23-million-acre decline in cropland pasture offsetting a 5-million-acre increase in cropland used for crops (see Table 3). Thus, land devoted to crop production increased by 5 million acres between 2007 and 2012. The increase in cropland used for crops was mostly in the Northern Plains region (ND, SD, NB, KS) and was from corn and soybean increases, with the largest decreases from hay (see Figure 10 and Figure 11). The MLU explicitly noted biofuels as a potential

Table 3. Major land uses (in millions of acres) from the MLU (Bigelow et al. 2017).

Land Use	1945	1949	1959	1964	1969	1974	1978	1982	1987	1992	1997	2002	2007	2012
	Million acres													
Cropland	451	478	458	444	472	465	471	469	464	460	455	442	408	392
Cropland used for crops	363	383	359	335	333	361	369	383	331	338	349	340	335	340
Idle cropland	40	26	34	52	51	21	26	21	68	56	39	40	37	39
Cropland pasture	47	69	66	57	88	83	76	65	65	67	68	62	36	13
Grassland pasture and range	659	632	633	640	604	598	587	597	591	591	580	587	614	655
Forest-use land	602	760	728	732	723	718	703	655	648	648	642	651	671	632
Grazed forest-use land	345	320	245	225	198	179	172	158	155	145	140	134	127	130
Other forest-use land	257	440	483	507	525	539	531	497	493	503	501	517	544	502
Special-use areas	85	87	123	144	141	147	158	270	279	281	286	297	313	316
Urban areas	15	18	27	29	31	35	45	50	57	59	66	60	61	70
Miscellaneous other land	93	298	293	277	291	301	301	224	227	224	236	228	197	196
Total land area	1305	2273	2271	2266	2264	2254	2254	2265	2265	2263	2263	2264	2264	2260

contributing source for the reported land use changes.[28] The MLU attributed the large decrease in cropland pasture as largely attributable to a methodological shift in the Census that occurred in 2007 and 2012.[29] Because of the coincidence of the methodological change and the passage of EISA, it is not possible with this dataset to attribute changes in cropland pasture as reported in the Census to any one

[28] The MLU states: "Another trend that has affected U.S. crop plantings over the past 30 years is the use of crops as a biofuel input source. Over the past decade, the use of corn for biofuel increased sharply due to the mandate in the Energy Policy Act of 2005 to increase the amount of renewable fuels in the U.S. fuel supply. This law, coupled with an expansion of required amounts of renewable fuels in 2007, boosted production of corn ethanol."

[29] From the MLU: "Cropland pasture estimates, one of two nonpermanent grazing uses tracked in MLU, declined nearly 80 percent in the past 10 years (2002-12) after exhibiting relative stability for more than 50 years. This decline is largely attributable to methodological changes [*i.e., change in wording and location of the question in the Census, emphasis added*] in the collection of cropland pasture data in the Census of Agriculture, the data source of the cropland pasture category…While there is no way to definitively determine the extent of the effects of changes in the placement and wording of the cropland pasture question, it seems likely, given the relatively stable cropland pasture acreage trend from 1949 to 2002, that the changes contributed to the large decrease between 2002 and 2012." (Bigelow et al. 2017). The changes are described further in Bigelow et al. (2017).

source. The MLU also reported a large increase in grassland between 2007 and 2012 (+41 million acres). However, this also was attributed to methodological changes in the Census and the USFS Forest Inventory and Analysis (FIA) causing a corresponding decrease in forest over the same period (-39 million acres).[30] It is important to consider these methodological changes through time, as trends can be mischaracterized if taken out of context. Historical estimates from the MLU are not updated as methods change, making trends analysis difficult to conduct if based solely on this resource. It is also important to note that the MLU reports net changes in agricultural land use at the county scale, making it impossible to track conversion of land from one cover/use type to another at the field scale. Only a dataset that explicitly tracks land use change for individual land units [e.g., (Lark et al. (2015); USDA (2015); Wright et al. (2017))] can quantify the amount and type of land use conversion across the U.S.

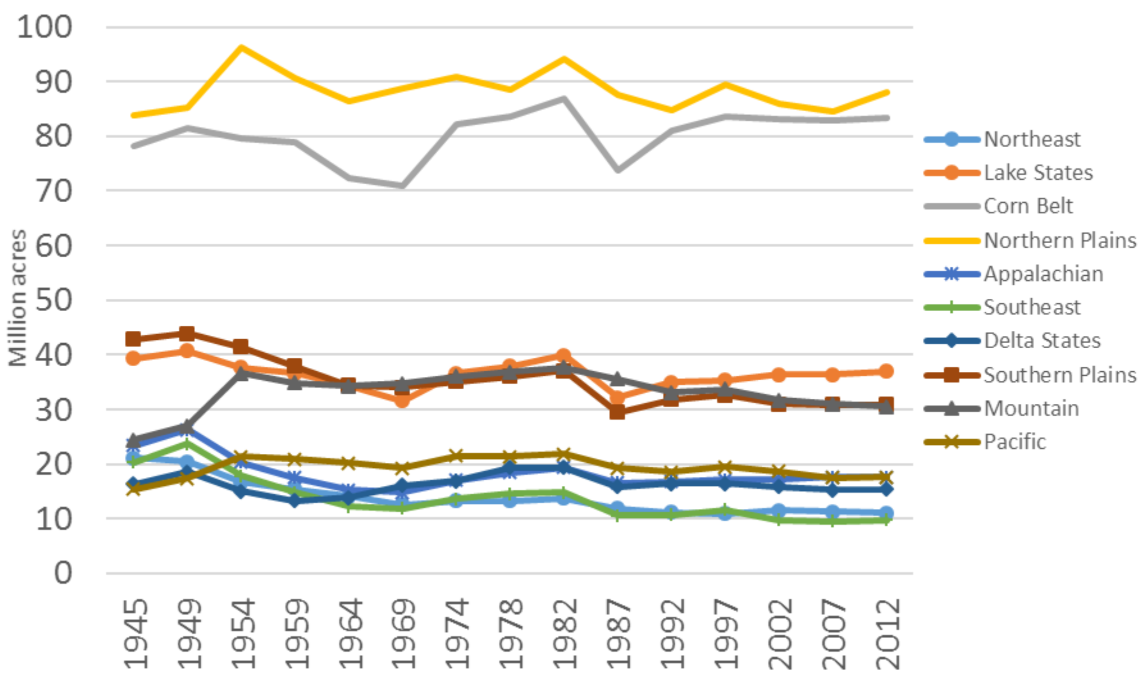


Figure 10. Changes through time (1945-2012) in cropland used for crops by MLU region (Bigelow et al. 2017).

[30] The MLU attributed the increase in grassland to a methodological change in the USFS FIA and the Census. For the FIA change, large areas of chaparral and shrubland which were originally classified as forests because of the presence of tree cover, were reclassified as woodland or grasslands because the relatively sparse tree cover meant the lands were more likely *used* as grassland and rangeland than for timber production (Bigelow et al. 2017). Changes to the Census that likely contributed to increases in grassland are from the same change to the cropland pasture question described in footnote 26.

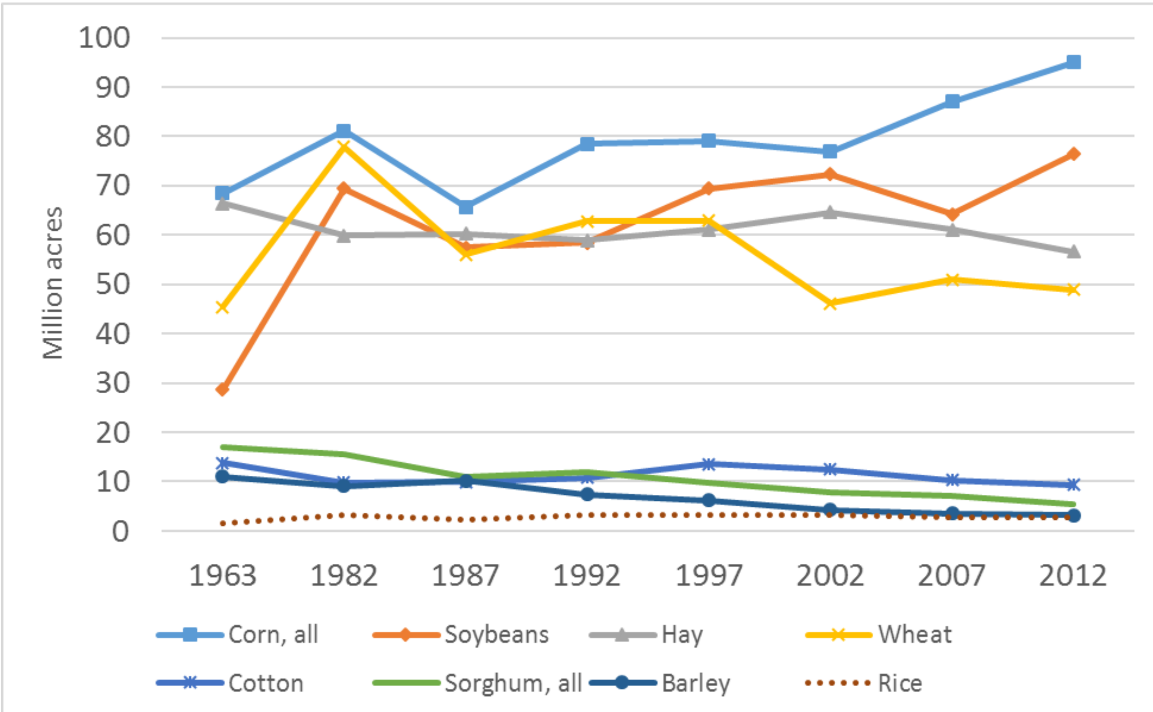


Figure 11. Changes through time (1963-2012) in principal crops harvested for the 48 contiguous States (Bigelow et al. 2017).

Data from the Census of Agriculture is used directly but not without adjustment by the MLU, thus it is not surprising that the 2012 Census reported total cropland decreasing by a similar amount over the same period (16 million acres from 2007-2012) (USDA 2014). However, as with the MLU, the Census includes different types of land that are managed quite differently in their definition of total cropland. Total cropland in the Census includes: (1) harvested cropland; (2) other pasture and grazing land that could have been used for crops without additional improvements[31]; and (3) other cropland (which includes three subcategories: cropland on which all crops failed or were abandoned, cropland in cultivated summer fallow, and cropland idle or used for cover crops or soil improvement but not harvested and not pastured or grazed). Thus, according to the definitions, on an annual basis potential cropland is *used* more like a pasture than a field of row crops.

Examining the individual land types that make up total cropland in the Census demonstrates an increase in harvested cropland from 2007 to 2012 by 5.4 million acres and a large decrease in potential cropland by 23 million acres (USDA 2014), similar to numbers reported in the MLU. Other cropland did

[31] For convenience, we use "potential cropland" for the Census category "other pasture and grazing land that could have been used for crops without additional improvements" due to its length.

not change much in aggregate from 2007-2012 (Rippey 2015).[32] Because the Census and the MLU include lands that are predominantly used as pasture in their definition of *total cropland*, focusing on changes in total cropland can mask conversions from pasture to rowcrops that can have significant environmental effects (EPA 2011). The most comparable term in the Census for croplands used as crops in the MLU is a combination of three terms (harvested cropland, failed/abandoned cropland, summer fallow cropland), which together increased by 7.8 million acres between 2007 and 2012. This discrepancy is likely due to methodological differences between the Census and the MLU.[33] Regardless, it is clear that both sources report an increase in actively managed croplands.

The 2012 USDA National Resources Inventory (NRI) is an independent data source from the Census and the MLU that is produced by the USDA Natural Resources Conservation Service (NRCS). The NRI uses a permanent statistical sampling frame that is used to obtain scientifically credible information on conditions and trends of soil, water, and related resources (USDA 2015). Instead of based on survey responses (e.g., the Census and consequently parts of the MLU), the NRI is a representative statistical sample of all non-Federal lands over a 30-year period (1982-2012 for the most recent NRI). The NRI reviews and revises historical estimates as necessary with each new Report as methods are updated (the Census and MLU do not); thus, changes due to methodology are removed from the NRI so long as historical comparisons are made within the same year's Report. A consistent methodology is a significant advantage when trying to examine trends through time. Thus, different reports have different strengths and weaknesses – estimates of trends may be better assessed with reports such as the NRI where methods through time are internally consistent, whereas estimates of acreages at a point in time are probably better reflected with more comprehensive assessments such as the MLU and NWALT.

The 2015 NRI reported that after a 25-year decrease from 1982 to 2007, total cropland increased by 3.9 million acres between 2007 and 2012 primarily from an increase in cultivated cropland of 4.3 million acres, as shown in Figure 12 (USDA 2015).[34] Uncultivated cropland (e.g., hay) was relatively steady at

[32] Other cropland increased from 61 to 62 million acres mostly from an increase in cropland failed or abandoned (+4 million acres), offsetting decreases in idle cropland (-1.6 million acres) and summer fallow (-1.5 million acres). The increase in failed/abandoned cropland was likely due to the 2012 drought in the Midwest (Rippey et al. 2015).

[33] In the MLU, annual estimates of cropland harvested are based on both Census data and NASS data on principal crops. Annual estimates of crop failure are based on differences in planted and harvested acreage of principal crops from the NASS data series. Annual estimates of cultivated summer fallow historically have been based on fragmentary data from a variety of sources. (Bigelow et al. 2017)

[34] Methodologically, the NRI separates total cropland into two types: cultivated (e.g., rowcrops and land in rotation with rowcrops) and noncultivated (e.g., permanent hay). See Appendix B for full definitions.

52.9 and 52.4 million acres in 2007 and 2012, respectively. The NRI does not include classifications for whether the crop failed or was abandoned. The increase in total cropland in the NRI came primarily from lands formerly in the CRP (50%) and pasture (41%). Net changes in land cover/use between 2007 and 2012 included large decreases in CRP (-8.2 million acres) and increases in total cropland and developed land (+3 million acres) as seen in Figure 13. Morefield et al. (2016) also reported conversion of CRP lands to row crops from 2010 to 2013, with almost 30% of the 1.3 million acres coming out of the program in the Midwest going to five row and grain crops (corn, soy, winter and spring wheat, and sorghum).

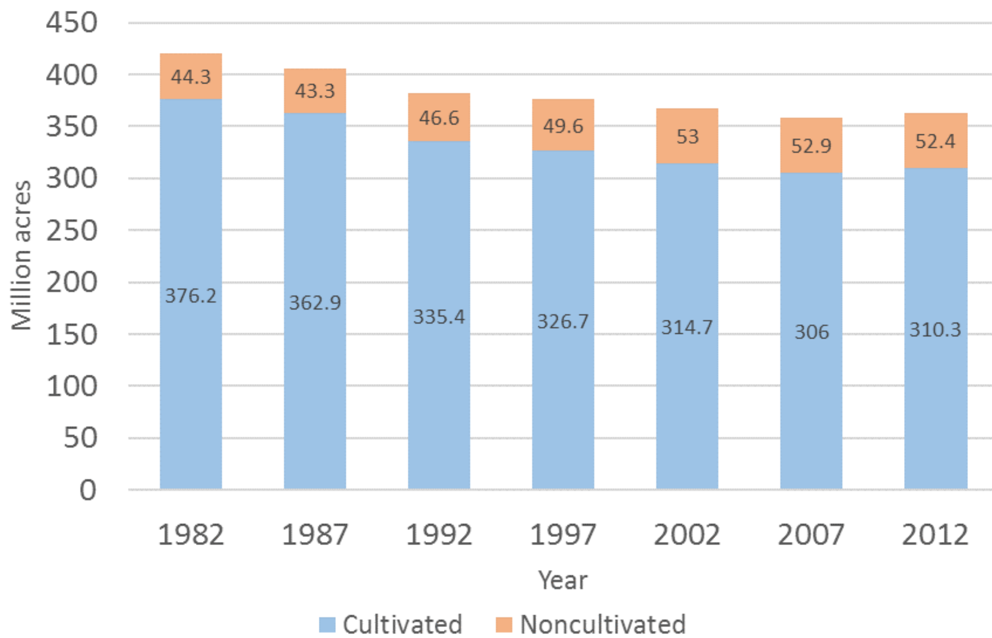


Figure 12. Changes in cultivated, noncultivated, and total cropland from 1982-2012 from the NRI (USDA 2015).

The USGS U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends (NWALT), 1974–2012 report (Falcone 2015) is, along with the MLU, the most comprehensive land use dataset available for assessment of land use trends in the US. The main purpose of the NWALT is to provide a comprehensive land use dataset that is consistent with the high resolution (60-m pixel) USGS National Land Cover Dataset (NLCD) schema and that can be hindcast to the 1970's as part of the USGS National Water Quality Assessment (NAWQA) Program. The 2015 NWALT is primarily based on satellite data from the 2011 NLCD (Jin et al. 2013), but it is supplemented and cross-validated with

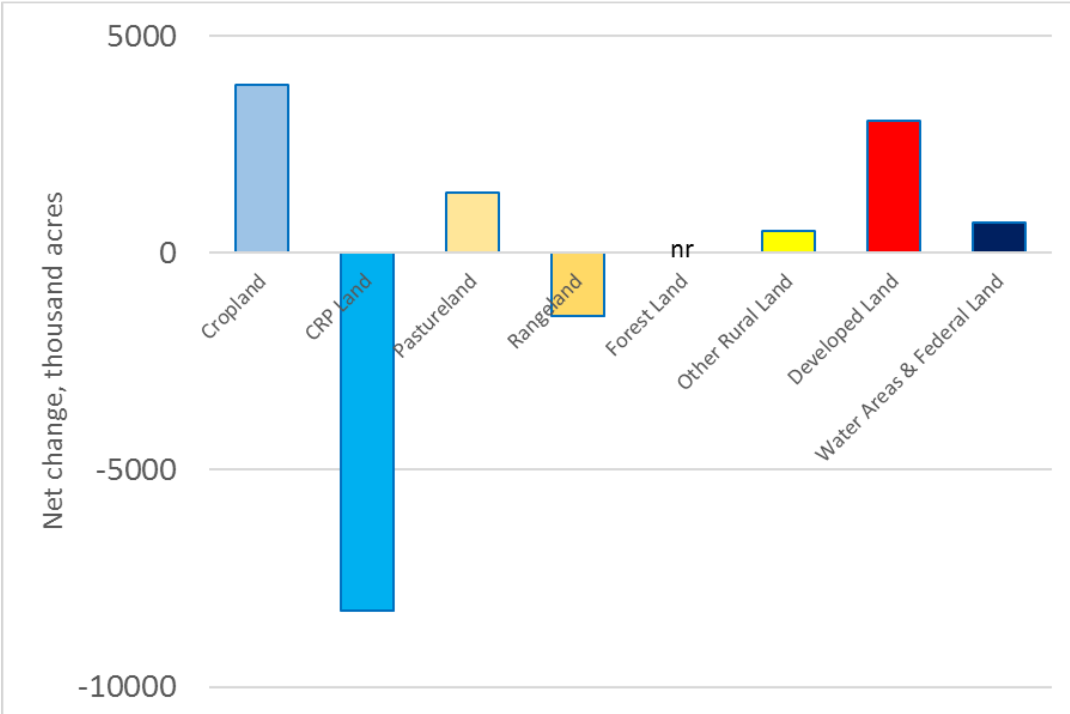


Figure 13. Net changes in all major land cover/use categories between 2007 and 2012 in the NRI (USDA 2015). The net change estimate for forest land is not reliable (nr) as the margin of error is greater than the estimate.

many other datasets [e.g., USDA Census of Agriculture, USDA NRI, USDA CDL, etc.; Falcone (2015)]. Because of the primary basis in the NLCD, changes in agriculture from the NWALT are muted compared to that in the Census.[35] The years covered in the NWALT are 1974, 1982, 1992, 2002, and 2012. Like the NRI, historical estimates in the NWALT are updated as methods change. Agriculture in the NWALT[36], although not directly based on the Census for agricultural data, is adjusted to match changes in total cropland at the county and state level from the Census.

[35] From Falcone (2015): "The NLCD typically shows smaller agriculture changes than would be suggested by the CoA. For example, in the CoA, for any 10-year period, approximately 60 percent of counties had a TC increase or decrease of more than 1 percent of county area, and about 40 percent had more than a 2 percent increase or decrease. For the NLCD 2001–2011, only 12 percent of counties show an Agriculture change of more than 1 percent of county area, and only 3 percent of counties had more than a 2 percent change. The magnitudes of agriculture changes in this product are typically somewhat more than what the NLCD indicates, but less than the CoA."

[36] "Agriculture" in the NWALT is the sum of classes 43 (Production, Crops) and 44 (Production, Pasture/Hay), and these two together are most comparable to the category of total cropland in the Census (Falcone et al. 2015). The NWALT also notes that class 45 (Production, Grazing Potential) is a "'swing'" category that could go into Agriculture or not depending on the user's goals.

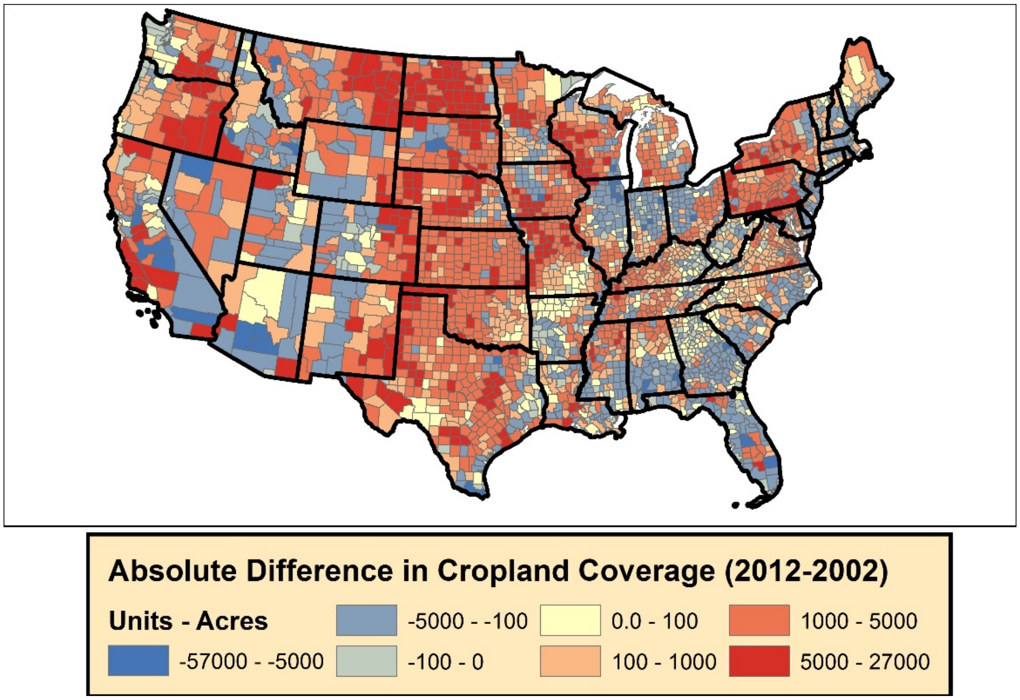
The NWALT reported that between 2002 and 2012, crop area increased by 3.9 million acres, pasture/hay decreased by 5.7 million acres, and developed lands increased by 5.5 million acres. No intermediate estimates are available for 2007. Geographically, the distribution of increased crop area was similar to other studies, with hotspots (i.e., greater than 5,000 acre increase in a county) in the eastern plains from Texas to North Dakota, along with pockets in already agricultural areas of the Midwest and elsewhere (see Figure 14). Decreases in pasture/hay were more universal across much of the country east of the Rockies (see Figure 14).

The NWALT undergoes an extensive quality assurance process, and Falcone (2015) reports general agreement with the trends and magnitudes with other products (e.g., the Census, NRI, etc.). For example, the 2015 NWALT compared the number of counties that gained or lost > 1% total cropland with the Census for all 1992–2002, 1982–1992, and 1974–1982, and found >93% agreement.[37] It is important to note that the NWALT did not perform comparisons with the Census with the 2002-2012 interval for agricultural lands because of "less certainty in the validation data" from the Census owing to the methodological changes that occurred in 2007 as mentioned above[33] (Falcone 2015).

The final national assessment of land use change since 2007 was a pair of studies led by researchers at the University of Wisconsin (Lark et al. 2015) and the University of Minnesota (Wright et al. 2017). They used the 2012 USDA Cropland Datalayer (CDL) along with several other datasets to assess land use change from 2008 to 2012. The CDL is a satellite-derived land cover data product (30-m resolution) produced by the USDA's NASS based on several satellite retrievals (MODIS, IRS-P6 Resourcesat-1, Landsat). Detailed accuracy assessments of the CDL are produced by NASS by comparing crop pixels with ground based samples from the FSA Common Land Unit (CLU) Program and with NLCD for non-crop pixels. This is an important distinction, because although comparison with the FSA CLU is considered very robust, there is no robust "noncrop" national datalayer with which to compare, with the NLCD as a reasonable substitute. Nevertheless, CDL accuracies vary by state and crop, are fairly high for corn and soy (>90%), and are lower for grassland (<50%) (Reitsma et al. 2016).[38] The Lark et al. (2015) and Wright et al. (2017) studies differ from many others that use the CDL, in that they went through an extensive screening process to make sure the lands they identified as

[37] The NWALT reported good agreement with the Census for counties that lost >1 percent total cropland (3,058 of 3,108 correct, 98.4 percent), and for counties that gained >1 percent total cropland (2,434 of 2,593 correct, 93.9 percent). This is overall for all time periods except 2002-2012. It is difficult to interpret the meaning of this agreement since the NWALT is partially calibrated with Census data.

[38] See also USDA National Agricultural Statistics Service, CropScape and Cropland Data Layer – Metadata at https://www.nass.usda.gov/Research_and_Science/Cropland/metadata/meta.php.



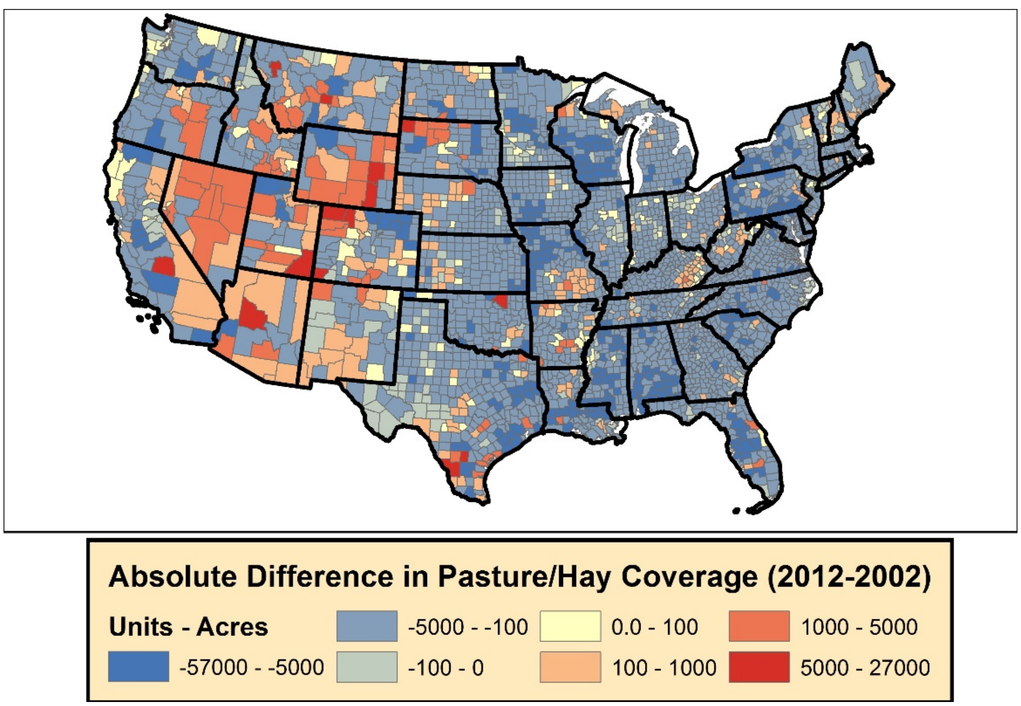


Figure 14 Difference in cropland (class 43, top) and pasture/hay (class 44, bottom) between 2012 and 2002 by county from the NWALT (Falcone 2015).

converted from non-crop to crop had no evidence of cultivation for 20 years or more.[39] Thus, their focus was on one-time conversion between 2008 and 2012 of areas with no evidence of cultivation (termed "conversion", and vice versa, termed "abandonment") and does not include intermittent pasture/cropland rotations as conversions.

Lark et al. (2015) found that total cropland from 2008 to 2012 increased nationally by 3.0 million acres, with gross land conversion[40] nearly four times greater than net land conversion. Grasslands made up the bulk of the source of land converted to new croplands (77%), with much lower percentages from shrublands (8%), idle (8%), forest (3%), wetlands (2%), or other land covers (2%). Roughly 50% of this expansion occurred on marginal lands as defined by the USDA's Natural Resource Conservation Service and an additional 15% on lands deemed unsuitable for agriculture.[41] The first crop planted on converted land was dominated by corn (27%), wheat (25%), soybeans (20%), and then alfalfa (7%).

The follow up study by Wright et al. (2017) focused on land use changes within 25, 50, 75, and 100 miles of the nearest biorefinery in order to try and isolate land use changes that may be attributable to biofuels. Furthermore, Wright et al. (2017) improved on the methodology in Lark et al. (2015) by including a validation step using aerial photography from the USDA's National Aerial Imagery Program and found that the estimates of net conversion in Lark et al. (2015) were biased low because of overestimation of agricultural abandonment. Instead of 3.0 million acres nationally converted from the

[39] Lark et al. (2015) and Wright et al. (2017) first combined the many individual CDL land cover types into "superclasses" (i.e., cropland and non-cropland) because although some individual crop accuracies may be low, the accuracy of aggregated classes is higher. Second, they removed any lands that "flip-flopped" between crop and non-crop within the period of the CDL they were examining (2008-2012). Thus, remaining lands either did not change superclass at all, or they changed once and remained in the new superclass (e.g., non-cropland to cropland, or vice versa). Third, they compared converted pixels with the NLCD from 1992, 2001 and 2006 to make sure that none of the lands identified as converted between 2008 and 2012 had been agricultural in any of those prior three years. Fourth, they compared their land cover superclasses with plot data from the USGS Land Cover Trends Project that identified areas that had been cultivated or not from 1973-2002. Lark also used NASS metadata for the CDL (https://www.nass.usda.gov/Research_and_Science/Cropland/metadata/meta.php) and determined that the accuracy of their crop and noncrop superclasses was fairly high, ranging from 97.0-97.5% and 79.8-87.2%, respectively, across years from 2008-2012 (Lark et al. 2017).

[40] Total cropland increases (extensification) is the net effect of two processes: gross conversion of land from non-cropland to cropland (expansion) and gross conversion from cropland to non-cropland (abandonment).

[41] Lark et al. (2015) used the USDA's Natural Resource Conservation Service's (NRCS) land capability classification (LCC) system to quantify the quality of converted land as "prime" (LCC 1-2; prime farmland), "marginal" (LCC 3-4; land characterized by severe to very severe limitations), and "unsuitable" (LCC 5-8; land with limitations that restrict use to non-crop purposes).

earlier study, Wright et al. (2017) reported roughly 2.7 and 4.2 million acres of noncropland converted to cropland within a 50- and 100-mile radius of biorefineries, respectively, across the nation. Fifty-miles is commonly cited as the economic “break-even point” for transporting feedstock to a biorefinery (Mueller 2010a; Mueller 2010b). Furthermore, they reported higher *rates* of conversion closer to the biorefineries. The finding of higher *rates* of conversion closer to the biorefineries is important and suggests a causal link, a finding that has been found in other regional studies of Kansas (Brown et al. 2014) and in a nine-state area in the Midwest (Motamed et al. 2016). There were hotspots of conversion all over the country (see Figure 15), but the bulk of the expansion was of soybeans, corn, and wheat in North and South Dakota, of soybeans in the steeper areas of southern Iowa and northern Missouri normally used for grazing, and of wheat in western areas of Kansas, Oklahoma, and Texas over the Ogallala aquifer (see Figure 16). Wright et al. (2017) estimated that expansion within 50 miles of biorefineries could generate roughly 0.37 billion gallons of ethanol per year.

Earlier estimates of cropland extensification from Johnston (2014) and Wright et al. (2013), although received with much attention, have significant limitations. Johnston (2014) used aerial imagery from the 1970s and 1980s to identify wetland areas that had been converted to agriculture by 2010-2011. This land conversion could have occurred long before EISA. Both the Johnston (2014) and Wright et al. (2013) studies (along with several others) used the USDA Crop Data Layer (CDL) without adjustment with the NLCD or other sources [unlike Lark et al. (2015) and Wright et al. (2017)], which can lead to an overestimate of land use change, particularly for grasslands (Dunn et al. 2017). In particular, Dunn et al. (2017) compared land use change estimates for 20 counties in the Prairie Pothole Region using three methods: unadjusted CDL, adjusted CDL [per Wright et al. (2017)], and the NAIP. They also reported that unadjusted CDL data could overestimate land use change, and found, consistent with Wright et al. (2017), that adjustments led to much lower estimates of land use than either unadjusted CDL and the NAIP for almost all counties examined. Nevertheless, these earlier studies qualitatively agree with patterns reported in more recent national studies.

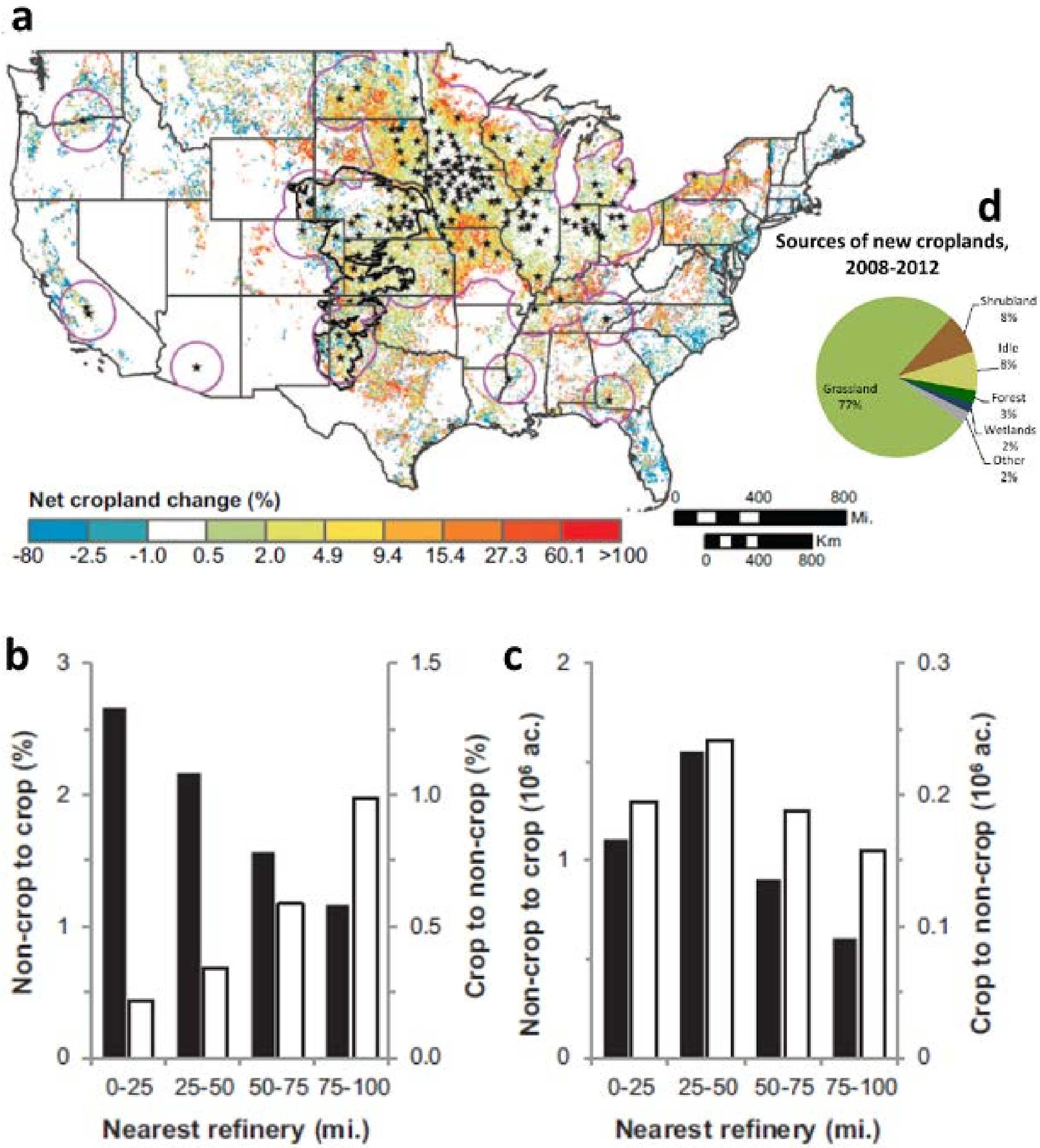


Figure 15 Net rates (a) of land use change from non-cropland to cropland between 2008 and 2012 (purple outlines represent 100 mile radius around biorefineries) from Wright et al. (2017). Positive numbers (green to red) denote net cropland expansion while negative numbers (blue) denote net cropland abandonment. Rates of land use change (b) and total acreage convered (c) from non-cropland to cropland (black bars and left axes) and from cropland to non-cropland (white bars and right axes) for different distances from nearest biorefineries for 2008-2012 [figure modified from Wright et al. (2017)]. Pixel size 3.5 mile. Also shown (d) is the source of new croplands [modified from Lark et al. (2015)]. © 2017 IOP Publishing Ltd for Wright et al. (2017).

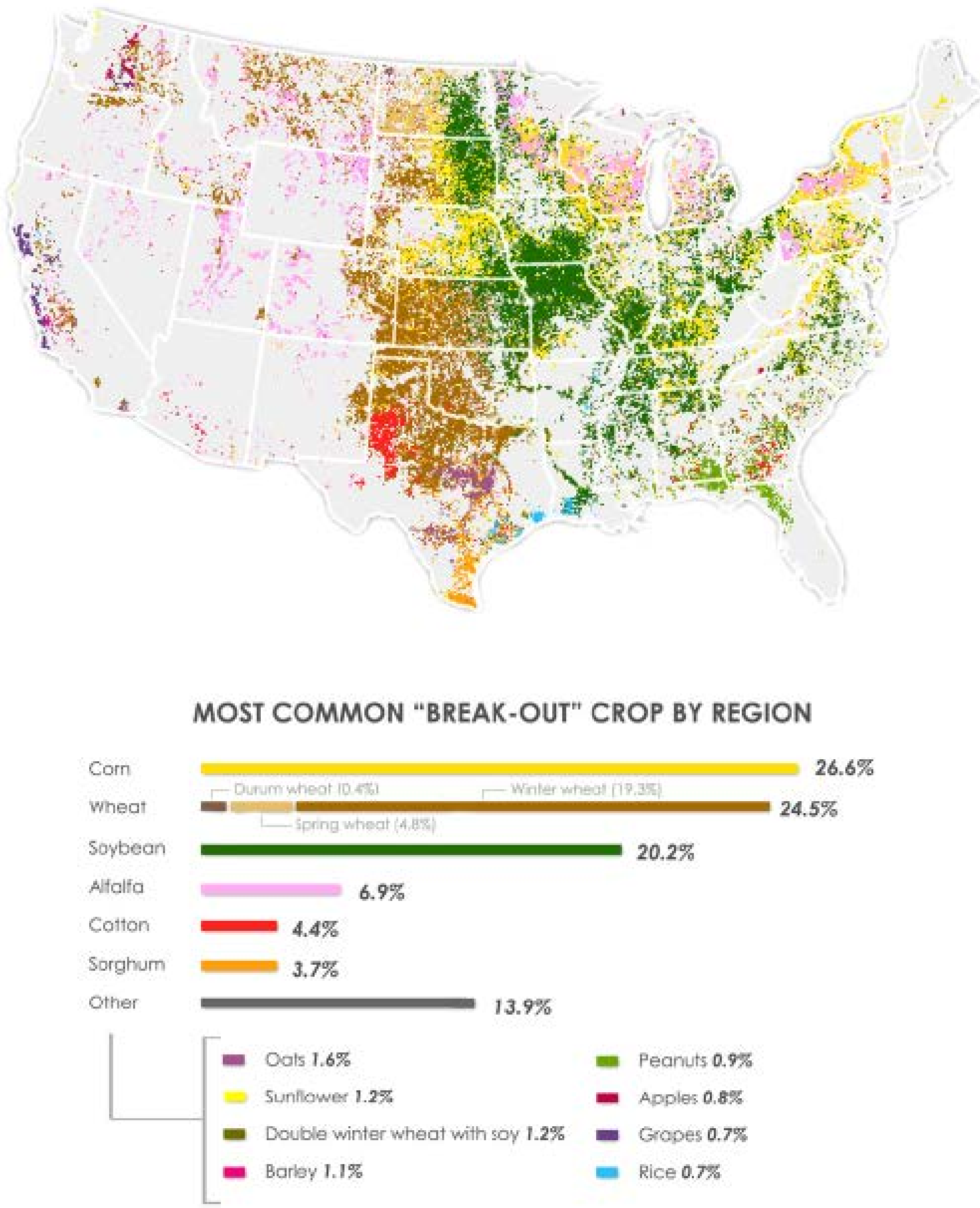


Figure 16 Most common 'break-out' crop by region from Lark et al. (2015). Map represents the most common first crop to be planted after conversion to cropland 2008–2012. Corn and soybeans dominated much of the Midwest and periphery of the Appalachians, while wheat becomes more common moving westward across the plains, with spring wheat in the north and winter wheat in the south. Note that the map depicts only the predominant type of breakout crop grown in an area and does not necessarily reflect the amount of each breakout crop grown there. Nationwide prevalence of each breakout crop is indicated in the legend bar graphs. © 2015 IOP Publishing Ltd for Lark et al. (2015).

Synthesizing all of these major national efforts (see Table 4), there is a consistent signal emerging that demonstrates an increase in actively managed cropland by roughly 4-7.8 million acres, whether from the MLU (+5 million acres, cropland used for crops), the Census (+7.8 million acres of harvested cropland, failed/abandoned crops, and summer fallow crops [42]), the NWALT (+3.9 million acres, production of crops [43]), or Lark et al. (2015) and Wright et al. (2017) (+4.2 million acres within 100 miles of a biorefinery [44]). Comparison of acreage amounts across studies is difficult due to aforementioned differences in definitions and scope, but we have tried to harmonize those to the degree possible. Comparisons of percent changes are more robust because differences among studies are normalized. These also show a consistent increase in actively managed croplands across all studies (1.2-2.4%, Table 4). This increase of actively managed croplands may be coincident with a decrease in total cropland, as lightly managed pasture has either been reclassified into grassland, or converted to actively managed cropland or urban areas. However, the reported decrease in total cropland in the MLU and the Census are not found in the NRI, and both could be from the same methodological change in the Census. The only estimate that is longitudinal and does not suffer from potential methodological changes in the Census is the NRI, which reported comparable estimates of 4.3 million acres. This acreage (4-5 million acres) is a small increase relative to the large agricultural land base,[45] but is a large increase in absolute terms, being almost the size of the land area of New Jersey. These changes are reported to be coming mostly from lands that were formerly in grassland for 20 or more years, and going to corn, soy, and wheat. These trends are likely occurring throughout the country but especially in the Northern Plains, the western margin of the corn belt, and with infilling of the central corn belt. It is unknown whether these trends have continued after approximately 2012.

[42] If one defines "active cropland" as the sum of harvested cropland, summer fallow cropland, and failed cropland, the increase according to the Census was 7.8 million acres between 2007 and 2012. This would exclude "idle cropland" and "potential cropland." Potential cropland has been defined earlier (footnote 30), and idle cropland is defined as "Cropland idle includes any other acreage which could have been used for crops without any additional improvement and which was not reported as cropland harvested, cropland on which all crops failed, cropland in summer fallow, or other pasture or grazing land that could have been used for crops without additional improvements." Idle cropland includes land used for cover crops or soil improvement but not harvested or grazed, land in Federal or State conservation programs, and a few other minor categories.

[43] Note that the NWALT estimates changes between 2002 and 2012, while the other studies estimate changes roughly between 2007 and 2012.

[44] A national estimate that includes the NAIP correction from Wright et al. (2017) has not been published.

[45] The Census estimates roughly 315 million acres of harvested cropland and 380 million acres of total cropland in 2012.

Table 4 Comparison of major national studies on land use change, harmonized to the degree possible. Shown are the source publication, the comparable term(s) and definition(s), years assessed, and the change in acreage in millions of acres (and % from study- specific reference)

Study	Comparable term(s)	Definition	Years reported	Change in million acres (%)
USDA MLU (2017)	Cropland used for crops	Three of the cropland acreage components—cropland harvested, crop failure, and cultivated summer fallow—are collectively termed cropland used for crops, or the land used as an input to crop production.	2007 - 2012	+5 (1.5%)
USDA Census (2017)	Harvested cropland + failed/abandoned + summer fallow	Harvested cropland - This category includes land from which crops were harvested and hay was cut, land used to grow short-rotation woody crops, Christmas trees, and land in orchards, groves, vineyards, berries, nurseries, and greenhouses. No separate definition for failed/abandoned, or summer fallow cropland	2007 - 2012	+7.8 (2.4%) (a)
USDA NRI (2015)	Cultivated cropland	Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops.	2007-2012	+4.3 (1.4%)
USGS NWALT	Production, Crops	Areas used for the production of crops, such as corn, soybeans, wheat, vegetables, or cotton, as well as perennial woody crops such as orchards and vineyards. Includes cultivated crops, row crops, small grains, and fallow fields.	2002 - 2012	3.9 (1.2%)
Lark et al. 2015	Net cropland	Net cropland increases (gross expansion - gross abandonment) of lands in the lower 48 states that have no evidence of cultivation since 1992.	2008-2012	3 (1%) (b)
Wright et al. 2017	Net cropland	Net cropland increases (gross expansion - gross abandonment) of lands within 100 miles of a biorefinery that have no evidence of cultivation since 1992.	2008-2012	4.2 (NA) (c)

a. Harvested cropland, failed/abandoned cropland, and summer fallow cropland changed by +5.4, +4.0, and -1.5 million acres, respectively between 2007 and 2012 according to the Census.
b. Estimates from Lark or Wright are likely to be lower because they focus on a subset of lands that had no evidence of cultivation for 20 years or more, rather than all land. We include these in the table for convenience and completeness.
c. We could not calculate the percent increase from Wright et al. (2017) because the 2008 baseline acreage within 100 miles of a biorefinery was not reported.

2.4.2.2 Trends in intensification: Double cropping and changes in crop plantings and rotations

There has been less information published on trends in U.S. agricultural intensification since the 2011 Report. As mentioned above, intensification can take many forms (e.g., double cropping, changes in fertilizer, chemical inputs, etc.). This section focuses on trends in double cropping and crop plantings and rotations, with other forms covered elsewhere in the report.

In the most recent national study to date on double cropping, Borchers et al. (2014) used NASS data from the June Area Survey (JAS) to assess the prevalence of double cropping across the contiguous U.S. Borchers et al. (2014) use the term double cropping broadly to mean two crops planted (not necessarily harvested) in the same field or two uses of the same crop. Thus, this could include cropland-livestock systems and cover crops, in addition to two crops planted and harvested for the market. Thus, to assess whether JAS-based estimates reflect actual harvesting of multiple crops, JAS-estimates were compared with those from the Census and were found to roughly agree.[46] The authors report that double cropping only occurred on roughly 2% of total cropland for most years between 1999 and 2012 and did not show a consistent trend for any of the seven regions examined. Thus, Borchers et al. (2014) suggest that increased double cropping does not contribute to intensification.

As for changes in crop plantings (e.g., wheat to corn) or changes in crop rotation patterns (e.g., corn-soy-corn to corn-corn-soy), much less has been published to date. Wallander et al. (2011) used the Agricultural Resource Management Survey to focus on land use change for corn and soybean farmers nationally, with an emphasis on 2006-2008. They found that corn acreage increased mostly on farms that previously grew soybeans, but other farms (primarily cotton) offset these shifts by shifting to soybean production (Wallander et al. 2011). However, the short time window for this study that centered on the relatively anomalous year of 2007 (see Figure 4) suggest that different conclusions may be reached if the time window were moved to subsequent years or for a longer period. Thus, whether these short-term changes in crop plantings have been sustained is unclear. More recently, Beckman et al. (2013) reported that increases in corn acreage from 2001-2012 resulted in a net decrease in barley, oats, and sorghum.

Regional studies on changes in crop plantings and rotations focused mainly on the central Midwestern areas that are already highly agricultural. An analysis across a nine-state area in the Midwest reported the area of continuous corn increased by 2.5-5 million acres from 2006-2010, with a smaller decrease in continuous soybean and little change in corn-soy rotations (Plourde et al. 2013). In contrast, a detailed study in eastern Iowa examining changes in corn and soybean rotations found that the most

[46] The Census-based estimate double cropping was 7% higher than the JAS-based estimate.

common rotation between 2002 and 2007 (corn-soy) was absent between 2007 and 2012, with 59% replaced by two or more years of continuous corn and 41% replaced by two or more years of continuous soybean (Figure 17) (Ren et al. 2016). The authors reported that corn tended to be planted on higher quality lands, while soybeans were pushed to lower quality lands. A study in Kansas compared corn extensification and intensification, focusing on land use changes to corn (Brown et al. 2014).[47] They reported that corn intensification far outweighed corn extensification (79% and 16% of corn land use changes, respectively), with more extensification in the arid west where there was already less corn and more intensification in the rest of the state where corn was already grown. As with Wright et al. (2017), Brown et al. (2014) also found an influence of biorefinery proximity, with an 8% increase in conversion to corn from already cropped land, and a 10% increase in extensification, when one moved 1% closer to a biorefinery that was 50 miles away

These region- and state-specific studies are not inconsistent with the national studies. For Kansas, Wright et al. (2017) also reported most of the conversions from non-crop to crop occurred in the west of the state. For Iowa, the areas of extensification in southern Iowa reported in Wright et al. (2017) were not included in the nine-county area of eastern Iowa examined in Ren et al. (2016). Thus, a consistent picture from multiple sources appears to be emerging, with both cropland extensification and crop switching towards more intensively managed crops occurring throughout the country. Intensification appears to be dominating in already agricultural areas, while extensification dominates along the large agricultural margins and within formerly uncultivated areas in the central Midwest.

[47] Brown et al. (2014) define extensification as a conversion from noncropland in 2007 to corn cultivation in 2008 and 2009, and intensification as a conversion from non-corn crop cultivation in 2007 to corn cultivation in 2008 and 2009.

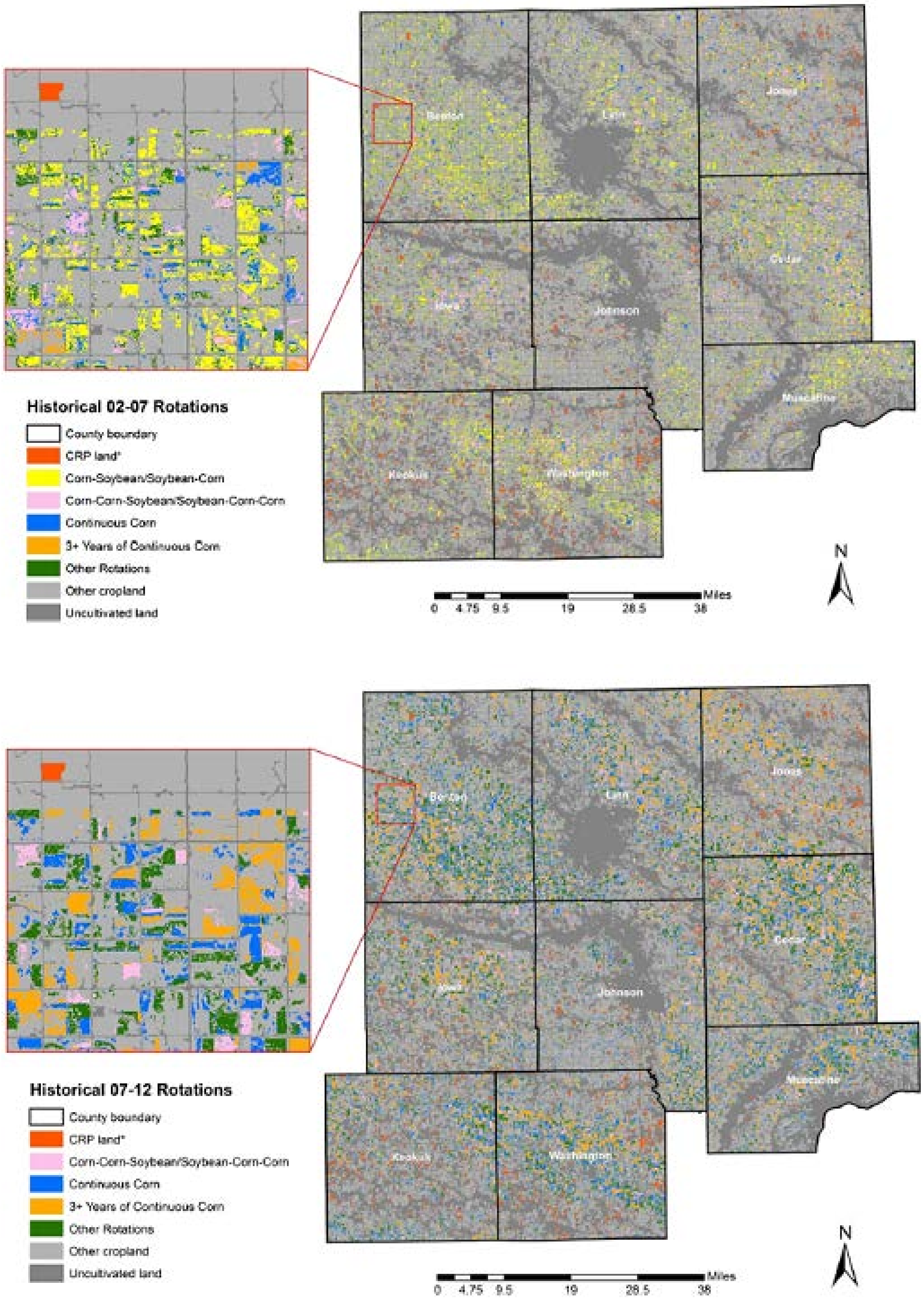


Figure 17 Crop rotation patterns for the 9-country area of eastern Iowa in Ren et al. (2016) for 2002-2007 (a) and 2007-2012 (b), with a blowup in Benton county for clarity. Note the disappearance of the corn-soybean rotation (yellow) between 2002-2007 and 2007-2012.

2.4.3 Economic-Based Projections of U.S. Land Use Change Impacts

The 2011 Report highlighted several studies using agro-economic models to simulate the potential impacts of land use change compared to different future biofuel scenarios (Searchinger et al. 2008; Malcolm et al. 2009; EPA 2010; Tyner et al. 2010). These were summarized by Wallander et al. (2011) and show a wide range of projections (see Table 2). Comparing the predicted changes summarized in Wallander et al. (2011) with the observed changes in sections 2.2 and 2.4,[48] suggest that the observed increase in corn and cropland, by roughly 10 million and 3.9-7.8 million acres respectively, are generally well approximated in the models. However, the observed increase in soybean by roughly 8 million acres was not well represented, and was only directionally consistent in Malcolm et al. (2009), with the other studies projecting decreases in soybean acreage. A comprehensive assessment of the performance of these and other models is beyond the scope of this assessment. Since 2011, multiple studies have continued to evaluate potential land use change impacts associated with increased biofuel use. Estimates of potential land use change impacts from increased biofuel demand continue to vary significantly, particularly when evaluating indirect and international land use change impacts (Dunn et al. 2013; Taheripour et al. 2013b; Macedo et al. 2015; Plevin et al. 2015; Valin et al. 2015). Because many of these studies are global in nature in order to incorporate global commodity trading, they are discussed below in Section 2.5 (International Land Use Change).

Regarding likely future land use changes in the U.S., the USDA reports no major changes in total cropland nor in the eight major crops reported (corn, soybeans, wheat, upland cotton, sorghum, rice, barley, and oats) in their long-term projections to 2026. USDA expects CRP acreage to hold near the maximum levels legislated by the 2014 Farm Bill at 24 million acres (USDA 2017). It is likely that these more recent efforts will have similar difficulty in matching observed changes, because of the complexity of these agro-economic systems as well as the inherent challenges of comparing models with observations.

2.4.4 Conclusions

- Biofuel feedstock production is responsible for some of the observed changes in land used for agriculture, but we cannot quantify with precision the amount of land with increased intensity of cultivation nor confidently estimate the portion of crop land expansion that is due to the market for biofuels.

[48] This exercise is only relevant for Malcolm et al. (2009) which projected to 2015 and Searchinger et al. (2008) which projected to 2016/2017. EPA (2010) projected to 2022.

- Recent research and anticipated updates to data are expected to improve our ability over the next three years to quantify the fraction of land use change attributed to biofuel feedstock production in the U.S.
- Evidence from multiple sources demonstrates an increase in actively managed cropland in the U.S. since the passage of EISA by roughly 4-7.8 million acres, depending upon the source.
- Much of this increase is likely occurring in the western and northern edges of the corn belt with reductions of pasture and grassland, but also through infilling of already agricultural areas.
- Thus, intensification likely dominates in already agricultural areas and extensification dominates in less agricultural areas.
- Research is needed to quantify changes in the intensity of cultivation on existing agricultural land.
- Research is also needed to more effectively connect changes in land use to the environmental impacts of concern.

There are five major national-scale studies that suggest that cropland has increased in total acreage in the U.S. by 4-7.8 million acres between 2007-2008 and 2012. For context, 4.7 million acres is approximately the land area of the state of New Jersey. This has been primarily a conversion of grassland or pasture to corn, soybeans, and wheat, along the extensive agricultural margin, and through infilling of previously uncultivated areas in the central Midwest. There are no national updates since 2012, but several are forthcoming, including the 2017 Census of Agriculture, 2017 MLU, and the 2017 NRI.[49] Thus, it is not known whether these national trends have continued to the present. There is also substantial evidence of crop shifting on existing agricultural lands from other row crops towards more corn and more sequential rotations of corn. The total U.S. acreage experiencing these shifts among croplands is unknown, but regional studies suggest that the magnitude may be larger than that of extensification. There is strong correlational evidence that biofuels are responsible for some of this observed land use change, but exactly how much remains unclear (see Box 3). Both of these trends in land use change have direct and indirect effects on many of the environmental end points listed in Section 204 of EISA and are elaborated further in Chapter 3 below. Additional research is needed to quantify changes in the intensity of cultivation on existing agricultural land and to more effectively connect changes in land use to the environmental impacts of concern.

[49] A follow up study from Wright et al. (2017) to examine trends from 2008-2016 is also in preparation (Tyler Lark personal communication).

2.5 International Land Use Change

This section discusses land use change drivers at the global scale. First, we provide an overview of observed land use change globally, including brief discussions of trends in agricultural intensification and land use changes in regions that have been major exporters of biofuels to the United States. Second, we discuss economic modelling studies that have attempted to estimate the global land use change impacts attributable to crop-based biofuels. Overall, we find that the conclusions from the 2011 Report on international land use change still apply.

2.5.1 Observed International Land Use Change

Land use changes that occur outside of the U.S. are also drivers of the environmental impacts associated with biofuel production. Such land use changes may be directly or indirectly linked with the production of biofuel feedstocks, and there are many other direct and indirect drivers for land use change, such as urbanization, economic development, and climate (UNCCD 2017).

While U.S. biofuel production is accountable for only a fraction of global crop land area, it is instructive to review global trends in land use that coincide with the recent ramp up in biofuel production. For context, the figure below shows net land use changes from 2000 to 2007 and from 2007 to 2014, as reported by the Food and Agriculture Organization of the United Nations (FAO).[50] The FAO land use data are an annual time series that can be used to evaluate any number of time period combinations. Figure 18 shows global land use change during the seven years preceding and following the enactment of EISA 2007.

For both time periods, Figure 18 shows gains in area harvested and arable land and permanent crops, and losses in the area of forests and permanent meadows and pastures.[51] For area harvested, crops that are planted and harvested more than once on the same field during the year are counted as many times as harvested. For the category called arable land and permanent crops (hereafter "arable land"), the same field would be counted only once per year. Arable land includes cropland currently in production as well as potential cropland (similar to the total cropland category reported in the USDA Census, discussed above in Section 2.4). Area harvested and arable land overlap and are therefore not mutually exclusive or additive. From 2000 to 2014, harvested area increased by 504 million acres, while arable

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[50] FAOSTAT, available at http://www.fao.org/faostat/en/#home, accessed January 2018. The most recent year reported for land use data was 2014 at the time the data were accessed.

[51] Definitions for these land use categories are provided in the Appendix B table of Key Terms for Major Land Use Change Studies.

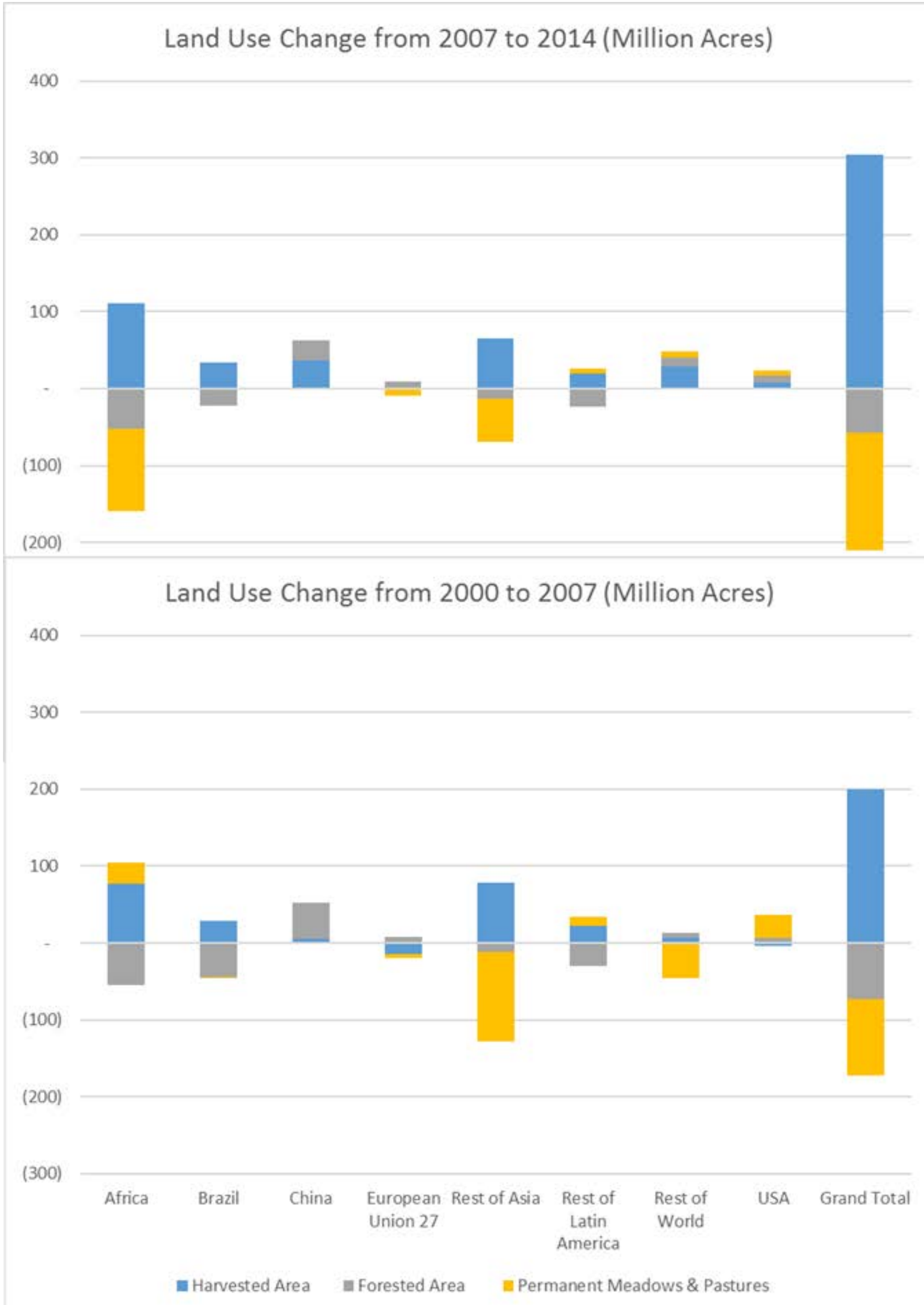

Figure 18 Global land use change by aggregate region (data from FAOSTAT[50]).

land increased by 116 million acres. The ratio of area harvested to arable land increased during this period, implying an increase in harvests per planted acre, an increase in the share of potential cropland planted and harvested, or likely some combination of both. The FAO data do not allow us to separate these effects, and understanding the extent and details of these intensification channels is an area for ongoing research.

With increased population, per capita income, and biofuel production during this time period, crop extensification would likely have been larger without concurrent intensification through increased crop yields, rates of harvesting planted areas and harvests per year (Ray et al. 2013; Langeveld et al. 2014; Babcock 2015). For example, based on data reported by FAO, from 2000 to 2014 crop production (total mass) increased by 42 percent, while harvested area increased by only 17 percent, accompanied by a 21 percent increase in yield (tons per acre). Total factor productivity (TFP) is a measure that provides a more comprehensive accounting of productivity gains than yield per acre (Fuglie et al. 2013). Data from USDA-ERS suggests that since the year 2000 TFP (growth due to getting more output from existing inputs) has been the main factor driving global agricultural output growth (see Figure 19). It is unclear whether or to what extent U.S. biofuel policies have contributed to such gains in TFP.

Although the use of agricultural land has intensified, cropland extensification and deforestation has continued. Cropland expansion that results in forest loss is a particularly acute driver of

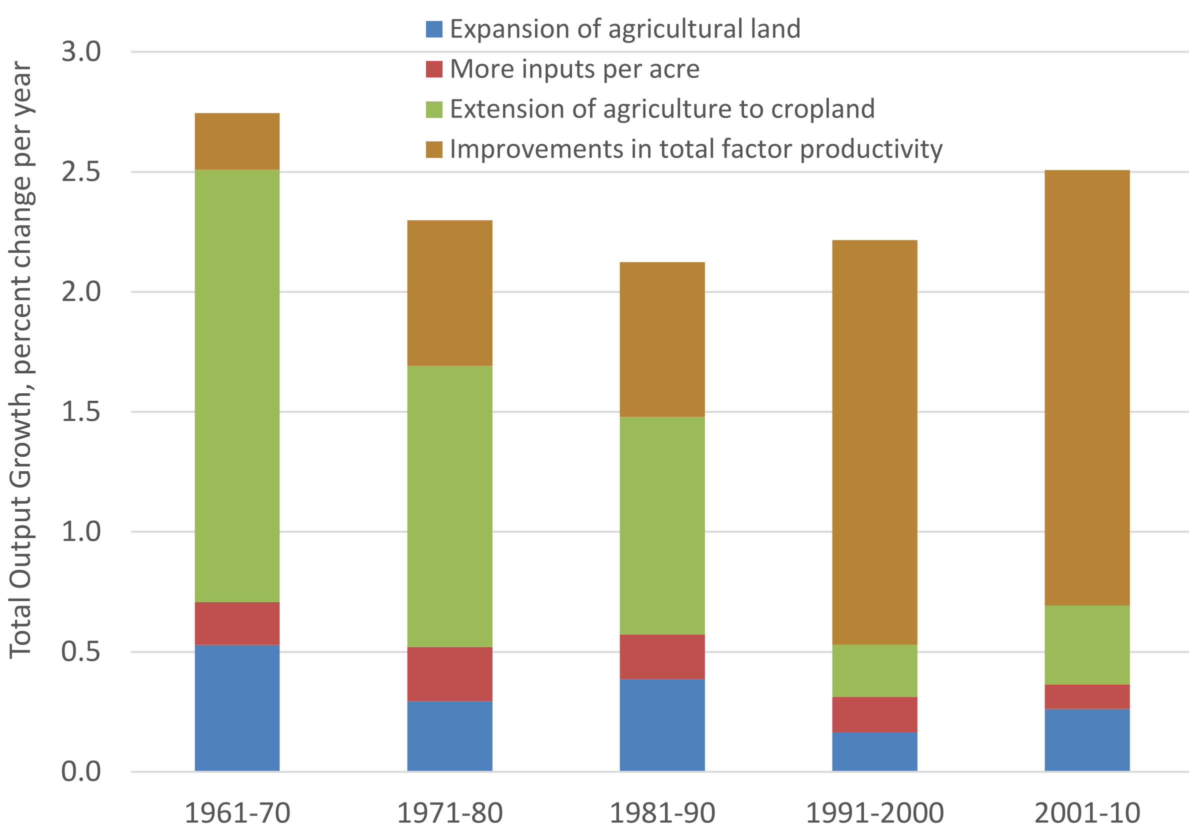

Figure 19 Global total factor productivity. From Fuglie et al. (2013), derived from FAO and other agricultural data using method described in Fuglie et al. (2012).

environmental impacts. Although forest loss is caused by many factors, it is instructive to look at recent trends. The 2015 Global Forest Resources Assessment (FAO 2016)[52] reported a total loss in forest area of 83 million acres from 2005-2015, with per year forest area losses being roughly equal between 2005-2010 and 2010-2015 [8.15 million acres/year during 2010-2015 (Keenan et al. 2015)]. Overall, the net annual rate of forest loss has slowed from 0.18 percent in the early 1990s to 0.08 percent during the period from 2010-2015.

In addition to looking at global trends, it is helpful to consider individual regions. Here we touch briefly on trends in recent land use changes in countries that are major exporters of biofuels to the U.S. In recent years, the largest sources of biofuel imports to the U.S. have been sugarcane ethanol from Brazil, soy biodiesel from Argentina and palm oil biodiesel from Indonesia (see Section 3.7). Planted and harvested area of sugarcane in Brazil increased by about 9.9 million acres between 2005 and 2010 (Adami et al. 2012; Marin et al. 2016), with an additional 3.3 million acres added between 2010 and 2014.[53] The sugarcane expansion occurred mainly through conversion of pasture land and has been linked to conversion of other natural vegetation including forests (Adami et al. 2012; Filoso et al. 2015). Soybean harvested area increased by 29.1 million acres from 2004 to 2017.[55] From 2004 to 2016, the annual rate of deforestation in the Amazon decreased from 6.9 to 2.0 million acres but has been increasing in recent years from a low of 1.1 million acres in 2012.[54] The harvested area of soybeans in Argentina increased by 1.6 million acres from 2004 to 2017,[55] largely at the expense of native grasslands in the Pampas region (Modernel et al. 2016). In Indonesia, harvested palm oil area has increased by 12.2 million acres from 2004 to 2017,[55] while forest loss has been around 1.7 million acres per year between 2010 and 2015 (Keenan et al. 2015).

Cropland expansion and natural habitat loss (including forests) have been observed internationally during the implementation of the RFS program. It is likely that increased biofuel production has contributed to these land use changes, but significant uncertainty remains about the amount and type of land use changes that can be quantitatively attributed to U.S. biofuel consumption (see Box 3 on Attribution).

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[52] FRA 2015 data was developed from responses to surveys by individual countries. The survey has a common reporting framework, agreed definitions and reporting standards.

[53] FAOSTAT. http://www.fao.org/faostat/en/#home.

[54] Brazil National Institute for Space Research (INPE). http://www.inpe.br/ingles/index.php.

[55] USDA Foreign Agricultural Service, Production, Supply, and Distribution, PSD Online. https://apps.fas.usda.gov/psdonline/app/index.html#/app/home.

2.5.2 Economic-Model Based Estimates of Biofuel-Induced Land Use Change

The 2011 Report reviewed modeled estimates of biofuel-induced land use changes and their impacts. It summarized the land use change results from USEPA 2010, which estimated land use change GHG impacts using two partial-equilibrium models (FASOM and FAPRI-CARD). EPA (2010) produced a range of results based on quantitative sensitivity analysis of the satellite data and land use change emissions factors used in the modeling framework. Quantitative sensitivity analysis was not performed for the economic parameters within the FASOM and FAPRI-CARD models, but a "high-yield" scenario was run for comparison. The 2011 Report compared EPA (2010) with other modeling projections available at the time and found "the results of modeling projected impacts are diverse and it not possible at this time to predict with any certainty what type of land use change in other countries will result from increased U.S. demand for biofuel or what its environmental consequences will be (p. 5-7)." This section reviews additional modeling studies since the 2011 Report.

Figure 20 below summarizes results from studies since the 2011 Report that estimated the land use change associated with corn ethanol. The figure and the discussion in this section focuses on corn ethanol since it is the most intensively studied biofuel in the U.S. and accounts for the largest volume of biofuel. Although GHG impacts are outside the scope of the current report, this figure presents GHG emissions per unit of corn ethanol produced as a proxy for the overall scale of land use change. This was the only single readily available common metric across the studies reviewed that summarizes the scale and nature of the land use changes projected. Displaying GHG results has the benefit of synthesizing multi-dimensional results into one comparable metric. All else equal, results with higher land use change GHG emissions are associated with greater areas of land use change and greater clearing of high-carbon stock lands such as primary forests. The figure is not meant to be comprehensive, and the results presented are limited to studies that did original modeling in a peer-reviewed publication or as part of a regulatory analyses performed for a governmental body.

The studies reviewed can be categorized by the type of model or analytical methods used. A number of studies used partial equilibrium models representing the agricultural sector (e.g., FAPRI) or the agricultural and forestry sectors (e.g., FASOM and GLOBIOM). Another group of studies used computable general equilibrium models (e.g., GTAP-BIO or MIRAGE). While CGE models have the advantage of representing the entire global economy, they often lack detail in the agricultural and forestry sectors compared to partial equilibrium models. Another group of studies developed reduced form models for the express purpose of evaluating the uncertainty flowing from certain aspects of biofuel-induced land use change modeling.

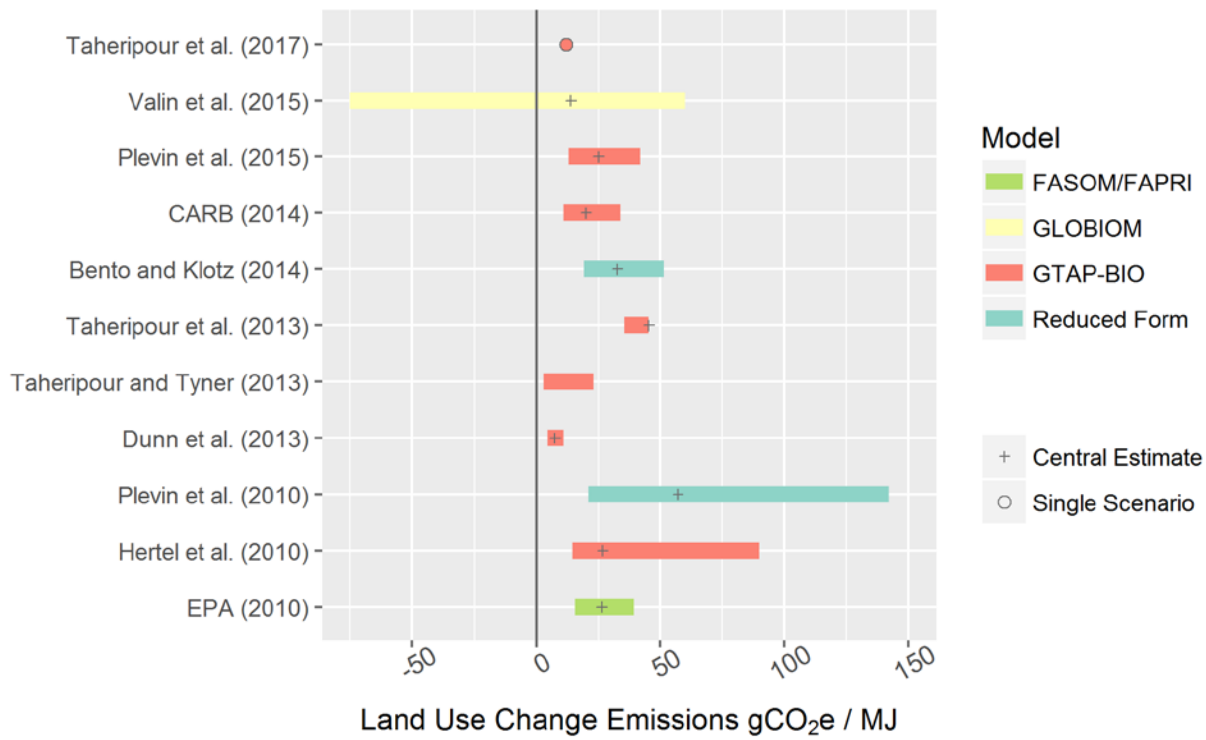

Figure 20 Summary land use change GHG emissions estimates for corn ethanol. Estimates include global land use change emissions (U.S. and international). Studies are ordered by year of publication and then alphabetical by author. Colors summarize the model or other analytical method used. The bars show the range of reported results for each study and the "+" sign denotes the study's central estimate based on our interpretation. One study, Taheripour et al. (2017), did not report a range. These studies evaluated different scenarios and amortized emissions over different time periods (20 or 30 years); no attempt was made to adjust or harmonize the reported results, other than reporting common units.

While some authors have concluded that international corn ethanol land use change estimates are trending downward over time (Souza et al. 2015), it is important to note that many of the recent studies with lower estimates are based on simulations with the GTAP-BIO model. Every model has important caveats and limitations that need to be considered before drawing conclusions from their results. For example, as a static model GTAP-BIO does not capture dynamic reference conditions such as changes in demography, technology, and climate, or whether biofuel-induced land use changes occur in regions where agricultural land is expanding or decreasing over time (Kløverpris et al. 2013). Furthermore, although the GTAP land cover database includes unmanaged areas (Baldos 2017), the GTAP-BIO model does not allow approximately four billion hectares of unmanaged grasslands or inaccessible forest to be brought into production (Gibbs 2010), which may significantly limit the estimates of land use changes associated with biofuel scenarios.

It is also important to note that many of the recent studies with lower estimates do not include quantitative sensitivity analysis. Many of the key parameters in land use change models remain highly uncertain [e.g., Zilberman et al. (2013) and Tokgoz et al. (2014)], which reduces the weight that can be given to any individual model run. Studies that have included robust sensitivity analysis have reported wide ranges of results stemming from parametric uncertainty alone [e.g., Plevin et al. (2015) and Valin et al. (2015)]. Other sources of uncertainty that are difficult to quantify flow from variability in model structures, scenario design, and other methodological choices (Plevin et al. 2017b).

Another set of studies (not shown in the figure above because they did not include original model estimates) reviewed biofuel modeling and proposed alternative methodologies for consideration. For example, Kim et al. (2012) suggested that some of the land use change that modelers have attributed to biofuels should instead be apportioned to consumers' dietary preferences. On the other hand, Searchinger et al. (2015) argued that current models project benefits for biofuels only because they assume global food consumption will be reduced as biofuel production increases. These studies highlight just a couple of the methodological questions surrounding biofuel modeling that have not been fully resolved.

The discussion in this section has focused on corn ethanol, but many of the same general observations apply to soybean oil biodiesel, sugarcane ethanol, and other biofuels derived from planted crops or trees. It is worth noting, however, that, indirect land use change "factors for biodiesel crops are considerably higher and subjected to higher uncertainty levels than ethanol crops" (Souza et al. 2015). Modeling results for dedicated energy crops tend to be lower than comparable studies for food-based crops [e.g., EPA (2010), Valin et al. (2015), and Dunn et al. (2013)], but since energy crop production has been limited in scale these studies must rely on assumptions about how the industry will develop.

As predicted by NAS (2011), "scientists will undoubtedly continue to refine their models to improve estimates of GHG emissions as a result of land use changes. However, uncertainty of GHG emissions from land use and land cover changes can be expected to remain large because actual land changes and their relation to increasing biofuels production in the United States will only be observed as markets adjust to increased biofuel production. Even with long-term empirical data on land use and landcover changes, measurement of associated GHG emissions, and data on agricultural markets, estimating the global GHG benefits or emissions from U.S. biofuel production will require a comparison to reference scenario, which inevitably is a simulation of what would have happened absent biofuels" (p. 192). Since a reference scenario cannot be measured, indirect land use change impacts are by definition uncertain. A recent study conducted by Woltjer et al. (2017)Woltjer et al. (2017) that reviewed biofuel land use change modeling since 2012 for the European Commission concluded that, "progress in the calculation of [indirect land use change] effects from biofuel production, and reduction of uncertainties, has been limited" (p. 95). Significant room for improvement remains in basic areas such as model comparison, data standardization, and empirical support for economic parameters (Souza et al. 2015). Many authors have also highlighted the inherent uncertainty associated with biofuel-induced land use changes and proposed various ways to address and factor such uncertainties into decision making (Kocoloski et al. 2013; DeCicco et al. 2016; Plevin et al. 2017a). New research since the 2011 Report has improved our understanding of biofuel-induced land use change modeling, but the overall conclusions we can draw from this body of modeling have not changed.

2.5.3 Conclusions

Conclusions for observed international land use change:

- Global cropland area has expanded since the year 2000, coinciding with the increase in U.S. biofuel production. During this period, the ratio of area harvested to arable land increased and crop yields increased significantly, due in large part to gains in total factor productivity.
- Agricultural extensification and deforestation have been documented in countries that are major exporters of biofuels to the U.S., including Brazil, Argentina, and Indonesia.
- Cropland expansion and natural habitat loss (including forests) have been observed internationally during the implementation of the RFS program. It is likely that increased biofuel production has contributed to these land use changes, but significant uncertainty remains about the amount and type of land use changes that can be quantitatively attributed to U.S. biofuel consumption (see Box 3 on Attribution).

Conclusions for economic-model based estimates of biofuel induced land use change:

- Researchers have continued to update and refine economic models to estimate biofuel-induced land use changes.
- Due to inherent challenges, uncertainties are large and progress in reducing the sources of uncertainty has been limited.

Box 3. Attribution of the Environmental Effects of Biofuels

Most environmental effects of biofuel production are associated with the feedstock production stage (EPA 2011; Hellwinckel et al. 2016). At the feedstock production stage, land use change has been identified as one of the primary drivers affecting environmental impacts. Farmers' decisions regarding land use and management are influenced in part by market prices (e.g., future price of corn), which are in turn affected by myriad antecedent factors, such as weather and policies (Roberts et al. 2013; Hellwinckel et al. (2016); Carter et al. 2017). The dominant biofuel feedstocks in the U.S. currently are corn and soybeans (see Section 2.2); thus, the environmental effects of biofuels at this time are due to some portion of the land use and management of growing corn and soybeans. However, these feedstocks are also produced for other purposes, such as animal feed, many food and industrial products, and export. Therefore, only a percentage of the environmental consequences of growing corn and soybeans can be attributed to biofuel feedstock production. The question is what percentage of the environmental effects of producing corn and soybeans are attributable to corn-grain ethanol and soy biodiesel, respectively? And, from this follows – what percentage of these environmental effects are attributable to the Renewable Fuel Standard Program specifically?

Understanding the type and location of land use attributable to biofuel feedstock production is a first step towards attribution of environmental effects. Changes in crop types or domestic land use, such as conversion of land to agriculture, can be caused by variety of factors, and allocating proportional causation to these factors, including biofuels, can be difficult (Efroymson et al. 2016). One simple method is to apply the percentage of corn grain and soybeans used for biofuels directly to land use, but there are limitations with this approach. Currently, approximately 40% of corn grain and 12% of soybeans produced nationally go to biofuels (see Section 2.2). But it would be inaccurate to assume that changes in these percentages are equivalent to changes in land use due to biofuel production. Improved production efficiency can result in more volume produced on the same land area. A constant level of biofuel feedstock production and associated land use combined with lower total production and land use would result in a higher percentage of production attributed to biofuels but no actual land use change.

A co-product of corn ethanol production is distillers dried grains with solubles, which can displace corn grown for animal feed and reduce the percentage of land (and environmental effects) attributable to corn grown for ethanol. A recent study examined this displacement and found that accounting for distillers grains reduced corn acreage attributable to ethanol from 40% to 25% nationally in 2011 (Mumm et al. 2014). Additionally, only soybean oil (which is approximately 20% of the soybean by weight; Section 2.2) is used for biodiesel production, so this means approximately 2.5% of the soybean harvest by mass is attributable to biodiesel. If feedstock production for biofuels were evenly distributed across the country, then 25% and 2.5% of corn and soybean acreage, respectively, are reasonable first estimates attributable to biofuels.

(Continued)

Box 3 (Continued)

However, biofuel-induced land use changes may not be evenly distributed, with feedstock production potentially concentrated in certain areas. Land use changes in areas around biorefineries or those with new plantings since 2007 may indicate the effects of biofuel production. This can be accounted for using another approach for assessing attribution: statistical correlative analysis. Motamed et al. (2016), for instance, estimated that for every 1% increase in an area's ethanol refining capacity, its corn acreage and total agricultural acreage increased by 1.5% and 1.7%, respectively. This finding suggests that corn ethanol production has been responsible for increasing corn production and land conversion around biorefineries.

Agro-economic models (e.g., FASOM, POLYSYS, and REAP) have also been employed, comparing land use and crop production with and without a given amount of biofuel production [e.g., Malcolm et al. (2009)]. As an example, Malcolm et al. (2009) estimated that biofuel volume targets would lead to an increase of approximately five million acres of cropland by 2015, with most due to corn ethanol. There are tradeoffs between direct measurements versus economic modeling approaches. It can difficult to assign cause using direct observation, while modeling studies may be overly simplistic, failing to account for key drivers or complex interactions, and it is difficult to validate model projections with historical data.

Besides land use, the environmental effects on air, soil, water quality, and other end-points depend not just on the crop and preceding land cover, but how the land is managed (e.g., no-till management, pesticide usage, riparian buffer strips, etc.). For example, if additional biofuel-induced corn is grown on marginal lands at higher rates than average, it could require more fertilizer and lead to higher nutrient runoff or leaching to waterways. Conversely, if additional biofuel-induced corn is grown on prime agricultural land with riparian buffers, less fertilizer and lower nutrient and sediment runoff could result. Empirical or dynamic ecosystem models (such as SPARROW, EPIC or SWAT) can help simulate these effects once land use is assigned. These estimates can then be used in life cycle analyses to determine environmental effects across all stages of production and use. Such combined analyses remain an area of emerging research.

Currently, we can state that biofuels are responsible for a percentage of domestic land used for—and the environmental effects from—corn and soybean production, including newly converted land. However, using peer reviewed information that forms the backbone of this Report, we cannot quantify these percentages with confidence at this time based solely on that information without new analyses. Since 2011, a clearer picture of U.S. land use change has emerged (see Section 2.4) potentially allowing a quantitative attribution to biofuels and estimation of environmental effects in the future. Moreover, the general relationship between the two current U.S. biofuel feedstocks (corn and soybeans) and environmental impacts is well known from decades of agricultural research. Each individual section in the Impacts Chapter below discusses what is currently known about the effects of corn and soybean production in general, and biofuel production specifically, on their respective environmental end-points.

3 Environmental and Resource Conservation Impacts

3.1 Air Quality

3.1.1 2011 Report Conclusions

According to the 2011 Report, the most negative air quality impacts from biofuel production were associated with production facilities using coal as the main energy source.[56] The report added that air quality impacts could be mitigated through use of cleaner fuels, such as natural gas, and more efficient processes and energy-generation equipment. In addition, energy-saving technologies such as those used by combined heat and power (CHP) facilities are also an effective means to reduce air emissions associated with biofuel production. The report concluded that the impacts from transport of biofuels are not expected to be significant, although air quality will be affected to a small degree locally by emissions from biofuel transport via rail, barge, and tank truck and by evaporative, spillage, and permeation emissions from transfer and storage activities.

The 2011 Report also concluded that, for ethanol blends, end-use emission rates were expected to be higher for nitrogen oxides (NO_x) relative to gasoline. The effect of ethanol on NO_x occurs because addition of ethanol to gasoline adds oxygen to the fuel and modifies the air/fuel mixture in a way that leads to higher NO_x emissions. The 2011 Report found that end-use emissions for ethanol blends were independent of feedstock. In 2011, the National Academy of Sciences also released its report, "Renewable Fuel Standard: Potential Economic and Environmental Effects of U.S. Biofuel Policy" (NAS 2011), which reached conclusions generally consistent with EPA's 2011 Report. It concluded that air quality modeling suggests that production and use of ethanol as fuel to displace gasoline is likely to increase such air pollutants as $PM_{2.5}$, ozone, and SOx in some locations. The NAS report was a synthesis of available research by a team of experts and emphasizes the spatial component of impacts with some effects being local (air quality) and others regional or global (greenhouse gases). Discussion of air quality impacts in the NAS report focused on ethanol.

The 2011 Report concluded that, relative to petroleum-based diesel fuel, biodiesel increases NO_x emissions and decreases particulate matter (PM), hydrocarbon, and CO emissions. It also found that

[56] EPA's RFS2 regulatory impact analysis (EPA, 2010) identified significantly higher CO, NO_x, PM10, $PM_{2.5,}$ and SO_x emissions for coal plants than plants using natural gas. However, the majority of ethanol is produced by plants using natural gas.

emissions of some pollutants (PM, nitrous oxide, CO) are higher with plant-based rather than animal-based biodiesel feedstocks.

The 2011 Report also summarized results of air quality modeling done for EPA's RFS2 rule (EPA 2010; Cook et al. 2011). This modeling suggested that the increased biofuel use mandated by EISA would increase ambient $PM_{2.5}$[57] in some areas and decrease $PM_{2.5}$ in others, with small ozone increases over much of the country along with improvements in a few areas. Ozone increases occur in NO_x-limited areas of the country (VOC levels are high relative to NO_x). However, in a few VOC-limited areas, such as Southern California, NO_x increases may decrease ozone. The RFS2 modeling also found little impact on ambient concentrations of most air toxics. In reaching these conclusions, the 2010 assessment of the RFS2 rule (EPA 2010) took into consideration offsetting emissions impacts associated with reductions in fossil fuel volumes due to replacement with biofuels. Uncertainties with that analysis included limited vehicle emissions data for advanced technology vehicles, uncertainty in the assumed fuel types and blend concentrations, uncertainty in emissions from (cellulosic) ethanol production plants, uncertainty in transport/fuel storage, and uncertainties in the model itself (e.g., in the chemical mechanism). However, since that study there has been some limited additional research on emissions impacts associated with production of different feedstocks [e.g., Zhang et al. (2016)], and especially biofuel end-use emissions, as discussed below. The relatively limited research that has been published over the last six years continues to support the findings discussed above.

In the following sections, we revisit these 2011 air quality conclusions. First, we provide a brief overview of the major changes in the drivers of air quality and their impacts since 2011. We address these by life cycle stage. Second, we highlight changes in our understanding of the connections between drivers and impacts since 2011. Third, we focus on likely future changes, and, finally, we provide bulleted conclusions.

3.1.2 Drivers of Impacts to Air Quality

Air quality, as measured by the concentration of air pollutants in the ambient atmosphere, can be directly affected by increased production and use of biofuels through changes in emissions of air pollutants during: (1) feedstock production; (2) conversion of feedstocks to biofuels; (3) transport of biofuels and feedstocks; and (4) combustion of biofuels in vehicles. Air quality can also be impacted indirectly, through price-induced impacts associated with increased production and use of biofuels, such as changes in petroleum fuel consumption and changes in agricultural production and land use. Direct

[57] Particulate matter with an aerodynamic diameter of 2.5 μm or less.

impacts on emissions occur due to changes in biofuel volumes produced and consumed and changes in technologies and practices in each of the previous four processes. For example, as farmers replace older equipment with new clean equipment with modern emission controls, emissions associated with feedstock production decrease. Indirectly, petroleum production displacement from increased use of biofuels impacts emissions, as do changes in fuel properties due to the addition of biofuels to petroleum fuels. Emissions of NO_x, sulfur oxides (SO_x), CO, volatile organic compounds (VOCs), ammonia (NH_3), and PM can be impacted at each stage of biofuel production, distribution, and usage and depend on feedstock type, land use change, and land management/cultivation practices. As a result, the emission impacts of biofuel production and distribution and the offsetting impacts on petroleum fuel production and distribution are substantial and must be considered along with end-use impacts for VOC, PM, and NO_x (EPA 2010). In addition, emission and air quality impacts associated with feedstock production and conversion of feedstock to biofuels are highly localized. The magnitude, timing, and location of all these emissions changes can have complex effects on atmospheric concentrations of criteria pollutants (e.g., O_3 and $PM_{2.5}$) and air toxics, the deposition of those compounds, and subsequent impacts on human and ecosystem health. In this review, we focus primarily on changes in emissions as a surrogate for changes in air quality.

As discussed in Chapter 2, current renewable fuel volumes are much lower than the applicable volumes specified in EISA. The vast majority of renewable fuel sold is ethanol, primarily produced from corn, and biodiesel, primarily produced from soybean but also other plant- and animal-based oils. There has been very little market penetration of fuels derived from cellulosic and other advanced feedstocks. As a result, research on biofuel impacts on air quality has focused on corn ethanol and soy biodiesel more than on biofuels from other feedstocks. The next section will focus on drivers impacting air quality from ethanol production and use, while the following section will focus on biodiesel. Discussion will focus on research published since the last report; thus, the discussion will be limited to those drivers where significant new information is available. Key drivers from ethanol use include production of feedstock, production of the biofuel itself, transport of the fuel, and end use of the fuel in vehicles.

Separately, recall that the current report does not address GHG emissions or associated impacts from biofuels. EISA established mandatory life cycle GHG reduction thresholds for qualifying renewable fuels that would replace petroleum-based fuels under the program.[58] In a previous analysis,

[58] The Act exempts fuel from facilities that commenced construction prior to EISA enactment, and ethanol from facilities fired by natural gas or biomass that commenced construction prior to December 31, 2009, from the minimum 20% lifecycle greenhouse gas reduction requirement that generally applies to non-advanced renewable fuels.

EPA used state-of-the-art models, data, and other information to assess the GHG emissions from biofuels (EPA 2010). The modeling of GHG emissions conducted for the RFS2 Regulatory Impact Analysis (RIA) provided a reasonable and scientifically sound basis for making determinations of whether various biofuel production pathways meet thresholds established in EISA. As discussed in Chapter 1, this report does not evaluate emissions of carbon dioxide or other GHGs from biofuel production and use, nor does it attempt to encompass GHG impacts in its conclusions. Instead, this report provides complementary information to the GHG impacts described in the RIA (EPA 2010), which should be consulted for more information on this topic.

3.1.3 Impacts to Air Quality

3.1.3.1 Impacts from Ethanol Emissions

3.1.3.1.1 Ethanol Feedstock Production and Transport

Recent research to characterize and/or quantify air quality impacts resulting from biofuel feedstocks has several unifying characteristics. A number of publications since the 2011 Report have developed spatially and/or temporally explicit life cycle inventories (LCI) of U.S. biofuel feedstock production systems' air pollutants (Tessum et al. 2012; Heath et al. 2013; Yu et al. 2013; Zhang et al. 2016).

Zhang et al. (2016) and the U.S. Department of Energy (DOE) (2017) also conducted extensive inventory analysis at the county level for various feedstocks. They concluded that switchgrass and miscanthus generate lower emissions than corn grain on a per unit biomass basis due to greater yield. They also concluded that among various cellulosic feedstocks, emission differences associated with production are offset by differences in emissions associated with transport due to differences in transport distance.

Tessum et al. (2012) described spatially and temporally explicit LCIs of air pollutants from gasoline, ethanol derived from corn grain, and ethanol from corn stover. Their results indicated that life-cycle air emissions of ethanol were concentrated in the Midwestern "Corn Belt," and that ethanol's life cycle emissions exhibit different temporal patterns when compared to gasoline. Their study also concluded that life cycle fine PM emissions were higher for ethanol from corn grain than ethanol from corn stover. They estimated that the production and consumption of ethanol from corn stover would increase Midwestern NO_X, NH_3, and $PM_{2.5}$ emissions but decrease Midwestern SO_X emissions.

Yu et al. (2013) estimated the emissions associated with hauling switchgrass and energy sorghum feedstocks for biofuel production facilities in Tennessee. Their study generated the least-cost solutions between the feedstock supply systems and biorefineries and estimated resulting emissions from hauling feedstock using EPA's MOtor Vehicle Emission Simulator (MOVES) model.[59] Their results indicated that the degree of feedstock draw area dispersion and the topography of the draw area around a biorefinery site are critical factors pertaining to the emissions associated with hauling feedstock to a biorefinery. On a more local scale, they determined that switchgrass was more suitable than energy sorghum for biofuel production in Tennessee, primarily due to the higher cost and hauling emissions associated with sorghum.

An overarching conclusion of these publications was that ethanol from corn grain had the highest overall air pollutant emission levels and that the magnitude of change in air pollutant emissions was directly connected to the spatial and temporal characteristics of the feedstock production site. These conclusions align with air quality impacts described in the 2011 Report.

3.1.3.1.2 Ethanol Production

As of mid-2017, there are approximately 200 ethanol production facilities in the U.S.[60] Over 90% of these facilities are dry mill facilities processing corn. Facilities producing ethanol from corn and cellulosic feedstocks tend to have greater air pollutant emissions relative to petroleum refineries on a per-BTU of fuel produced basis, but emission rates vary widely among facilities (EPA 2010). Emissions for the vast majority of biofuel plants are included in EPA's National Emission Inventory (NEI). Current NEI data support the conclusion of the 2011 Report that ethanol plants relying on coal have the highest air pollutant emissions.[61] However, a 2015 study based on airborne measurements suggests that emissions of hydrocarbons may be substantially underestimated in the NEI for one of the largest coal-fired biofuel production plants in the country (de Gouw et al. 2015), which could indicate more systematic underestimation if confirmed at other facilities. However, industry characterization data indicate that the number of plants relying on coal as an energy source is relatively small (less than 10% of all ethanol production facilities, accounting for less than 15% of production) and has slowly decreased over time. The changing nature of ethanol production facilities indicates that additional research on emissions from biofuel plants and factors that impact these emissions is desirable.

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[59] MOVES and Related Models, U.S. Environmental Protection Agency, https://www.epa.gov/moves/previous-moves-versions-and-documentation.

[60] EPA: Public Data for the Renewable Fuel Standard, at https://www.epa.gov/air-emissions-inventories.

[61] EPA: Air Emissions Inventories, at https://www.epa.gov/air-emissions-inventories.

3.1.3.1.3 Ethanol Distribution and Storage

While the 2011 Report concluded that emissions from biofuel distribution and storage, including emissions from loading and unloading, are not significant, the regulatory impact analysis for the RFS2 rule indicated that EISA-mandated volumes of ethanol (36 billion gallons in 2022) could result in additional annual U.S. emissions of 7,600 tons of NO_x from combustion processes during storage and transport and 19,000 tons of VOCs, primarily from storage and transport losses for ethanol and ethanol/gasoline blends. Strogen et al. (2012) concluded that, although transport of ethanol has a small impact on the overall transportation sector, suboptimal transportation (i.e., supply chain inefficiencies) of ethanol during the 2000-2009 timeframe resulted in unnecessary emissions. Some of this suboptimal transport could be reduced by direct blending of E85 at ethanol plants, a practice that is becoming more prevalent.

3.1.3.1.4 End Use

Light-duty Vehicle Fleet Emissions Testing Since 2011. Federal Tier 2 light-duty vehicle emission standards regulating NO_x, non-methane organic gases (NMOG), CO, PM, formaldehyde, and fuel sulfur began phasing in starting in 2004. These standards were fully implemented at the time of the preparation of the 2011 Report, but at that time only limited data on air emissions were available on the biofuel-related tailpipe and evaporative emissions of Tier 2 light-duty vehicles, within the peer reviewed literature or from direct vehicle and engine testing conducted by EPA. From 2009 to 2013, EPA conducted a joint study with DOE and the Coordinating Research Council, known as the EPAct/V2/E-89 Phase 3 Study. The study assessed the effects of five gasoline properties,[62] including ethanol volume, on exhaust emissions from light-duty vehicles certified to Federal Tier 2 Standards (EPA 2013a; EPA 2013b). This study continued to find that ethanol increased NO_x emissions, even though modern technology vehicles have near instantaneous control of the air/fuel ratio, as most emissions occur in these systems during times when the vehicle catalyst is not yet warmed up or air/fuel ratio is not perfectly controlled. The relationships between fuel properties and emissions developed from the EPAct/v2/e-89 Phase 3 study have been incorporated into the EPA MOVES2014a model to develop emission inventories that account for the geographic variation of in-use gasoline properties. This updated information on the effects of ethanol and other fuel properties on Tier 2 vehicle emissions addresses the significant uncertainty in EPA’s RFS2 air quality modeling analysis about the effects of ethanol on Tier

[62] Ethanol volume, aromatic content, Reid Vapor Pressure (RVP), T50 distillation point, and T90 distillation point.

2 vehicle emissions (EPA 2010). The EPA and the states use the updated MOVES2014a model for emissions analysis that informs regulations and transportation planning.

In 2016, EPA analyzed data from four different test programs[63] to determine exhaust emissions differences between light-duty flexible fuel vehicles (FFVs) fueled with E85 relative to E10 (EPA 2016a). Only non-methane hydrocarbons (NMHC) and CH_4 emissions showed statistically significant differences between E10 and E85, with reductions in NMHC and increased CH_4 for E85 relative to E10. These results for FFVs fueled with E85 have been incorporated into the EPA MOVES2014a model.

Mid-level Ethanol Blends. The use of mid-level (20% and 30% ethanol content) ethanol blends specifically formulated to increase octane to between 96 and 101 research octane number (RON) has been investigated as a means to reduce GHG emissions by allowing powertrain design changes such as additional engine downspeeding, increased compression ratio, and/or further downsizing of boosted engines (i.e., higher boost and maximum brake mean effective pressure levels) with improvements in protection against abnormal combustion phenomena such as preignition and knocking combustion (Stein et al. 2013; Leone et al. 2014; Theiss et al. 2016). Vehicles with specific design attributes to take advantage of higher RON mid-level ethanol blends have yet to be introduced. While current FFVs are capable of operation on such high RON mid-level ethanol blends, they are currently specifically designed to allow operation on lower RON E10 fuels and do not have the design attributes necessary to take full advantage of increased RON fuels. Therefore, potential air quality improvements from broad adoption of these technologies has not been seen or studied.

3.1.3.2 Impacts from Biodiesel Emissions

3.1.3.2.1. Biodiesel Production

As of mid-2017, there are approximately 119 biodiesel production facilities in operation in the U.S.[63] Emissions are associated with extraction, flaring, boiler operation, and cooling processes. The most recent emission estimates are found in the 2014 NEI.

3.1.3.2.2. End Use

Renewable diesel and biodiesel blends up to 5% (B5) are fully fungible with petroleum diesel fuels and meet the ASTM D975 specifications for summer and winter grades of light distillate diesel

[63] (1) EPAct/v2/e-89 Phase 3 with 4 FFVs; (2) National Renewable Energy Laboratory, E40 with 9 FFVs (Yanowitz et al.); (3) Coordinating Research Council, E-80 with 7 FFVs (Haskew et al. 2011); and (4) EPA's Office of Research and Development with 2 FFVs (Hays et al. 2013; George et al. 2014).

fuel. Heavy-duty diesel engines without catalysts and certified to the 1994 – 2004 heavy-duty emissions standards were found to have slightly lower PM emissions, but slightly higher NO_x emissions with the use of B5 biodiesel blends (EPA 2010). However, engines equipped with exhaust catalysts (MY2007 and newer for PM; MY2010 and newer for NO_x) are not anticipated to experience any significant impact on criteria pollutant emissions due to use of these fuels, compared to petroleum diesel fuel.[64]

Diesel engine manufacturers normally consider compliance with in-use emissions requirements and emissions control system durability prior to approving specific biodiesel blend levels for use in engines. Use of fuels that contain higher levels of biodiesel than approved by the engine manufacturer's recommendations could adversely affect the durability of diesel exhaust catalyst systems and result in significantly higher pollution emissions. Therefore, use of the correct biodiesel blend and emissions control systems in heavy-duty diesel vehicles is critical to ensuring low emissions needed to meet and maintain local air quality goals. For example, deviation from recommended biofuel content may result in increased NO_x and secondary nitrate particulate matter emissions due to potassium or sodium ash impacts on selective catalytic reduction (SCR) systems used for diesel NO_x control (Williams et al. 2011). Such deviation may also result in higher ash accumulation within the catalyzed diesel particulate filter (CDPF). This can result in shorter CDPF ash maintenance intervals in heavy-duty applications or CDPF plugging in light-duty/light-heavy-duty applications that may not include CDPF cleaning as part of regularly scheduled maintenance (Brookshear et al. 2013).

3.1.4 Potential for Future Changes in Impacts

As of this report, only limited data are available on the impacts of biofuels on the exhaust and evaporative emissions from vehicles using advanced gasoline engine technologies (e.g., turbocharging/ downsizing, GDI, and Atkinson/Miller Cycle) to meet current and future light-duty GHG emission standards. However, major impacts of biofuels on inventories of criteria pollutant emissions from vehicles with advanced gasoline engines are not anticipated since vehicles complying with the 2017-2025 light-duty GHG standards also must comply with Tier 3 emissions standards.[65] Tier 3

[64]MOVES and Related Models, U.S. Environmental Protection Agency, https://www.epa.gov/moves/previous-moves-versions-and-documentation.

[65] In 2014, EPA finalized Tier 3 light-duty vehicle emissions and fuel standards. Implementation of Tier 3 began in 2017, with gradual phase-in of more stringent emissions standards from 2017 to 2025.[65] The new standards require light-duty vehicles to meet a lower fleet-average tailpipe emissions standard and per vehicle emissions standards, which represent reductions of 60% and 70%, respectively, from Tier 2. Exhaust and evaporative emissions under Tier 3 are projected to result in U.S. fleet-average emissions at approximately the same levels as California partial-zero-emission vehicle requirements. Tier 3 is expected to reduce emissions of NO_x, VOCs, $PM_{2.5}$ and sulfur dioxide (SO_2) emissions by 60 to

Box 4. Toxicology Research Related to Biofuels

Since 2011, EPA's Office of Research and Development (ORD) has conducted a series of studies to examine the potential for adverse biological responses associated with inhalation exposure to biofuel vapors or emissions from engines using biofuels. The effects of vapors from ethanol-gasoline mixtures (up to E85) with repeated exposures were tested in several animal models. These studies observed only mild effects in the exposed rodents at exposure concentrations estimated to be four- to six-fold greater than those experienced by the general population during fueling operations (Beasley et al. 2014; Boyes et al. 2014; Oshiro et al. 2014; Bushnell et al. 2015; Oshiro et al. 2015).

Additional research by ORD since 2011 demonstrated that biodiesel combustion emission exposure – to either 100% biodiesel or a blend in petroleum diesel – can induce biological effects (Madden 2015; Madden 2016). In order to minimize emissions variability, ORD researchers conducted multiple exposure studies using the same fuel lot across assays ranging from bacterial mutagenicity to rodent models of human sensitivity (EPA (2010); Bass et al. 2015; Farraj et al. 2015; Hazari et al. 2015). The evidence from this work suggests biodiesel emissions can have some similar effects to petroleum diesel emissions on inflammatory, vascular, mutagenic, and other responses. There are few findings to date in the available literature on whether repeat-exposure scenarios to biodiesel emissions can induce human effects or even a weaker response compared to emissions from petroleum diesel.

Additional research and analyses are needed to adequately understand the potential health effects of exposure to biofuels and emissions from vehicles using biofuels under real-world conditions, concentrations, and exposures including to susceptible human populations. It would be appropriate to study health effects in populations exposed to biodiesel and ethanol blends in "hotspots," such as fuel production sites, and those exposed to combustion products of biodiesel and ethanol blends, especially at high blend levels. Such studies could include drivers of vehicles utilizing those fuels.

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70 % and emissions of air toxics, including benzene, 1,3-butadiene, acetaldehyde, formaldehyde, acrolein, and ethanol, by approximately 10% to 30% relative to Tier 2.

standards include specific provisions for emissions compliance using E10 test fuels for non-FFVs and E85 test fuels for FFVs. Studies similar in scope to the EPAct/V2/E-89 study have yet to be conducted for light-duty Tier 3-compliant vehicles or for vehicles using advanced gasoline engine technologies to comply with current and future GHG standards. Such studies would improve understanding of emissions impacts of biofuels. For example, if advanced engine technologies change the speciation profile of VOC and PM, the same mass may have a different potential for forming ozone or secondary PM.

Impacts of changes in criteria pollutant levels due to increased biofuel use have the potential to adversely impact human health (see Box 4). Any alterations in the criteria pollutant concentrations, such as ozone and $PM_{2.5}$, that have impacts on expected health outcomes are more fully addressed in the appropriate Integrated Science Assessment for each pollutant, which summarize the substantial body of literature on the respective topics.[66]

3.1.5 Conclusions: Air Quality

- There is no new evidence that contradicts the conclusions of the 2011 Report concerning air quality. Those conclusions emphasized that life cycle emissions of NO_x, SO_x, CO, VOCs, NH_3, and particulate matter can be impacted at each stage of biofuel production, distribution, and usage. These impacts depend on feedstock type, land use change, and land management/cultivation practices and are therefore highly localized. The impacts associated with feedstock and fuel production and distribution are important to consider when evaluating the air quality impacts of biofuel production and use, along with those associated with fuel usage.
- Ethanol from corn grain has higher emissions across the life-cycle than ethanol from other feedstocks.
- Ethanol plants relying on coal have higher air pollutant emissions than plants relying on natural gas and other energy sources.
- The magnitude, timing, and location of all these emissions changes can have complex effects on the atmospheric concentrations of criteria pollutants (e.g., O_3 and $PM_{2.5}$) and air toxics, the deposition of these compounds, and subsequent impacts on human and ecosystem health.

[66] Since 2008, EPA's Integrated Science Assessments (ISAs) have formed the scientific foundation for the review of the National Ambient Air Quality Standards by providing the primary (human health-based) and secondary (welfare-based, e.g. ecology, visibility, materials) criteria assessments. See https://www.epa.gov/isa.

- Ethanol increased NO_x emissions from light-duty vehicles certified to Federal Tier 2 Standards, likely occurring during times when the vehicle catalyst is not yet warmed up or air/fuel ratio is not perfectly controlled. However, only limited data exist on the impacts of biofuels on the tailpipe and evaporative emissions of light-duty Tier 3 vehicles and light-duty vehicles using advanced gasoline engine technologies to meet GHG emissions standards.

3.1.6 Research Needs: Air Quality

- Comprehensive studies of the impacts of biofuels on the emissions from advanced light-duty vehicle technologies (Tier 3), similar in scope to studies cited in this report for light-duty Tier 2 vehicles, would improve the understanding of the potential for biofuel-specific pollutants and associated health impacts as new technologies enter the vehicle fleet. These studies should consider engine technologies being phased into use for compliance with current and future light-duty GHG standards, with a focus on vehicles compliant with the Federal Tier 3 or California LEV III criteria pollutant emissions standards currently under implementation. Such technologies would include engine downsizing with addition of turbocharging, gasoline direct injection, and non-traditional thermodynamic cycles such as Miller or Atkinson.
- Additional research and analyses are needed to adequately understand the potential health effects of exposure to biofuels and emissions from vehicles using biofuels under real-world conditions, concentrations, and exposures including to susceptible human populations. It would be appropriate to study health effects in populations exposed to biodiesel and ethanol blends in “hotspots,” such as fuel production sites, and those exposed to combustion products of biodiesel and ethanol blends, especially at high blend levels. Such studies could include drivers of vehicles utilizing those fuels.
- Updated modeling is needed to incorporate improved emissions estimates as laboratory, field, and other studies lead to a better understanding of biofuel-related emissions changes and associated changes in the magnitude and composition of pollutants on air quality, health, and attainment of ambient air quality standards.

3.2 Water Quality

Water quality is adversely affected by the production of biofuel feedstocks, primarily due to the sediment, nutrients, pesticides, and pathogens directly or indirectly released during different biofuel production phases (e.g., upstream feedstock production, biofuel production, and transportation) (EPA 2003). These releases are dependent on the biofuel feedstock source, the feedstock production site's management practices, and direct or indirect land use changes associated with feedstock production. Water quality impacts, in the context of this report, are characterized as either proximal (i.e., geographically close to the water body's emission source) or as downstream water quality impacts (with more distant emission sources). Chemical (e.g., nitrogen, phosphorus) and sediment loadings are the most significant proximal effects related to biofuel production. Hypoxia and harmful algal blooms are the most significant downstream water quality impact related to biofuels, which can be found in coastal and non-coastal waters.

3.2.1 2011 Report Conclusions

The 2011 Report concluded that water quality impacts from biofuels are primarily driven by chemical inputs at the feedstock production stage (EPA 2011). The Report concluded that effluent discharge and other already-regulated factors associated with processing biomass into biofuel would likely have a lesser impact on water quality. At the time of the Report's publication, water quality impacts from EISA were characterized as negative, particularly due to corn and soybean production intensification, which was associated with higher levels of erosion and agricultural chemical inputs (e.g., nitrogen fertilizer, pesticides). The 2011 Report linked increased fertilizer runoff to eutrophication and coastal hypoxia, but it also argued that these impacts can be mitigated through conservation practices. Mitigation efforts, particularly in the Mississippi River Basin, have included the establishment of loading reduction goals and research on improved drainage strategies and the use of created and restored wetlands and vegetated buffers. The 2011 Report also suggested that water quality benefits could be achieved through perennial grass cultivation (e.g., switchgrass, giant miscanthus) on land designated for row crops. While commercial-scale use of those feedstocks was only a potential at that time, perennial grass cultivation was argued to have lower chemical inputs and higher utilization efficiencies when compared to traditional feedstocks like corn and soybeans. Lower chemical inputs and less soil disturbance may ultimately lead to lower sediment and nutrient losses to the surrounding environment.

The 2011 Report also concluded that water quality, including acreage and function of waters, was affected by pollutants discharged from biofuel production processes. Different pollutants were attributed to different biofuel production processes, where biological oxygen demand (BOD), brine, ammonia-nitrogen, and phosphorus were characterized as primary pollutants of concern from ethanol facilities, and BOD, total suspended solids, and glycerin were primary pollutants of concern from biodiesel facility effluent. The 2011 Report noted that explicit impacts resulting from biofuel production-related pollutants were dependent on a range of factors, including the type of feedstock processed, biorefinery technology, effluent controls, water re-use/recycling practices, facility location, source water, and receiving water.

The 2011 Report also pointed to leaks and spills of biofuel from above-ground, underground, and transport tanks as potential contaminant sources to ground, surface, and drinking water. Additionally, the Report noted that leaking tanks present increased risk potential for fires and explosions. The 2011 Report suggested that water contamination via spills and leaks can be minimized by enforcing existing regulations concerning corrosion protection, leak detection, spill prevention, and overfill prevention. Additionally, the 2011 Report suggested that biofuel leaks could be prevented by using appropriate materials, material standards, and/or manufacturer recommendations.

Lastly, the 2011 Report concluded that impacts on surface waters from algal cultivation for biofuel would depend on the configuration of any eventual production at commercial scales; either releases of nutrient-rich waters and/or removal of nutrients from existing polluted waters by algae were considered feasible, which could lead to either more severe or less severe water quality impacts, respectively.

3.2.2 Drivers of Impacts to Water Quality

The drivers discussed in Chapter 2 (i.e., biofuel volumes, land use, conversion technologies, agricultural practices) are inherently connected to water quality, including to the acreage and functions of waters. Direct and indirect water quality impacts attributed to biofuel volumes are dependent on several biofuel life cycle processes, including but not necessarily limited to: upstream feedstock production, biofuel production, and transportation. Land use for biofuel feedstock production has direct water quality impacts, which can include effluents and/or discharges occurring at a feedstock production site. The application of nutrients, pesticides, and/or other chemical additives for feedstock production can also ultimately affect the water quality of a feedstock production site or the surrounding area of a feedstock production site (EPA 2003; EPA 2011).[67]

As noted in Chapter 2, since the 2011 Report, corn production has intensified on land already under cultivation, and corn, soybeans, and wheat have expanded to land that was previously uncultivated. Strong correlational evidence exists that suggests biofuel production contributes to these changes (Brown et al. 2014; Wright et al. 2017), but we cannot yet quantify how much (see Attribution

[67] U.S. Environmental Protection Agency, Water Quality Assessment and TMDL Information, https://ofmpub.epa.gov/waters10/attains_index.home.

Box 3 in Section 2). Despite varying nutrient application and runoff characteristics of these different practices, direct connections between increased feedstock production and water quality impacts are beginning to be assessed. Research to evaluate the impacts of increased biofuel production and use on water quality has largely been based on modeling rather than observed changes. Models enable evaluation of the change in water quality attributable to biofuel feedstock production, which is an exceptionally difficult problem to examine by field measurements.

3.2.3 Impacts to Water Quality

In the following sections we examine the proximal water quality impacts (those near the sources of emissions into water bodies) as well as the impacts to water quality more distant from emission sources.

3.2.3.1 Proximal effects: Pesticides, sediment, nutrient, and pathogen levels in waters

Corn ethanol and soy biodiesel are currently associated with the highest national production levels. Due to their high national output, studies since the 2011 Report have evaluated water quality impacts associated with existing, projected, and/or hypothetical national biofuel production levels for corn ethanol and soy biodiesel. Several drivers can impact water quality, including the type of feedstock, management practices at a feedstock production site, and direct or indirect land use changes associated with feedstock production. Demissie et al. (2012) simulated water quality impacts in the year 2022 for the Upper Mississippi River Basin based on projected national feedstock production characteristics, which included: increased corn production, increased wheat production, increased idle land, decreased soybean production, decreased pasture-hay land, decreased use of conventional and reduced soybean tillage, and no change in soybean no-till area. While it is not possible to comprehensively evaluate the accuracy of these assumptions based on the empirical record, short-term trends (2008-2012, see land use change discussion in Section 2.4) suggest that these assumptions are consistent with observations, although soybean production may be increasing in this area. Demissie et al. (2012) concluded that projected feedstock production has mixed effects on water quality, projecting a 12% increase in annual suspended sediment and a 45% increase in total phosphorus loadings, but a 3% decrease in total nitrogen loading.

Similarly, Wu et al. (2012a) developed future scenarios of biofuel feedstock production to assess potential water quality and quantity changes associated with an increase in biofuel production and converting land to switchgrass production. Garcia et al. (2017) simulated groundwater nitrate contamination responses associated with nitrogen (N) fertilizer application and increased corn production at a national level (with an emphasis on agricultural areas throughout the U.S.). They concluded that increased corn production between 2002 and 2022 could result in approximately a 56% to 79% increase in nitrate-N groundwater concentrations.

These studies were based on projected impacts; future work with a focus on observable and attributable water quality impacts resulting from biofuels is needed to evaluate the accuracy of those projections. One instance of such work is the U.S. Geological Survey (USGS) interactive online mapper that provides results from the largest-ever assessment of water-quality changes in the U.S. (USGS 2017). The mapper illustrates and provides data for surface water chemistry trends (i.e., nutrients, pesticides, sediment, carbon, salinity) and aquatic ecology from 1972 to 2012. An example from the mapper is shown in Figure 21, which presents total nitrogen concentration trends between 2002 and 2012.

This resource unfortunately has little data from many of the hotspots of land use change identified in Section 2.4 (e.g., South Dakota, North Dakota). However, it does show in the central agricultural areas that total nitrogen concentrations appear to be declining in Iowa and increasing in Oklahoma between 2002-2012. Total phosphorus concentrations appear to be decreasing in Iowa and increasing in Kansas, Oklahoma, and parts of western South Dakota. Future reports could use the USGS

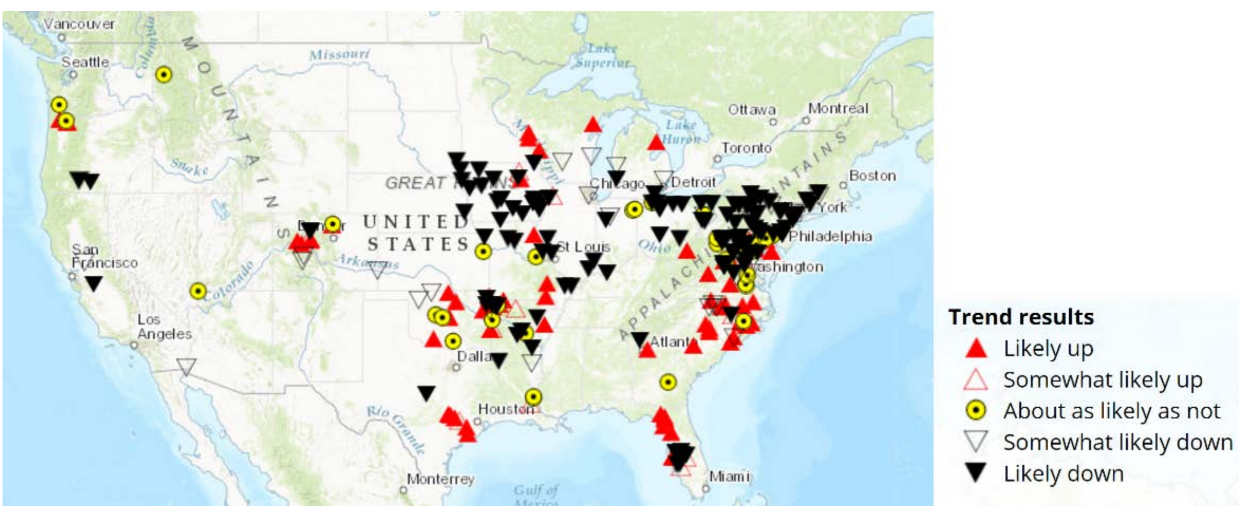

Figure 21 USGS mapper tool showing total nitrogen concentration trends between 2002 and 2012.[68]

[68] U.S. Geological Survey, Water-Quality Changes in the Nation's Streams and Rivers, https://nawqatrends.wim.usgs.gov/swtrends/.

mapper and other related tools to evaluate the water quality impacts attributable specifically to biofuel feedstock production.

Estimates of fertilizer increase from biofuel cropland expansion can be deduced from existing related studies. For example, according to the USDA Economic Research Service, the average nitrogen fertilizer input rate and the average phosphate fertilizer input rate for corn are approximately 140 pounds/acre and 60 pounds/acre, respectively.[69] Lark et al. (2015) estimated that approximately 1.28 million acres of extensification in the U.S. is due to corn. There is also an unknown amount of net conversion to corn from other crops at a national level, as well as changes in crop rotations to more continuous corn. Regional studies suggest these unknowns could be significant (Plourde et al. 2013; Ren et al. 2016). Using the national extensification estimate and nitrogen fertilizer input rates of Lark et al. (2015), these studies suggest an approximate increase of 170 million pounds of nitrogen fertilizer usage, with the potential for some of this to eventually reach waterways.

It is important to recognize that there are many factors that affect the fraction of nitrogen, or any other nutrient or chemical, applied that might reach water bodies. Higher crop yields (bushels per acre) can take up additional nutrients and conservation measures such as no-till production can reduce loss of nutrients or chemicals that run off into water bodies (Wade et al. 2015).

Since 2011, studies have quantified and confirmed the findings related to cellulosic biofuels suggested in the 2011 Report. For example, there have been several studies that have quantified nitrate-runoff reductions from croplands (VanLoocke et al. 2012; VanLoocke et al. 2017). One model found that certain scenarios of increased miscanthus production (in favor of 40% corn production devoted to ethanol) would result in a 6% reduction in dissolved inorganic nitrogen to runoff and streamflow throughout the drier portions of the Mississippi-Atchafalaya River Basin (VanLoocke et al. 2017). The collection of corn stover in places with high rates of production (e.g., Iowa) allows implementation of no-till agriculture, which is known to reduce runoff and improve water quality compared to the alternative (Dale et al. 2017). Furthermore, switchgrass uses less fertilizer than corn and thus can reduce adverse water quality effects relative to corn (Parish et al. 2012).

3.2.3.2 Downstream Effects

The 2011 Report noted that biofuel demand-related increases in corn and soybean cultivation would likely increase nutrient loadings to streams, rivers, and lakes, adding to existing high levels of

[69] U.S. Department of Agriculture, Fertilizer Use and Price: https://www.ers.usda.gov/data-products/fertilizer-use-and-price/.

impairment due to eutrophication and affecting the function of the waters (EPA 2011). Eutrophication impacts to surface waters have included harmful algal blooms (HABs), particularly in fresh waters, and hypoxia, particularly in coastal waters. Recent modeling studies have continued to conclude that row crop agriculture plays an important role in driving these downstream impacts, and they continue to suggest that biofuel feedstock production is a contributing factor. Downstream effects are also driven by weather patterns, including temperature rises, as well as the timing, amount, and form of precipitation.

Harmful Algal Blooms in Freshwater Systems. A major harmful algal bloom (HAB) observed in western Lake Erie in 2011 was attributed to unusual weather patterns coupled with long-term trends in agricultural practices that increase runoff of dissolved reactive phosphorus (DRP) (Michalak et al. 2013). A modeling study by Michalak et al. (2013) concluded that, if corn acreages continued to be at recent high levels, along with projected future increases in spring precipitation, similar events could be more likely in the future.

The main driver of HABs in western Lake Erie is phosphorus (P), particularly from the Maumee River watershed. Two recent studies indicated that biofuel production could contribute to increased P loadings to surface waters (LaBeau et al. 2014) and aquatic systems (Jarvie et al. 2015). Modeling scenarios using the Soil and Water Assessment Tool (SWAT)[70] suggest that conservation practices (e.g., filter strips, cover crops, riparian buffers) can help achieve total P targets, whereas DRP is much more responsive to reductions of P application to fields (especially inorganic P). Modeling also suggested that conversion to perennial grasses such as switchgrass and Miscanthus, even with manure application, would significantly reduce P runoff into water bodies (Muenich et al. 2016).

While P loadings determine the physical volume of a HAB, N loading appears to play a critical role in determining bloom composition. The cyanobacterium *Microcystis*, which produces the hepatotoxin microcystin, lacks the N-fixing capability of other cyanobacteria and therefore is favored by the presence of excess N. The detection of microcystin led to a temporary shutdown of the Toledo, Ohio, water supply during a Lake Erie HAB in 2014 (Levy 2017). Analyses by Taranu et al. (2017) confirm that total N concentration in lake water is a much stronger predictor than total P of the probability of detecting *Microcystis* in U.S. lakes; the percent of land cover that was agriculture within the ecoregion of a given lake was also a strong predictor (Taranu et al. 2017). Therefore, while it appears likely that demand for biofuel feedstocks increases agriculture-related nutrient loadings to surface waters, the

[70] http://swat.tamu.edu/

appearance of HABs, and in particular the prevalence of algal toxins in HAB events, will depend on a complex interplay of land use, conservation practices, and weather events.

Downstream Effects on Coastal Waters. The size of the Gulf of Mexico hypoxic zone (i.e., area with bottom dissolved oxygen < 2.0 mg/L) is a function of climate, weather, basin morphology, circulation patterns, water retention time, freshwater inflows, stratification, mixing, and nutrient loadings (Dale et al. 2010). The hypoxic zone size is also a function of loading of nitrate-plus-nitrite from the Mississippi and Atchafalaya River system during May, as well as the periodic action of tropical storms to re-aerate the bottom layer (Turner et al. 2016). However, the nature of this relationship is changing – a given nitrate/nitrite load is causing a larger hypoxic zone in recent years than in earlier years (Figure 22).

Assumptions about future nitrogen loadings from agricultural areas, and the influence of biofuel feedstock cultivation on those loadings, are critical to the estimation of future impacts. Future scenarios of increased biofuel production for Europe, simulated through the year 2050 using the Global Nutrient Export from WaterSheds (Global NEWS) model (van Wijnen et al. 2015), suggested that riverine loadings of N and P would increase as a consequence, resulting in increased risks of HABs and hypoxia in vulnerable coastal areas. This modeling exercise assumed constant nutrient use efficiencies by crops.

By contrast, in modeling scenarios of future agriculture response to biofuel demand in the Mississippi River Basin (MRB) using a similar NEWS-derived model, McCrackin et al. (2017) assumed a 24% improvement in nutrient recovery efficiency over the period 2002-2022 and further assumed that fertilizer application was matched to crop requirements. In spite of projected 28% increase in corn plantings over the period, these researchers estimated that dissolved inorganic nitrogen export from the MRB would decrease by 8%. It should be noted that the assumptions' values used by McCrackin et al. (2017) may differ from observed values moving forward.

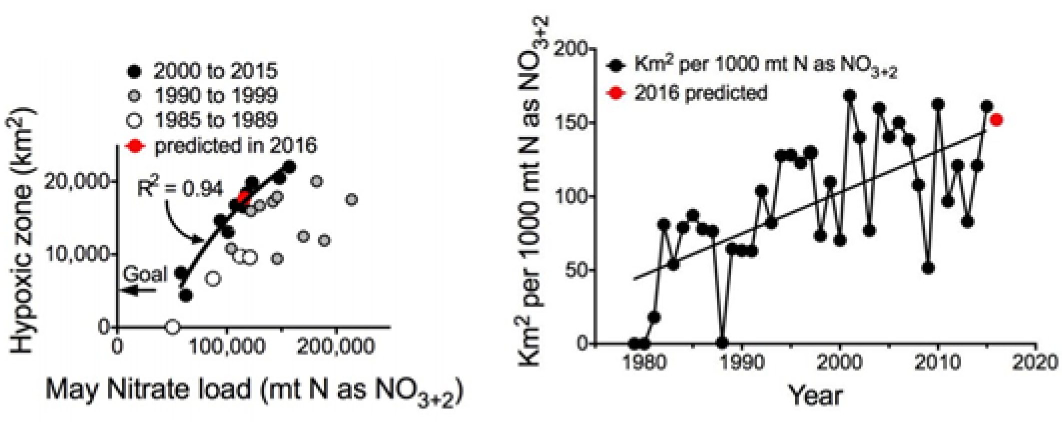

Figure 22 Changes in the measured size of the GoM hypoxic zone as related to the amount of nitrate-nitrate loading. (Turner and Rabalais 2016).

3.2.4 Potential for Future Changes in Water Quality Impacts

Recent research has shown that changing precipitation patterns influence water quality. Loecke et al. (2017) statistically connected drought-to-flood transitions (termed "weather whiplash") to increases in riverine nitrogen loads and concentrations, and pointed out that these "whiplash" events are projected to increase in the future. Given that recent studies have connected cellulosic biofuel feedstock production to relatively lower nitrogen loadings in surface waters, there is potential to decrease the water quality impact of "weather whiplash" events under specific biofuel feedstock production scenarios.

In addition, cellulosic-based biofuel production could increase in the future, which may impact water quality. Corn stover is already being used at the POET biorefinery in Iowa, and studies have shown that as a perennial, native plant, switchgrass offers several advantageous qualities, including: drought and flood tolerance; high yield capacity with little to no fertilizer application; the ability to stabilize soils and sequester carbon with long root systems; and the potential to improve water quality (McLaughlin et al. 1998; Tolbert et al. 2002; Dale et al. 2014).

3.2.5 Conclusions: Water Quality

- The 2011 Report found that corn production intensification was associated with higher levels of erosion, chemical loadings to surface waters, and eutrophication.
- Modeling studies since the 2011 Report suggest that demand for biofuel feedstocks, particularly corn grain, may contribute to harmful algal blooms, as recently observed in western Lake Erie, and to hypoxia, as observed in the northern Gulf of Mexico.
- Empirical studies documenting cropland extensification and crop switching to more corn suggest water quality impacts, but the magnitude of these changes is variable across the landscape and so may be detectable only in some regions.
- Implementation of conservation practices has been observed to result in a decrease of nitrogen, phosphorus, and soil erosion.
- Changes to future nitrogen and phosphorus loadings will depend on feedstock mix and crop management practices. Decreases in nitrogen and phosphorus loadings are possible should perennial feedstocks become dominant.
- Specific biofuel production scenarios expected to improve water quality may help decrease the water quality impact of predicted future extreme weather events.

3.2.6 Research Needs: Water Quality

- Studies are needed of water quality impacts associated with leaks and/or spills from biofuel production facilities and storage tanks. Such work would address the effectiveness of existing leak detection and cleanup approaches to address releases to the environment and resulting contamination plumes.

3.2.7 Opportunities for Future Environmental Improvements

- A decrease of N and/or P loadings is possible should perennial feedstocks become dominant.

3.3 Water Quantity

3.3.1 2011 Report Conclusions

The production of biomass feedstocks and the conversion of those feedstocks to biofuel requires water resources. The 2011 Report generally interpreted water availability in EISA Section 204 as referring to water quantity. The report concluded that water use for feedstock production would "likely not change appreciably if production takes place, as the majority does now, in regions where irrigation is not needed" (EPA 2011). However, the 2011 Report also noted that water use for feedstock production could increase under certain conditions. Some of those conditions included expansion of feedstock production into regions where irrigation is required and cultivation of row crops instead of perennial grasses with lower irrigation requirements. The 2011 Report also suggested that the water use for irrigation of feedstocks greatly exceeds the water required for conversion of feedstocks to biofuels. Water use for biofuel conversion could have localized impacts, depending on facility size and water reuse, whereas feedstock production covers a larger regional area. Finally, the 2011 Report highlighted the difficultly in generalizing the impacts of water use on water availability, suggesting that "impacts are most likely to be adverse in already stressed aquifers or surface watersheds." (EPA 2011).

Since the 2011 Report, several studies have advanced our understanding of the water footprint of biofuels [see Wu et al. (2014) as a review]. We will first discuss research that has taken a life cycle assessment (LCA) perspective, starting from feedstock production through conversion to end-use.[71] These studies have also examined water use looking at different aspects of water use, where withdrawals

[71] We note that life cycle assessment (LCA) in this section focuses only on *water use* for biofuel production supply chains. The system boundaries may differ from other LCA studies for other environmental impacts. For example, studies of the life cycle water use for ethanol production may not fully account for co-products such as distillers' grains for livestock operations.

represent the total water removed and consumptive water use the part of water withdrawn that is evaporated, transpired, incorporated into products or crops, and not returned to the same watershed.[72] A number of studies also further differentiate the consumptive water use between blue water use (irrigation water sourced from surface and groundwater and consumed through evapotranspiration [ET]) and green water use (water from precipitation and soil moisture consumed through ET) in the feedstock growing stage.[73] We will then describe how research has also moved toward more refined spatial analysis of watersheds when accounting for feedstock-production water use, recognizing the differences across regions within the U.S. as well as the influence of agricultural management practices. Finally, we will look at potential future water use impacts related to cellulosic feedstock production and provide bullets for conclusions and areas for future research.

3.3.2 Drivers of Impacts to Water Quantity

As noted above, the primary driver of impacts to water quantity is the water used for irrigation of biofuel feedstocks. To the extent that feedstock production expands into regions where irrigation is required, the demand for water will increase, whether the expansion is a direct consequence of production specifically for biofuel feedstocks or an indirect result of increased production for all uses. The question of attribution to biofuel feedstock production was addressed in more detail in the land use section in Chapter 2. Water demand for biofuel conversion processes can also drive impacts to water quantity. Although water quantity impacts may be much smaller at a national scale than those related to feedstock production, they may be locally consequential in areas that are already experiencing stress on water availability.

3.3.2.1 Feedstock Production

Several highly cited and visible articles compared the life cycle water use of biofuels relative to petroleum-based fuels on the basis of “gallons of water per mile” or “gallons of water per gallon of fuel.” These early studies characterized this issue as biofuel’s water intensity (King et al. 2008), embodied water (Chiu et al. 2009), and water footprint (Dominguez-Faus et al. 2009; Scown et al. 2011).[74] Scown

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[72] These definitions are consistent with the (U.S. Geological Survey’s (USGS) compilation of data on the nation’s water use (see www.usgs.gov/watuse).

[73] There is also a grey water category that accounts for the “virtual quantity of water required to assimilate the pollutant load from the permissible standards down to the natural background concentration” (Chiu et al. 2012). However, for this report, water quality issues are addressed separately in Section 3.2.

[74] Dominguez-Faus et al. (2009) also characterized the water demands of transportation biofuels as a “drink or drive” issue, i.e., the water is available for either drinking or for fuel production.

et al. (2011) compared different transportation energy sources and found ethanol from corn-based feedstocks to be one of the most significant uses of freshwater. Calculating the gallons of water consumed per mile of travel, they found the full life cycle water footprint of corn grain and stover to ethanol (using average irrigation rates) would require almost seven times as much surface water consumption as any other transportation power source and an order of magnitude more groundwater consumption when compared to other transportation energy sources.

Researchers have continued to refine the LCA-based water footprint of biofuels with a focus on feedstock production for both current biofuels crops and future feedstocks. Because more than 90% of corn is located in rain-fed areas where corn production is non-irrigated, Wu et al. (2014) suggested that, at the highly aggregated level, the "national water footprint of corn is consistently low to modest." However, water quantity demands depend on the crops grown, where they are grown, and how they are grown. In terms of differences among feedstocks, Dominguez-Faus et al. (2009) calculated the irrigation water required for corn-based ethanol at an average of approximately 600 liters of water per liter of ethanol-equivalent (liter/liter) and soybean biodiesel at 1300 liter/liter. Sorghum, used in some primarily corn-based ethanol facilities, was estimated to have irrigation water requirements of roughly 1500 liter/liter [see supporting information in Dominguez-Faus et al. (2009)].

Where and how crops are grown also matter because irrigation rates for the same crops can vary enormously, from no irrigation in rain-fed acres in the Midwest to high irrigation rates in more arid regions in the West. Dominguez-Faus et al. (2013) calculated a range of irrigation water use for corn ethanol between 350 and 1400 gal/gal. They estimated that if 20% of corn production was used to produce 12 billion gallons per year of ethanol in 2011 (irrigated at a weighted average of 800 gal/gal), that would amount to 1.8 trillion gallons (7 trillion liters) of irrigation water withdrawals per year. While not an insignificant amount, it represents only 4.4% of all irrigation withdrawals (Dominguez-Faus et al. 2013). Other researchers have similarly focused on the wide range of water intensity estimates between rain-fed and irrigated acreage and among a variety of crops (see Figure 23). Gerbens-Leenes et al. (2012)

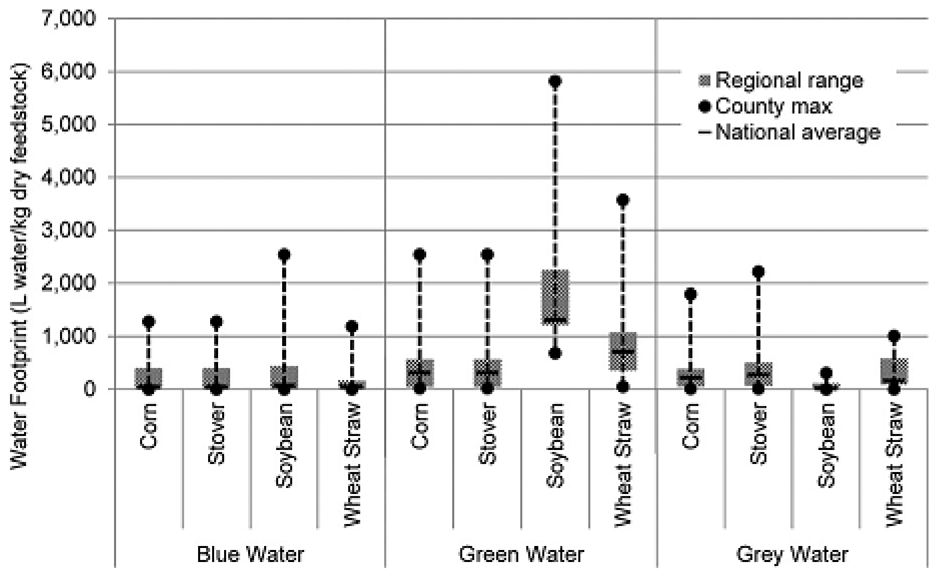

Figure 23 An estimate of the blue, green and grey water footprint associated with corn grain, stover, wheat straw and soybean during the crop growing phase. The national production-weighted average is represented by the horizontal bar, while the regional ranges (USDA regions including the Corn Belt, Southern Plains, etc.) are represented by the shaded bars. County-level variation in feedstock water footprints, shown in dashed lines, are driven by differences in irrigation and evapotranspiration (ET). [Source: Chiu and Wu (2012)].

estimated Nebraska's blue water (irrigation) footprint at three times higher than the U.S. weighted average blue water footprint. Many other corn producing states have minimal irrigation demands relative to Nebraska. Yet, it should be noted that after Iowa, Nebraska is the second largest producer of corn-based ethanol in the U.S., with 25 active ethanol facilities, many concentrated in southern Nebraska (EIA 2017). Moreover, higher irrigation demands may coincide with areas of already-stressed surface and groundwater resources, such as the Ogallala Aquifer. A report by the National Academy of Sciences (NAS 2011) highlighted the groundwater drawdown in the Ogallala Aquifer, noting that Nebraska is "among the states with the largest water withdrawals for irrigation, and its usage has continued to increase in recent years, largely driven by the need to irrigate corn for ethanol." This suggests that the majority of groundwater consumption would come from areas like Nebraska that are already impacted by over-pumping due to their high blue water footprint for corn production (Gerbens-Leenes et al. 2012).

3.3.2.2 Biofuel Conversion

Studies of water use for biofuels conversion facilities have generally quantified water consumption as gallons of water per gallon of biofuel produced, with most of the focus on ethanol, especially dry mill facilities. Process level engineering studies and surveys of biofuel facilities (Mueller 2010a) have shown declines in water requirements from 5.8 gallons of water per gallon of ethanol (gal/gal) in 1998 to 2.7 gal/gal in 2012 (Wu et al. 2012b). Anecdotal evidence[75] also points to decreases in the water intensity of ethanol facilities through more efficient water use, water recovery, and use of treated wastewater for processes such as fermentation or possibly cooling towers. There are no recently published surveys of water consumption representing all current ethanol facilities, and there are no comprehensive data on the type of water sources utilized (e.g., groundwater, surface freshwater, public supply, etc.).

3.3.3 Changes in Relationships between Drivers and Impacts

Because of the need to better understand the variation in feedstock energy use and the actual impacts on local water resources, since the 2011 Report, researchers have moved toward watershed hydrological modeling (see Table 5). A number of studies have projected future water use for scenarios of higher cellulosic feedstock production, discussed below in Section 3.3.5.

Because the majority of the growth in biofuels production has come from corn grain-based biofuels, the water consumption impacts *to date* would have come from additional water use for corn and soybean acreage. To our knowledge, there have been no studies of the changes in irrigated acres, rates of

Table 5. Methods and metrics used to assess the water quantity impacts of biofuels.

Method / metric	Definition	Example studies
LCA / Water footprint (e.g., gallons of water consumed per gallon of biofuel)	Volume of water used in a biofuel production pathway. It can include blue (irrigation) and green (rainfall) water. Calculating gallons of water per gallon of biofuel requires data or assumptions regarding feedstock water consumption, feedstock yields, and biofuel conversion rates. This water footprint can also be compared to other fuel/energy pathways.	Chiu et al. (2012); Scown et al. (2012)
LCA / Water stress index (e.g., index from 0.01 to 1.00)	The share of water consumed that is considered to be no longer available for downstream users. This is used in LCA to show the impact of water consumption on water resources, usually at a more local level, focusing on areas such as drought-prone regions.	Pfister et al. (2014)
Watershed modeling / Streamflow (total flow as $m^3 s^{-1}$ or % change)	The rate of water flow measured or modeled at a watershed outlet. This can be reported as predicted stream flow under biofuel production scenarios or as percent changes in stream flow.	Cibin et al. (2016); Housh et al. (2015)

[75] http://www.ethanolproducer.com/articles/8860/dropping-water-use

irrigation, or changes in surface and groundwater supplies associated specifically with the increased production of corn grain-based ethanol or soybean-based biodiesel. The land use section (Chapter 2) highlights analyses in Lark et al. (2015) and Wright et al. (2017) that show changes in land use, including cropland expansion in the western Dakotas and Kansas, which are areas unlikely to have sufficient precipitation for corn growth. However, there are no similar analyses that explicitly attribute recent changes in management practices, such as irrigation, to increased biofuels production and feedstock demand.

USDA Farm and Ranch Irrigation Surveys provide a general indication of the changes in water demands. From 2007 to 2012, there was a decrease in total irrigated acres of nearly 0.8 million acres in the U.S. The USDA notes that "most of the area decline occurred in the Western U.S. where drought conditions contributed to water-supply scarcity across the region" and that irrigation area is not static, but dynamic, across the U.S.[76] Over the same time period, irrigated acres of corn for grain and seed increased from 12.0 million acres to 13.3 million acres harvested, along with a higher irrigation rate of 1.1 acre-feet applied in 2012 compared to 1.0 acre-feet applied in 2007 (USDA 2013). Irrigated corn grain/seed acres are heavily concentrated in Nebraska (5.4 million acres) followed by Kansas (1.5 million acres) (see Figure 24), up by 6% and 10% respectively from 2007 levels.

Changes in irrigation practices are dependent on a number of economic and agronomic factors that affect how land is managed, making it difficult to attribute expanded irrigation to biofuels production and use without more detailed analysis. That said, studies of land use change rates have noted that "along the Ogallala Aquifer, elevated rates of land use change to corn production in Western Kansas, Oklahoma and Texas coincided with areas experiencing groundwater depletion rates ranging from 5-20% per decade" (Wright et al. 2017) (see Figure 25). Because of the potential impact on surface and groundwater resources, further studies of both land use change and land management practices should examine the linkages between increased biofuel feedstock production and changes in irrigation demands. Moreover, this work should have a particular focus on water stressed areas such as the Ogallala Aquifer.

[76] https://www.ers.usda.gov/topics/farm-practices-management/irrigation-water-use/background.aspx

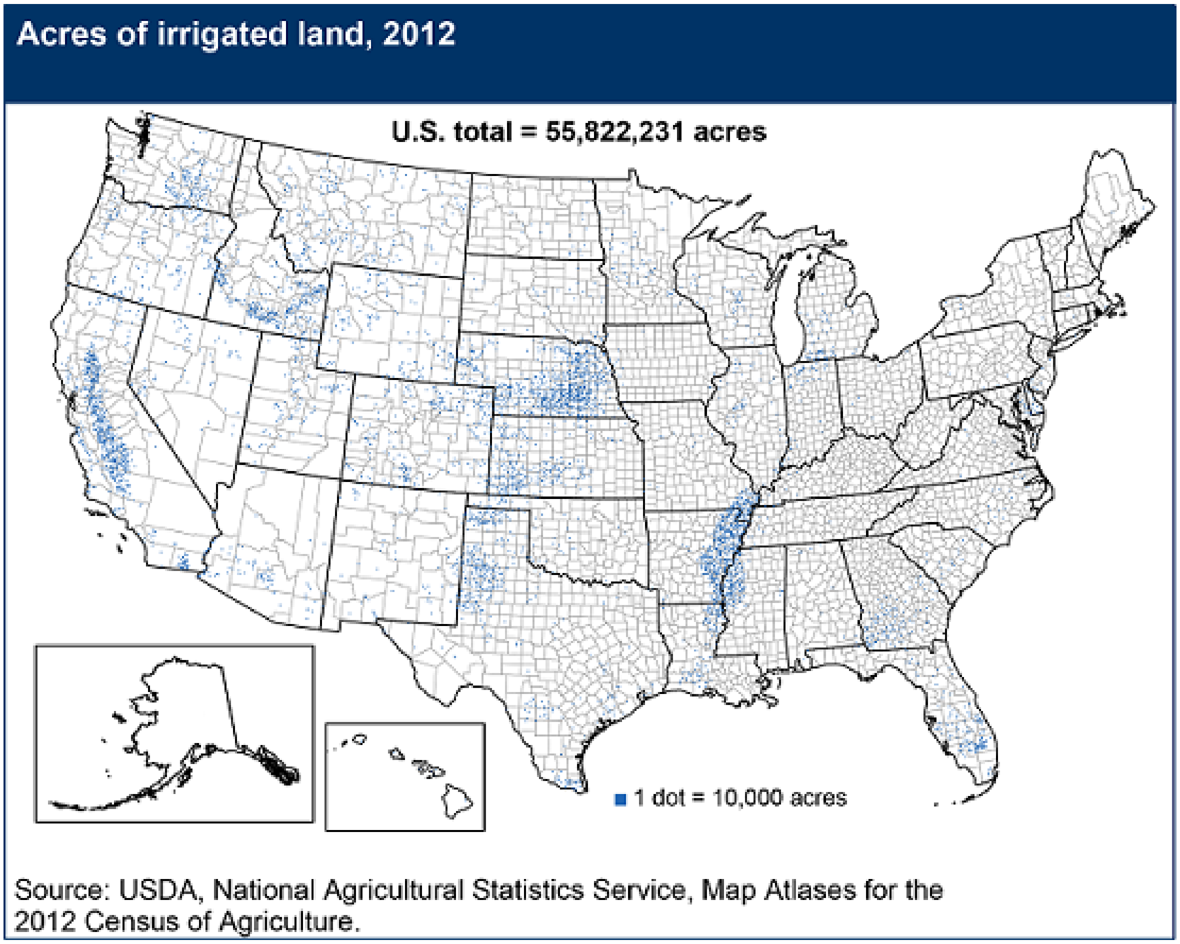

Figure 24 Acres of irrigated land in 2012, based on the USDA Farm and Ranch Irrigation Survey. Source: https://www.ers.usda.gov/topics/farm-practices-management/irrigation-water-use/background.aspx

There also have been advancements in understanding the drivers of water use for cellulosic biofuel feedstocks. For cellulosic feedstocks, given the small amounts of crops such as switchgrass or miscanthus actually in production, any assessed water use impacts will be based more on modeling studies or research and experimental scale production, rather than on widespread commercial production levels. We also caution that results from these studies depend on aggregate feedstock scenarios and simulations, compared to what can be observed empirically for corn-based ethanol production and changes in water demand and stress. Impacts will depend on which cellulosic feedstocks are grown, where and how they are grown, which best management practices are followed, technological change in irrigation practices, and potential changes in rainfall and air temperatures due to a changing climate (Le et al. 2011; Dominguez-Faus et al. 2013; Ha et al. 2017).

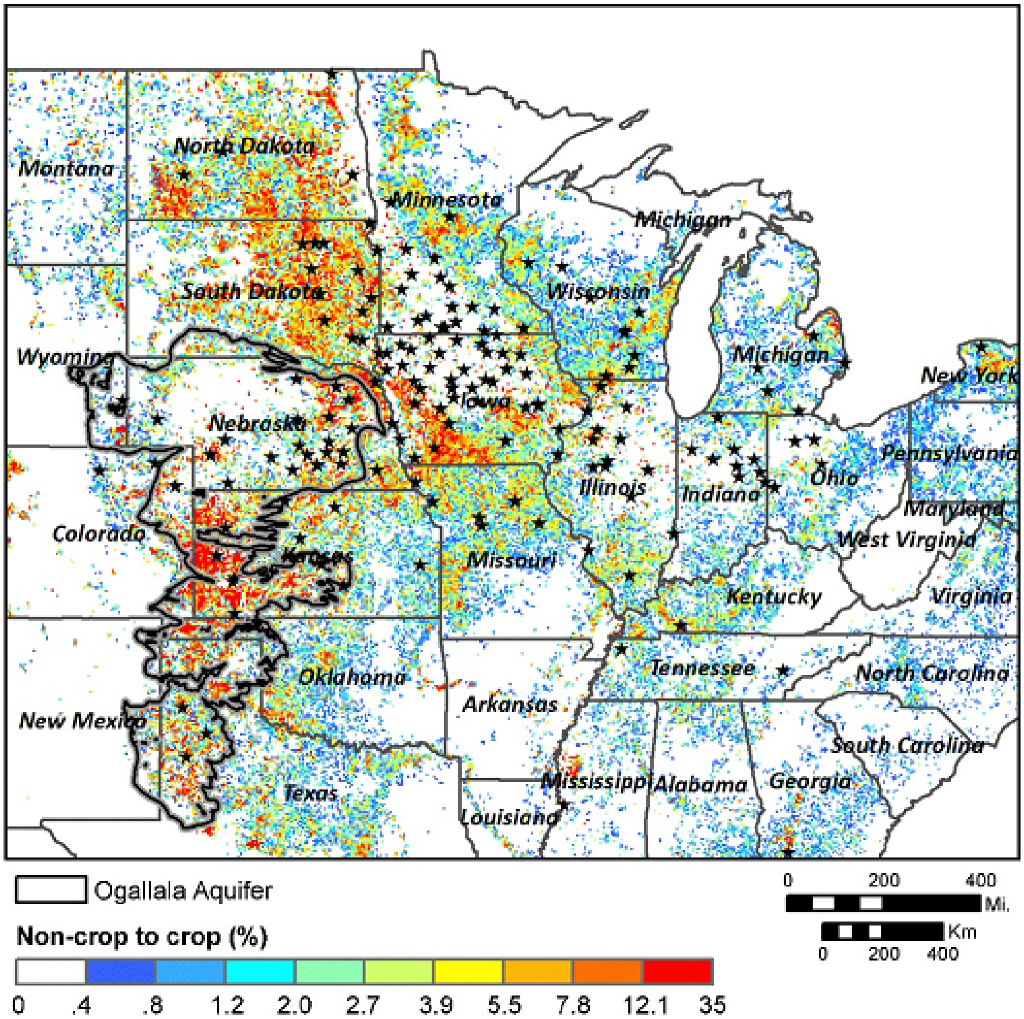

Figure 25 Relative conversion rates of arable non-cropland to cropland (2008-2012), including conversion located along the Ogallala aquifer. Stars denote biofuel production facilities. (Source: Wright et al. 2017)

3.3.4 Potential for Future Changes in Impacts to Water Quantity

Studies have examined the water-use implications of removal of corn stover and for future scenarios of perennial feedstocks (Demissie et al. 2012). Switchgrass, *Miscanthus*, and forest wood have high evapotranspiration (ET) rates, longer growing seasons, and therefore higher green water requirements, which can be important if looking at how these feedstocks could affect the broader hydrologic cycle when produced at a large scale. However, because of lower irrigation requirements, they are anticipated to have a smaller blue water footprint. Some studies (Demissie et al. 2012; Wu et al. 2012b; Cibin et al. 2016) have coupled multiple scenarios of feedstock production with watershed models, such as SWAT, to translate projected changes in land use/management – driven by demand for biofuel feedstocks – into changes in water demands. These watershed models often focus on changes in streamflow (see Table 5), along with indicators of water and soil quality (see Sections 3.2 and 3.5). Cibin et al. (2016) used a modified SWAT model (Trybula et al. 2015) that included improved representation of perennial bioenergy crops like *Miscanthus* and switchgrass and data from research plots. The model was used to assess the impacts of 13 biofuel scenarios for two watersheds in the Midwest. Cibin et al. (2016) found slight reductions in stream flow under biofuel production scenarios, ranging from 0.2% to 4.5%, with somewhat greater reductions for *Miscanthus*. They highlight that water use and water quality (e.g., nutrient removal) trade-offs need to be assessed carefully and that even with some reductions in stream flow "*Miscanthus* and switchgrass production may be a strong candidate for implementation in watersheds that would generally benefit from sediment and nutrient load reduction and can sustain base flows during drought conditions" (Cibin et al. 2016). These and other studies (Housh et al. 2015) indicate that while energy crops can reduce nitrate run-off they can reduce sub-surface water flows and streamflow. During low flow periods or drought, these reductions can have negative impacts on aquatic and riparian ecosystems. Thus, water quantity and water quality effects have potential trade-offs that should be carefully assessed in any cellulosic scenario.

Advances have also been made in development of publicly available tools for assessing future feedstock scenarios. An online web-based model WATER (Water Analysis Tool for Energy Resources)[77] characterizes county level water footprint for biofuel produced from corn, soybean, wheat, perennial grasses, and forest wood residue via various conversion processes for the U.S. The model presents a geospatial distribution of water consumptions (blue, green, and grey water) under historical and future land use scenarios. A number of studies have been based on this model. Most recently, DOE (2017) has underscored the importance of appropriate land management planning and choice of feedstock mix, including use of non-agricultural based feedstocks. Under a highly optimistic scenario of high-yields and production for both agricultural and wood-based feedstocks, and shifts toward non-irrigated perennial crops, they suggest that the states in Ogallala Aquifer region could actually reduce irrigation water consumption if planned and managed carefully (DOE 2017).

In terms of biofuel-processing water use, cellulosic ethanol facilities are anticipated to be more water intensive at first, ranging from 6 to 10 gallons of water per gallon of ethanol, primarily based on process engineering studies (Davis et al. 2015). Looking ahead, data collection could better quantify water use efficiency for existing and future cellulosic biofuel conversion facilities, along with their water

[77] Access to the WATER model is available at: http://water.es.anl.gov/

source and water demands relative to local water availability, particularly for potential hot spots of high water demands in water stressed areas.

3.3.5 Conclusions: Water Quantity

- As discussed in the 2011 Report, the irrigation of corn and soybeans grown for biofuels is the predominant water quantity impact. Water use for feedstock production is significantly greater than water use in the biofuel conversion process.
- There are indications of increased water consumption in irrigated areas for corn between 2007 and 2012 and elevated rates of land use change to corn production in more arid Western states including the Ogallala region. Adverse water availability impacts will most likely arise in already stressed aquifers and surface watersheds.
- Irrigation practices are dependent on a number of economic and agronomic factors that drive land management practices making attribution of increased irrigation and water quantity to biofuels difficult.

3.3.6 Research Needs: Water Quantity

- Studies are needed to determine the extent to which increases in water consumption and withdrawals – due to changes in land use/management change – can be attributed to feedstock production for biofuels.
- In particular, studies should continue to explore increases in water demands that have occurred or are occurring along water-stressed areas, both for surface and groundwater.
- Research, both modeling and field work to verify modeling parameters, is needed to better understand future cellulosic feedstock water demands while assessing water quantity, water quality, and soil quality in an integrated manner.

3.3.7 Opportunities for Future Environmental Improvements

- Priority should be placed on identifying effective strategies to manage withdrawals in “hot spots” (e.g., Ogallala aquifer) where high water demands and water stress are coinciding.
- While cellulosic feedstocks such as perennial grasses can provide environmental benefits for biodiversity and ecosystem services, their potential impact on streamflow within a watershed should be carefully considered.

3.4 Ecosystem Health and Biodiversity

Diverse biological communities are crucial to establishing and maintaining healthy ecosystems, as each species fulfills unique and necessary roles for maintaining ecosystem function. Ecosystem health can also be viewed in terms of resilience – the ability to resist external stressors over relevant temporal and spatial scales. The 2011 Report concluded that a variety of environmental factors related to biofuel production can affect ecosystem health and biodiversity, including changes to land use and land management, especially with regard to crop management and runoff from nutrients, pesticides, and sediment. Furthermore, the 2011 Report noted that overuse or misuse of management techniques can impact biodiversity and ecosystem health far beyond the confines of the farm field, potentially causing lasting impacts up and down the production chain (EPA 2011).

3.4.1 2011 Report Conclusions

The 2011 Report addressed terrestrial and aquatic biodiversity and ecosystem health with respect to grasslands, forests, wetlands, and impacts to aquatic systems (EPA 2011). In general, biofuel feedstock production was found to negatively impact biodiversity through loss of habitat, often in sensitive areas, and especially if idled lands in the Conservation Reserve Program (CRP) (with established conservation covers) were to be returned to crop production. Quantitative data linking biofuel production and ecosystem health remain sparse. Most of this information is qualitative in nature and often regional, and as such, merits broader research. The following paragraphs detail some of the most supported conclusions from the 2011 Report, including topics such as grasslands, forests, and feedstock management.

Grasslands are at the forefront of conversations surrounding biofuel production landscapes. Conversion of grasslands to row crops has been found to displace species reliant on grassland habitats whereas retaining some grasslands for perennial grass feedstocks could mitigate this loss of habitat (EPA 2011). Furthermore, using grasslands for buffer zones could reduce erosion and runoff, thereby reducing the likelihood of exposure to nutrients, pesticides, or other chemicals at levels above those determined to be protective.[78] The 2011 Report also noted that, while some feedstock management practices exacerbate the release of sediment, nutrients, pesticides, and pathogens into downstream waters, other more conservation-based practices (e.g., constructed or restored wetlands) can increase habitat availability for certain freshwater species. Releases or discharges with high concentrations of nutrients, total suspended

[78] Conservation Buffers to Reduce Pesticide Losses, USDA Natural Resources Conservation Service, March 2000 http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_023819.pdf

solids (TSS), and other contaminants can decrease ecosystem health and lead to fewer sensitive species in affected bodies of water, especially in areas where streamflow is already low (EPA 2011).

In terms of terrestrial impacts linked to biofuel production, the 2011 Report focused on how changes to forest harvests could affect biodiversity. The 2011 Report noted that shortening the harvest interval for short rotation woody crops and residue harvesting could decrease habitat availability and biodiversity, while moderate thinning could increase species diversity and abundance for certain species. However, given that the United States does not yet have woody biomass-based feedstock in production at the commercial scale, this topic is left for future research.

3.4.2 Drivers of Impacts to Ecosystem Health and Biodiversity

As concluded in the 2011 Report, ecosystem health and biodiversity are impacted by environmental factors, such as changes in land use and land management, including cropland extensification, cropland conversions and intensification, and nutrient, pesticide, and sediment runoff (EPA 2011). In addition to these site-specific factors, some environmental health indicators are associated with the spatial and structural arrangement of different habitat types across the landscape. The location of biofuel production and types of management practices employed affect the ecosystem impacts and potential mitigation opportunities.

Wright et al. (2017) reported that approximately 2 million acres of grassland within the standard draw of a biorefinery plant (50 miles) were converted to row crops between 2008 and 2012. Smaller acreages were reported as converted to row crops for forests (60,000 acres), shrublands (52,000 acres), and wetlands (14,000 acres). Figure 26, modified from Wright et al. (2017), illustrates the geospatial distribution of these conversions. The bulk of the grassland conversions occurred in South Dakota (348,000 acres), Iowa (297,000 acres), Kansas (256,000 acres), Missouri (239,000 acres), Nebraska (213,000 acres), and North Dakota (176,000 acres). The conversion reported by Wright et al. (2017) explicitly included only lands that had not been in cropland for at least 20 years, so although they may not represent pristine habitats they are expected to represent habitats in a relatively natural state.

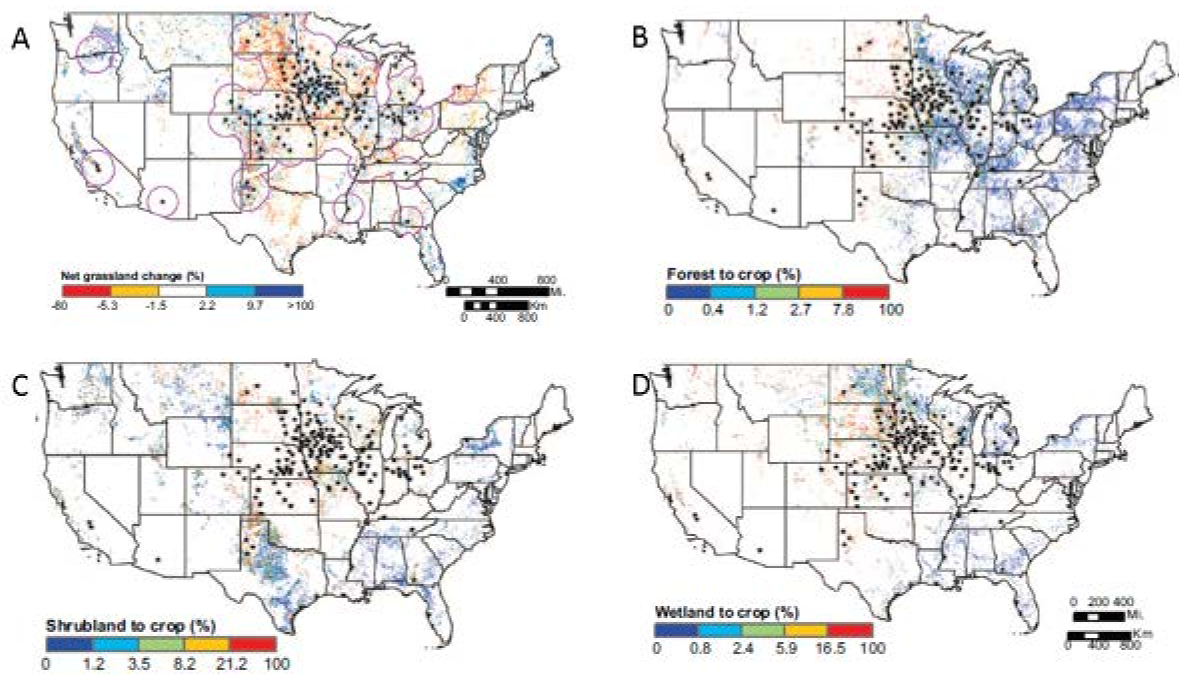

Figure 26 Relative conversion rates to cropland of (a) grassland, (b) forest, (c) shrubland, and (d) wetland from 2008 to 2012. Each rate is relativized by type of ecosystem within a 3.5-mile spatial grid (modified from Wright et al. 2017).

The row crop expansion and intensification is correlated with higher chemical inputs, including fertilizer and pesticides (Meehan et al. 2011; Meehan et al. 2015). Each of these classes of chemicals has the potential to impact ecosystem health and biodiversity (Malaj et al. 2014; Chagnon et al. 2015). Corn receives substantial levels of fertilizer, while soybeans typically are not fertilized. Corn and soybean, the dominant feedstocks for biofuels currently, are both treated with herbicides, fungicides, and insecticides. Neonicotinoid seed treatments (including thiamethoxam, clothianidin, and imidacloprid) are widely applied to biofuel feedstock crops, including corn and soybeans.[79] Approximately 90% of corn (Douglas et al. 2016) and 30% of soybean fields planted during 2008-2012 contained neonicotinoid seed treatments.[80] Detection of neonicotinoids in aquatic systems in regions of high corn and soybean

[79] Pesticides can be sold or distributed only after EPA approval and users must follow directions specified on the pesticides label to ensure safe use. Prior to being approved for use, EPA assesses a wide variety of potential human health and environmental effects associated with use of each pesticide, based on scientific data pertaining to the composition, potential adverse effects, and environmental fate of each pesticide. See https://www.epa.gov/pesticide-registration/about-pesticide-registration.

[80] Benefits of Neonicotinoid Seed Treatments to Soybean Production (2014). U.S. EPA memorandum, Office of Chemical Safety and Pollution Prevention. https://www.epa.gov/sites/production/files/2014-10/documents/benefits_of_neonicotinoid_seed_treatments_to_soybean_production_2.pdf.

production has raised concerns about the effects of neonicotinoids on aquatic communities and ecosystems (Hladik et al. 2014; Hladik et al. 2015; Miles et al. 2017). Proper application can reduce the risks of adverse environmental effects. For example, EPA's risk assessment for imidacloprid found the lowest overall aquatic risk profile for aquatic invertebrates when using seed treatments, although risks were still identified with some use scenarios. Alternatively, soil application was found to exceed the acute risk level of concern for over half of the agricultural and non-agricultural use scenarios modeled (EPA 2016b).

3.4.3 Impacts to Ecosystem Health and Biodiversity

3.4.3.1 Grassland Birds and Ducks

Widespread changes in land use for biofuel production (e.g., the conversion of environmentally sensitive land to cropland) have negative impacts to ecosystem health and biodiversity [see review by Immerzeel et al. (2014)]. The production of other forms of energy also have negative impacts on ecosystem health and biodiversity. As noted above, assessment of the environmental impacts of other energy sources is beyond the scope of this report. The type and severity of impacts depend on factors such as crop type, geographic location, and management practices. For example, degradation and loss of grasslands has been found to adversely affect grassland bird populations (Meehan et al. 2010; Fletcher et al. 2011; Robertson et al. 2011; Robertson et al. 2012; Blank et al. 2014; Werling et al. 2014; Evans et al. 2015). Studies of the effects of bioenergy feedstock production suggest that grassland bird species of conservation concern are more adversely affected by increased corn production than are more common species of birds (Fletcher et al. 2011; Blank et al. 2014). Similarly, the loss of wetlands to row crops and related production practices is associated with reduced duck habitat and productivity of duck food sources, including aquatic plants and invertebrates (Gleason et al. 2011; Wright et al. 2013). Increasing grassland cover by planting perennial grasses, including biofuel feedstocks, and replacing marginal croplands can also enhance ecosystem services, including pollination and biological control (Bennett et al. 2014a; Bennett et al. 2014b; Werling et al. 2014; Landis et al. 2017).

3.4.3.2 Pollinators

Pollinators, such as wild and commercial bees, are also affected by land use changes (e.g., forage loss from grassland conversion to corn and soybeans), among other pressures (National Research Council 2007). Commercial bees in the Midwestern and Great Plains states experience implications for honey production and colony health (Koh et al. 2016; Otto et al. 2016; Smart et al. 2016). Based on model estimates, these states include counties with large areas of pollinator-dependent crops and low levels of bee abundance (see Figure 27) (Koh et al. 2016). The expansion of corn and soybeans results in landscape simplification that exacerbates insect pest pressure and is linked to increased use of insecticides, such as neonicotinoids (Meehan et al. 2011; Meehan et al. 2015). Exposures of pollinators to neonicotinoids has been evaluated for the potential to cause adverse impacts on pollinators (Krupke et al. 2012; Krupke et al. 2015; EPA 2016b; EPA 2017) and other non-target organisms (Bonmatin et al. 2015; Pisa et al. 2015; van der Sluijs et al. 2015). Neonicotinoids also travel through the soil food chain and detrimentally affect beneficial arthropods, disrupt biological control of crop pests, and reduce soybean yields (Seagraves et al. 2012; Douglas et al. 2015a; Douglas et al. 2016).

The concerns about possible impacts to pollinators due to neonicotinoid exposure have been widely discussed in recent years. While it is beyond the scope of this report to review those studies, Godfray et al. (2014) summarize the issues well:

> "There is clear evidence of the great value of neonicotinoids in agriculture as well as the importance of the ecosystem services provided to agriculture by managed and wild pollinators. Pollinators also have intrinsic importance as components of natural biodiversity that cannot, or can only inexactly, be accorded economic value. In some cases, intelligent regulation of insecticide use can provide 'win-wins' that improve both agricultural and biodiversity outcomes but in other cases there will be trade-offs, both within and between different agricultural and environmental objectives. Different stakeholders will quite naturally differ in the weightings they attach to the variety of

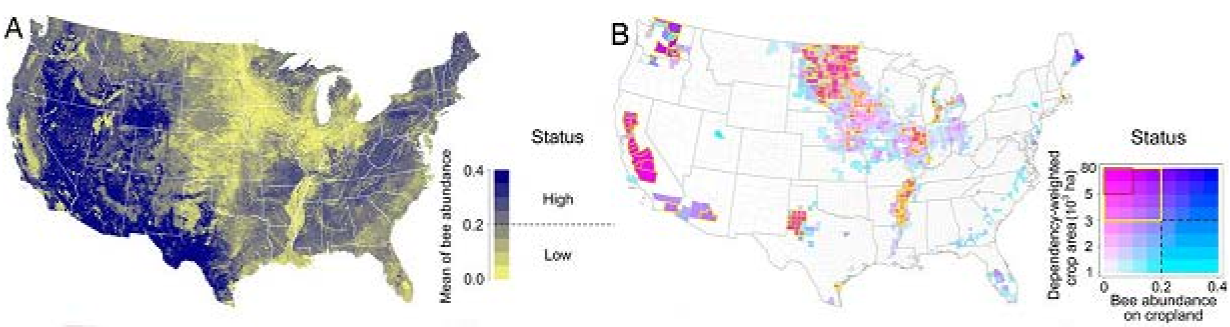

Figure 27 Map of (A) wild bee status and (B) status of wild bee supply vs. demand for pollination services (summed area of animal-pollinated crops, weighted by respective pollinator dependence) across coterminous U.S. (modified from Koh et al. 2016). The yellow border lines in B identify counties that have high acreage of pollinator dependent crops (y axis on the legend figure) but low bee abundance in crop land (x axis on legend figure).

objectives affected by insecticide use, and there is no unique answer to the question of how best to regulate neonicotinoids, an issue that inevitably has both economic and political dimensions."

In their review, Godfray et al. (2015) concluded that there is evidence of adverse impacts to pollinators due to neonicotinoid exposure but that the evidence is mixed, with several studies reporting no effects of exposure. They note that major gaps remain in our understanding of "how pollinator colony-level (for social bees) and population processes may dampen or amplify the lethal or sublethal effects of neonicotinoid exposure and their effects on pollination services" (Godfray et al. 2015).

EPA's preliminary assessment of the risk to bees from imidacloprid, clothianidin, and thiamethoxam found on-field risk to be low for these pesticides applied to corn, which is the dominant use pattern for this crop (EPA 2016b; EPA 2017). For other biofuel crops (e.g., soybeans), risks were considered uncertain at the time and are currently undergoing re-evaluation by EPA with the submission of additional exposure and effects data. Neonicotinoids, like all pesticides, are approved for use under specific conditions that are designed to protect ecosystems and human health. EPA has expanded its pesticide risk assessment process specifically for bees to quantify or measure exposures and relate them to effects at the individual and colony level.[81]

Crop intensification also influences the population dynamics of pollinators and pest organisms. Loss of milkweed in agricultural fields from the increased adoption of herbicide-tolerant corn and soybeans and related usage of herbicides negatively affect monarch butterfly populations in Midwest landscapes (Pleasants et al. 2013). Extensive adoption of transgenic corn and intensification are reported as primary drivers of the resistance of western corn rootworm, *Diabrotica virgifera virgifera* LeConte, and corn earworm, *Helicoverpa zea* Boddie, to multiple Cry proteins expressed in Bt corn (Dively et al. 2016; Gassmann et al. 2016; Jakka et al. 2016). Fausti (2015) noted that U.S. biofuel energy policy was a key contributor for rapid Bt corn adoption in U.S. corn production system. Through analysis of causal relationships, Fausti (2015) reported that 7-9% of the increase in Bt corn acres was induced by biofuel policies. The evolution of insect resistance to transgenic corn varieties, given the increased adoption of transgenic corn and intensification, hence, is an ecological impact that may be partially associated with increased biofuel feedstock production. To help manage western corn rootworm resistance, EPA recently announced enhancements to its long-standing requirements for companies that supply Bt corn to

[81] How We Assess Risks to Pollinators, U.S. EPA, https://www.epa.gov/pollinator-protection/how-we-assess-risks-pollinators; accessed March 11, 2018.

implement integrated pest management programs in the Corn Belt, including measures such as crop rotation to an alternate non-corn rootworm host crop (typically soybean).[82]

3.4.3.3 Aquatic Ecosystems

The effects of biofuel crop production on aquatic ecosystems are understudied in comparison to terrestrial ecosystems, partly due to a lack of monitoring data on aquatic species (Immerzeel et al. 2014). Crop expansion in the ecologically-sensitive Prairie Pothole region resulted in significant loss of wetlands and their associated biodiversity (aquatic plants and invertebrates) and ecosystem services, such as surface water flow, groundwater recharge and reduction in sedimentation (Gleason et al. 2011; Wright et al. 2013; Johnston 2014). Models predict that changes in hydrologic and sediment generation dynamics through land use change – mainly conversion to row crops – may extirpate native mussel populations due to shifts in river ecology in the Minnesota River Basin (southern Minnesota, parts of South Dakota, Iowa, and North Dakota) (Foufoula-Georgiou et al. 2015).

Increased applications of the pesticides imidacloprid and atrazine resulting from corn and soybean expansion/intensification have also been shown to have aquatic ecological effects. EPA (2016b) generated acute and chronic risk quotients (RQs) for aquatic organisms by modeling risks to aquatic organisms from agricultural uses of imidacloprid.[83] Aquatic invertebrates were correlated with the greatest risk from imidacloprid, where foliar spray and combination applications resulted in acute RQs ranging from 1.6 to 44, with the chronic RQs ranged from 39 to 2130, which are above the level of concern for acute and chronic risk (0.5 and 1.0, respectively). An aquatic exposure assessment for atrazine combined modeling approaches and monitoring data to estimate atrazine occurrence in surface water at different spatial scales (EPA 2016c). The report found that, on an acute exposure basis, atrazine is moderately toxic to freshwater and estuarine/marine fish, highly toxic to freshwater aquatic invertebrates, and even more toxic to estuarine/marine aquatic invertebrates. Effects on survival, growth, and/or reproduction were also shown from chronic exposure studies for freshwater and estuarine/marine fish, aquatic phase amphibians, and aquatic invertebrates. The risks from atrazine application and

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[82] US EPA: Regulation of Biotechnology under TSCA and FIFRA. Framework to delay corn rootworm resistance. https://www.epa.gov/regulation-biotechnology-under-tsca-and-fifra/framework-delay-corn-rootworm-resistance.

[83] EPA uses a deterministic approach or the quotient method to compare toxicity to environmental exposure. In the deterministic approach, a risk quotient (RQ) is calculated by dividing a point estimate of exposure by a point estimate of effects. This ratio is a simple, screening-level estimate that identifies high- or low-risk situations. See https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/technical-overview-ecological-risk-assessment-risk.

exposure to biota are not limited to corn and soybeans used for biofuels. Attributing these risks to specific crops faces the same attributional challenges as other endpoints (see Attribution Box 3, Section 2.4). Nevertheless, since over 80% of atrazine use is for corn, according to USGS,[84] and some fraction of corn production is attributable to biofuels, a direct link is present even if not explicitly quantified.[85]

3.4.4 Key Points from Recent Literature

Recent literature has emphasized: (1) impacts to biodiversity and ecosystem health due to the conversion of environmentally-sensitive lands; (2) the loss of ecosystem services, such as groundwater recharge, reduction in sedimentation, nutrient cycling, biological control of crop pests, and pollination; and (3) the need for better environmental data collection and monitoring (Newbold et al. 2015; Landis et al. 2017). Field studies and simulation models report that increased corn and soybean production often leads to the loss of grasslands (native mixed and tallgrass prairie), which negatively impacts ecosystem services, including pollination, biological control of crop pests, and nutrient cycling. Increasing grassland cover by planting perennial grasslands as biofuel feedstock and by replacing marginal croplands can enhance biodiversity and ecosystem services (Bennett et al. 2014a; Bennett et al. 2014b; Werling et al. 2014; Koh et al. 2016; Landis et al. 2017). Modeling approaches that incorporate both economic factors and ecosystem services frameworks, such as those popularized by the Millennium Ecosystem Assessment,[86] may be of benefit to better understand how impacts to ecosystem health and biodiversity can affect the broader economy.

3.4.5 Potential for Future Changes in Impacts to Ecosystem Health and Biodiversity

Given the trends in biofuel feedstock production and technology development discussed in Chapter 2, relatively few near-term changes in direction are expected for biodiversity and ecosystem health. A decrease in CRP lands could lead to decreases in biodiversity and terrestrial ecosystem health. This affects habitat availability as well as species diversity and abundance. More effective agricultural management practices would reduce incidences of sedimentation and eutrophication in streams and rivers, and increase streamflow levels. Reduced use of insecticides and genetically engineered species could lead to an increase in biodiversity. Increased adoption of pollinator conservation practices could

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[84] USGS National Water Quality Assessment Program: Pesticide National Synthesis Project. State-level pesticide use estimates by major crop and crop groups. https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/.

[85] US EPA. 2017. Refined Ecological Risk Assessment for Atrazine. External Review Draft. Environmental Fate & Effects Division, Office of Pesticide Programs, Washington, DC.

[86] http://www.millenniumassessment.org/en/Framework.html.

benefit pollinators and other beneficial insects.[87] Such changes, however, would require widespread coordination and adoption to be effective at regional and/or national scales.

Future improvements to and expansion of conservation practices related to biofuel feedstock production could play a major role in reducing the magnitude and severity of impacts to ecosystem health and biodiversity. For instance, the use of precision agriculture, as well as guidance systems, has become more prevalent over the past 20 years.[88] Increased adoption of fertilizer technologies such as time-release and other enhanced-efficiency fertilizers, alternative fertilizer placement methods, and variable-rate application could further improve nutrient-related conservation practices in biofuel feedstock production.

3.4.6 Conclusions: Ecosystem Health and Biodiversity

- Loss of grasslands and wetlands is occurring in ecologically sensitive areas, including the Prairie Pothole Region.
- Loss of habitat and landscape simplification are associated with negative impacts to pollinators, birds, soil-dwelling organisms, and other ecosystem services in both terrestrial and aquatic habitats.
- Increased fertilizer applications of nitrogen and phosphorus have negative effects on aquatic biodiversity.

3.4.7 Opportunities for Future Environmental Improvements: Ecosystem Health and Biodiversity

- Planting perennial grasslands and replacing marginal croplands with perennial grasslands can enhance ecosystem services.
- Increased use of effective conservation practices can have multiple benefits, from reduced stream sedimentation and nutrient runoff to protection of pollinator habitat.
- Increased adoption of other technologies such as time-release and other enhanced-efficiency fertilizers, alternative fertilizer placement methods, and precision agriculture could further improve conservation practices in biofuel feedstock production.

[87] Using 2014 Farm Bill Programs for Pollinator Conservation (2015). USDA Biology Technical Note No. 78 (2nd Ed.) https://directives.sc.egov.usda.gov/OpenNonWebContent.aspx?content=37370.wba.

[88] USDA Economic Research Service (2017). Tailored Reports: Crop Production Practices; https://data.ers.usda.gov/reports.aspx?ID=17883

3.4.8 Research Needs: Ecosystem Health and Biodiversity

- Studies that target the interactive effects of land use change and feedstock production could help identify impacts to specific organisms.
- Research on the efficacy of methods to expand pollinator habitat in agricultural systems can improve understanding of appropriate methods and their potential tradeoffs for different agricultural areas.

3.5 Soil Quality

The production of biofuel feedstocks can also affect soil quality, which is the capacity of a soil to function.[89] The EPA's 2011 Report focused on soil erosion, soil organic matter (SOM), and soil nutrients as general indicators of soil quality (EPA 2011). Soil erosion can impact soil quality by preferentially removing the finest soil particles at the soil surface that are generally higher in organic matter, plant nutrients, and water-holding capacity than the remaining soil. Soil organic matter is critical to soil quality because it provides plant nutrients and water, promotes soil structure, and reduces erosion, while also sequestering carbon from the atmosphere (Sparks 2003).[90] Lastly, soil nutrients (e.g., nitrogen, phosphorus) are necessary for plant growth. Too little of these nutrients can reduce crop yields; too much can lead to eutrophication of waterways via runoff or leaching.

3.5.1 2011 Report Conclusions

Overall, the 2011 Report concluded biofuel feedstock production could either negatively or positively affect soil quality depending upon the feedstock used, the particular land converted, and the management of the feedstock. For corn and soybeans, environmental effects were estimated to be most negative if these crops were produced on former CRP land or other relatively unmanaged grasslands. Conversely, the soil quality effects were estimated to be minimal if the feedstocks were produced on

[89]The USDA Natural Resources Conservation Service defines soil quality as "The capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation. In short, the capacity of the soil to function" (USDA-NRCS 2017). Here, soil conservation and soil environmental quality, as listed in Section 204 of the 2007 Energy Independence and Security Act (EISA), are subsumed under this broader heading of soil quality. The term soil quality in this section is used as a general term, independent of area—it is used both to describe effects on single soil types and cumulative effects across large areas and multiple soil types.

[90]Soil organic matter is defined by Brady et al. (2000) as "The organic fraction of the soil that includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil population." The main elemental constituents of SOM are carbon (52-58%), oxygen (34-39%), hydrogen (3.3-4.8%) and nitrogen (3.7-4.1%) (Sparks 2003).

land currently in corn and soybeans. Effects could be moderated by the use of conservation practices (e.g., no-till management). For corn stover, high removal rates were found to increase erosion and loss of SOM. Soil quality impacts of perennial grasses and woody biomass were estimated to be largely determined by the type of land converted, as in the case for annual feedstocks, whereas algae production was considered to have minimal-to-no-effects on soil quality.

In the following sections, we revisit these 2011 soil quality conclusions. First, we provide a brief overview of the major changes in the drivers of soil quality (feedstocks, type of land-converted, and production practices) and their impacts since 2011. We address these by feedstock type. Second, we highlight a few changes in our understanding of the connections between drivers and impacts since 2011. Third, we focus on potential future changes, and, finally, we provide a bulleted list of conclusions. As in 2011, we discuss effects on soil erosion, SOM, and soil nutrients, while also acknowledging it may be advantageous to add other soil quality indicators in future reports.

3.5.2 Drivers of Impacts to Soil Quality

Corn-grain ethanol and soy biodiesel account for most of the biofuel volumes produced to date. As a result, almost all the soil quality impacts from biofuels thus far are from the production of corn and soybeans. Since the 2011 Report, new evidence adds support to the understanding that grasslands, including CRP grasslands, have been converted to corn and soybeans (see Section 2.4). Biofuels are responsible for a proportion of this change, although, as noted in Box 3, the percentage attributable to biofuels cannot now be quantified with confidence, nor can the resulting effects on soil quality be quantitatively attributed to biofuel production. In general, however, grassland-to-annual-crop conversion negatively impacts soil quality because it increases erosion and the loss of soil nutrients and SOM, including soil carbon (Gregorich et al. 1985; Gelfand et al. 2011; Qin et al. 2016; Yasarer et al. 2016). These in turn increase sediment and nutrient loadings to waterways and carbon loss to the atmosphere (Lal 2003; Yasarer et al. 2016).

A couple of factors can mitigate—to an extent, but not entirely—the negative soil quality impacts of this land use conversion. First, the type of CRP land, conservation lands, or other grasslands converted can affect soil quality. In a modeling study, LeDuc et al. (2017) simulated that more erosion and loss of soil carbon and nitrogen occurs from converting low productivity, highly sloped CRP grasslands compared to those with higher productivity soils and lower slopes. Second, the effects will also depend upon production practices. Most corn and soybeans are grown using conservation tillage, with a smaller percent grown using no-till management (USDA-NRCS 2010; Wade et al. 2015).[91] Conservation tillage, including no-till, reduces soil erosion and increases SOM content relative to conventional tillage (Cassel et al. 1995; West et al. 2002). Use of conservation tillage practices can partially mitigate the effects of converting CRP areas or other grasslands to corn or soybeans (Follett et al. 2009; Gelfand et al. 2011).

Since 2011, evidence has become stronger that corn and soybeans are replacing other cropland, not just CRP land and other grasslands (see Section 2.4). The soil quality impacts of converting to corn or soybeans from other crops, such as wheat, are generally less than those of the conversion of grasslands (Zuber et al. 2015; Qin et al. 2016; Yasarer et al. 2016). Zuber et al. (2015) observed similar soil effects of no-till, continuous corn rotations, and corn-soybean-wheat rotations on high organic matter, fine textured soils. From this evidence, they suggest a movement from wheat to corn may not materially affect soil quality, provided a shift from no-till to conventional tillage does not occur concomitantly. Qin et al. (2016), in a meta-analysis, found that corn replacing other cropland (e.g., soybean, wheat) increased soil organic carbon, whereas the opposite occurred when corn replaced grassland or forest land. Notably, the percent increase in soil organic carbon of other-cropland moving to corn was exceeded in magnitude by the percent decrease in soil organic carbon by the conversion of grassland-to-corn (Qin et al. 2016).

In contrast to corn grain and soybeans, the use of other feedstocks for biofuels has been much more limited. Since 2011, at least two commercial-scale corn stover ethanol plants have started operations – the DuPont and POET-DSM corn stover plants in Iowa (EPA 2016d). Partial stover removal can increase corn yields in some locations, in part by reducing nitrogen uptake from the soil by microorganisms and potentially by increasing soil temperatures in no-till systems (Coulter et al. 2008; Karlen et al. 2014). Yet too much stover removal can increase soil erosion, decrease SOM and soil nutrients, and ultimately decrease corn yields as noted in the 2011 Report. Whether corn stover can be harvested sustainably, and at what removal rate, depends on many site-specific factors, including yields, topography, soil characteristics, climate, and tillage practices (Karlen et al. 2014). In a study across multiple locations in seven states, stover harvesting slightly increased corn grain yields, although the

[91]Conservation tillage is defined as any tillage practice leaving at least 30% of the soil surface covered by crop residues; whereas conventional tillage leaves less than 15% of the ground covered by crop residues (Lal 1997). No-till management, a subset of conservation tillage, disturbs the soil marginally by cutting a narrow planting strip. Nationally, approximately 30% and 45% of the area planted to corn and soybeans, respectively, are under no-till (Wade et al. 2015). Since 2000, there has been a general trend toward greater percent residue remaining after planting for both crops (USDA-ERS 2018 https://data.ers.usda.gov/reports.aspx?ID=17883; data accessed 2/15/2018)

authors cautioned against extrapolating these results to other sites and noted the need to conduct site-specific planning with soil testing (Karlen et al. 2014). DuPont limits harvesting stover to corn fields in a no-till or conservation tillage system, with yields of 180 bushels per acre or higher, and on relatively flat land with a slope of four percent or less (DuPont 2017). POET-DSM recommends harvesting only 20-25% of stover on slopes less than four percent, coupled with soil testing to monitor soil nutrients and SOM (POET-DSM 2017). These criteria are designed to encourage stover harvest at sustainable rates and locations, although additional research is needed to understand effects on soil quality if these criteria are followed.

In contrast, perennial grasses, woody biomass, and algae generally have not been used yet as biofuel feedstocks at the commercial scale, with a few exceptions (e.g., algal biofuels for the U.S. Navy; Ziolkowska et al. 2014). Therefore, there have not been major changes in the drivers and soil quality impacts from these feedstocks since the 2011 Report.

3.5.3 Changes in Relationships Between Drivers and Impacts

Since 2011, research has improved our understanding of the relationship between drivers and soil quality impacts. On the negative side, the scientific literature suggests there may be a relationship between no-till management and the loss of nutrients such as phosphorus to waterways (e.g., Jarvie et al. 2017; see Water Quality Section). On the positive side, a recent study suggests the carbon benefits of no-till corn may have been previously underestimated due to a failure to account for carbon accrual at greater soil depths (Follett et al. 2012). For corn stover, recent research has focused on the use of cover crops, manure, or biochar to add organic matter to the soil to compensate—at least partially—for the organic matter removed (Blanco-Canqui 2013).[92] The scientific literature continues to emphasize that perennial grasses or woody biomass grown on marginal lands (e.g., abandoned agricultural land) can help restore soil quality [e.g., Blanco-Canqui (2016)]. Notably, however, effects of these perennial feedstocks can depend upon the plant species grown and the type of land converted (Robertson et al. 2017), and literature definitions of what constitutes marginal land and estimates of its extent vary widely (Emery et al. 2016). Finally, the 2011 Report concluded algae production would have minimal-to-no-effects on soil quality. It is possible, however, that some of the algal residues following oil extraction could be used as a soil amendment, increasing soil carbon content (Rothlisberger-Lewis et al. 2016).

[92]Biochar is the product of heating biomass in the absence of or with limited air, with the resulting material rich in organic carbon (Lehmann et al. 2015). This material can be used as a soil amendment.

3.5.4 Potential for Future Changes in Impacts to Soil Quality

It is likely that corn and soybeans will be the predominant biofuel feedstocks grown in the near future, which is expected to continue to put pressure on soil resources. Leaps forward in biotechnology and/or increasing yields may ameliorate some of these concerns (e.g., Brusamarello-Santos et al. 2017). The two new, commercial-scale, corn stover plants now in operation could signal a beginning of a corn stover industry, and the use of soil amendments, such as biochar, may be expanded to counterbalance organic matter removal in both agricultural and forest soils (Blanco-Canqui 2013; Scott et al. 2016). Should the large-scale production of perennial grasses or woody biomass become economically viable as feedstocks, they would fundamentally alter effects on soil quality, most likely positively if grown on marginal lands or lands with soils otherwise limited by physical or chemical problems (Blanco-Canqui 2016).

3.5.5 Conclusions: Soil Quality

- Corn-grain ethanol and soy biodiesel account for most of the biofuel volumes produced to date. As a result, almost all the soil quality impacts from biofuels, thus far, are from the production of the dominant conventional feedstocks.
- Conversion of grasslands to annual cropland typically negatively affects soil quality, with increases in erosion, and the loss of soil nutrients and soil organic matter, including soil carbon. Impacts of this conversion can be partially mitigated – though not entirely – through the adoption of management practices such as conservation tillage.
- The soil quality impacts of converting other crops to corn or soybeans are generally less than those of the conversion of grasslands. The production of corn on existing cropland can provide soil carbon benefits, although these benefits are outweighed on a per area basis by the negative effects of grassland conversion.
- Overall, these land use trends suggest that negative impacts to soil quality from biofuel feedstocks have increased since 2011, but this has not been quantified and the magnitude of effects depends predominantly on the relative areas of grasslands converted versus existing croplands attributable to biofuels.
- Corn stover is now being harvested at the commercial-scale in Iowa, and the scientific literature indicates this must be done carefully to avoid negatively affecting soil quality and crop yields.

3.5.6 Opportunities for Future Environmental Improvements: Soil Quality

- Alternative biofuel feedstocks, such as perennial grasses and woody biomass, are not yet used at commercial scales. Studies have shown that these feedstocks can improve soil quality relative to current conditions, contingent on species grown and type of land converted (e.g., marginal, abandoned, or degraded lands).

3.5.7 Research Needs: Soil Quality

- Quantitative estimates of the cumulative soil quality effects are needed for the land use changes described in Section 2.4 and the proportion attributable to biofuel feedstock production (this includes both the conversion of land to corn and soybeans and the management practices implemented).

3.6 Invasive Species

The National Invasive Species Council defines an invasive species as "with regard to a particular ecosystem, a non-native organism whose introduction causes or is likely to cause economic or environmental harm or harm to human, animal, or plant health."[93] In the context of biofuels, and similar to the 2011 Report, this report also includes additional characteristics of species for evaluating environmental impacts and invasiveness.

3.6.1 2011 Report Conclusions

The 2011 Report noted that biological traits of some plant species and perennial grasses favored as biofuel feedstocks overlap with those of high invasion potential (fast growing species that form dense stands, efficiently use resources, tolerate broad environmental conditions and perturbations, are disease and pest resistant, and are able to disperse and establish widely) (EPA 2011). The 2011 Report listed several mitigation options for reducing the potentially negative environmental impacts from perennial grass production. Prominent options included conducting a weed risk assessment (WRA) and rejecting planting species or varieties that are predicted to be invasive. Under the RFS requirements and in collaboration with USDA, EPA examines invasion risk WRA and includes further regulatory requirements (e.g., a Risk Mitigation Plan) as needed to reduce the invasion potential and other negative

[93] Executive Order 13751, "Safeguarding the Nation from the Impacts of Invasive Species," December 5, 2016.

environmental impacts. The 2011 Report also concluded that corn, soybean, and perennial grasses, such as Giant Miscanthus, pose little invasive species risk (EPA 2011).

3.6.2 Drivers of Impacts to Invasive Species

The renewable volume obligations through 2016 have principally been met with renewable fuels from corn grain (ethanol) and soybean (biodiesel) as feedstocks, and these crops do not pose a risk of invasion in the U.S. The invasiveness of these crop species has not altered since the 2011 Report. To date, no cases of invasive corn or soybeans have been reported in natural areas in the U.S.

Recent studies have linked the increased adoption and extensive cultivation of corn and soybean that are genetically engineered to resist glyphosate and the widespread application of this herbicide (see Chapter 2) to development of glyphosate resistance in 15 weed species in total (Benbrook 2012; Heap 2014; Benbrook 2016; Myers et al. 2016). This results in increased alternative herbicidal treatments and higher active ingredient application per unit area, which further increases evolutionary pressure for resistance (Benbrook 2016; Myers et al. 2016) and other environmental impacts. Resistance to particular herbicides provides a fitness advantage to weeds in areas where direct or indirect exposure to those herbicides occurs, which may effectively enhance the potential invasiveness of weed species in certain habitats.

For other feedstocks, reports highlight the invasion potential of *Panicum virgatum* L. (switchgrass) in areas where the species are non-native within the U.S. For example, reports predict that in California, where it is potentially invasive, switchgrass could establish successfully in disturbed riparian areas (Barney et al. 2012). Comparison of different switchgrass cultivars with the wildtypes in Ohio and Iowa showed some cultivars performing better, supporting the need for further assessments prior to large-scale planting for biofuels (Palik et al. 2016). Previous reports also noted that giant miscanthus (*Miscanthus* × *giganteus*) posed little risk of invasion because it is sterile and propagated by cutting (Heaton et al. 2010; Gordon et al. 2011). However, not all *M.* x *giganteus* cultivars are sterile. Spatial demographic models indicate that sterile and fertile cultivars of *M.* x *giganteus* have substantially different invasive potential. Whereas frequent and severe habitat disturbances are predicted to raise invasion risk for feral populations of sterile *M.* x *giganteus*, fertile cultivars would likely be difficult to contain (Matlaga et al. 2013). Comparison of many noninvasive and invasive species of the genus *Miscanthus* in Virginia and Georgia showed that *M.* × *giganteus* is less likely to be invasive in conventional agricultural fields subject to tillage or herbicide applications (Smith et al. 2014). Recent results suggest that potentially invasive *Miscanthus* species could become established outside of cultivated areas, but a lag in any impacts on receiving communities presents a widow of time for management (West et al. 2017).

3.6.3 Potential Changes in Relationships Between Drivers and Impacts

Since the 2011 Report, EPA has approved pathways for feedstocks using *Camelina sativa* (Camelina), *Saccharum spp.* (energy cane), *Arundo donax* (giant reed), and *Pennisetum purpureum* (napier grass) (EPA 2013d; EPA 2013c). For the highly invasive giant reed and napier grass (Gordon et al. 2011; USDA 2012), approval requires a risk mitigation plan demonstrating these species will not pose a significant likelihood of spread beyond the intended planting area.[94] Additional registration, reporting, and record keeping requirements to address potential invasiveness are also required (EPA 2013c). Energy cane is a hybrid of different *Saccharum spp*. As *S. spontaneum* is on the Federal Noxious Weed List,[95] it is excluded as a potential feedstock although hybrids derived from *S. spontaneum*, developed and publicly released by USDA (Bischoff et al. 2008) are included in this definition of the energy cane feedstock. Among other approved species, the risk of invasion by *Camelina sativa* in the northern Great Plains region is low (Davis et al. 2011) based on a two-tiered approach (incorporating demographic models to field-estimated parameters in addition to weed risk assessment). These feedstocks are not yet used for commercial scale production of biofuels, which precludes monitoring and assessment of any additional invasion impacts. Other potentially invasive feedstocks that have been analyzed by EPA with respect to lifecycle greenhouse gas emissions include *Thlaspi arvense* (pennycress), *Jatropha curcas* (Jatropha), and *Brassica carinata* (Carinata),[96] but these have not yet been approved for RIN-generating renewable fuel production. Weed risk assessments of other potential biofuel species conclude that *Eucalyptus camaldulensis* and *Eucalyptus grandis* have high potential to become invasive in the U.S. (Gordon et al. 2011).

Because these advanced-generation biofuel feedstocks are not now used for commercial scale production of biofuels, the invasive impacts remain a potential, rather than current, risk. Thus, the full

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[94] U.S. Environmental Protection Agency, Office of Transportation Air Quality. Approved Pathways for Renewable Fuel. Policies and Guidance. https://www.epa.gov/renewable-fuel-standard-program/approved-pathways-renewable-fuel.

[95] U.S. Department of Agriculture, Natural Resources Conservation Service. Introduced, Invasive, and Noxious Plants. https://plants.usda.gov/java/noxious

[96]U.S. Environmental Protection Agency, Office of Transportation Air Quality. Other actions for the renewable fuel standard program. Policies and Guidance. https://www.epa.gov/renewable-fuel-standard-program/other-actions-renewable-fuel-standard-program.

invasive impacts of newly approved feedstocks and others with a completed lifecycle analysis remain unclear and unknown.

3.6.4 Potential for Future Changes in Impacts to Invasive Species

Recent genetic engineering of biofuels feedstock trees such as *Populus* spp. (poplar) and grasses, such as *P. virgatum* (switchgrass) and *Miscanthus* spp., have focused on improving the conversion of cellulosic feedstocks to ethanol. There have also been efforts to engineer improvements of both oil yield and oil quality for biodiesel from crops including *Glycine max* (soybean), *Brassica napus* (canola), *C. sativa*, and *J. curcas* (NAS 2016). So far, there are no data to indicate that these modifications are changing the invasiveness of the engineered crops. Nevertheless, there is well documented potential for gene flow to native populations or indigenous species from trees, grasses, and crucifers used to produce biofuels. Thus, crop protection genes that are engineered or bred into such feedstocks, along with those genes for improvement of other crop qualities, may also be introduced into recipient populations, depending on the ecological and management context of the species involved (DiFazio et al. 2012; Gressel 2015; NAS 2016; Chang et al. 2018). Future impacts from invasive species remain to be determined.

Studies suggest methodological advancements and strategies, including modifications to weed risk assessments, for improved evaluation of the invasion risk of biofuel crops (Davis et al. 2011; Hulme 2012; Lewis et al. 2014; Quinn et al. 2015a). This includes a ‘white-list’ approach for policy decisions on incentivizing the cultivation of promising new feedstocks without increasing the probability of non-native plant invasions in natural systems (Quinn et al. 2015a). Studies also point to shortcomings in the regulatory framework for weed management and stress the need to incorporate insights from other commercial industries (horticulture, forestry, agroforestry) to inform strategies to reduce environmental impacts due to invasive biofuel feedstocks (Richardson et al. 2011; Quinn et al. 2013; Quinn et al. 2015b).

3.6.5 Conclusions: Invasive Species

- Biofuels are primarily produced in the forms of bioethanol and biodiesel derived from food crops (i.e., non-invasive first generation biofuels – corn and soy). Hence, current production of biofuel feedstocks poses little risk of invasion, consistent with findings in the 2011 Report.
- Weed risk assessments, part of the formal biofuel regulatory process, provide information on invasion risk and are designed to inform protective management of species and varieties that are predicted to be invasive.

- Increased cultivation of crops engineered for herbicide tolerance (e.g., glyphosate) and concomitant application of the herbicide has led to a widespread increase in the number of glyphosate-resistant weed species.
- Potentially invasive species approved as feedstocks require risk management actions under current RFS requirements. However, invasive species are not presently being used for commercial scale production of biofuels.

3.6.6 Research Needs: Invasive Species

- Methodological advancements for weed risk assessments and lessons from other industries (e.g., horticulture) should be incorporated to inform on potential invasiveness of biofuel feedstocks.
- Modeling and field work are needed to investigate the impacts of gene flow between novel feedstock varieties (genetically engineered, selectively bred, or a combination) and local natives.

3.7 International Impacts

3.7.1 2011 Report Conclusions

The 2011 Report showed that in the global context, biofuel demands have direct and indirect impacts for biofuel-producing countries as well as those importing agricultural commodities. Potential environmental impacts included direct and indirect effects from land use change and impacts on air quality, water quality, and biodiversity. This section focuses on the potential environmental impacts in foreign countries from implementation of the RFS2 standards in the United States. Simulations prepared for the RFS2 projected that the EISA biofuel targets could alter U.S. and international trade patterns and commodity prices (EPA 2010). The manner in which countries respond to U.S. market conditions, including influences from deforestation, and biofuel feedstock crop expansion and intensification could affect net GHG savings derived from biofuels.

The 2011 Report anticipated import volumes to be very low in years preceding 2015, followed by a significant increase in import volumes between 2015 and 2022. Similarly, it anticipated a decrease in exports of corn and soybeans for agricultural or other uses, probably resulting in land use change through conversion to agriculture in other countries, and other environmental impacts. As with biofuel production in the U.S., these impacts depend largely on where the crops are grown, forest and agricultural management practices and technologies used, and the efficacy of environmental policies. Therefore, land use changes in other countries and other environmental impacts could not be quantified.

However, the 2011 Report noted that, if natural ecosystems are converted to cropland, the environmental impacts may be more severe.

3.7.2 Drivers of International Impacts

The volume and location of U.S. imports and exports, both of biofuel and displaced agricultural goods through international trade, affect the severity of the direct and indirect land use impacts.

3.7.2.1 Trends in Annual U.S. Imports

Ethanol imports have decreased significantly since 2012 (see Table 6), as predicted in the 2011 Report, likely due to increased U.S. ethanol production, limitations on U.S. demand, and other economic and policy factors. Actual import volumes were much lower than the estimates from the 2011 Report (compare with Figure 5-2 in 2011 Report). Brazil has been the dominant source of ethanol between 2011 and 2016, and the overall decrease in imports is largely due to the decrease in imports from Brazil. Before 2015, significant volumes of ethanol were reported as being imported from other countries in South America and the Caribbean; however, these volumes were likely produced in Brazil and imported through the Caribbean Basin Initiative. Although the EIA does not publish ethanol import data by feedstock, the vast majority of the ethanol imported from Brazil was likely produced from sugarcane.[97]

Table 6. Annual U.S. ethanol imports by country of origin (million gallons)[1]

Year	Brazil[2]	Other Latin America[3]	Canada	EU	Total
2011	101	69	2	-	172
2012	404	82	4	4	494
2013	322	50	5	-	377
2014	56	11	5	2	74
2015	88	-	3	-	92
2016	36	-	1	-	36

1. Source: https://www.eia.gov/dnav/pet/pet_move_impcus_a2_nus_epooxe_im0_mbbl_a.htm, 4/28/17
2. Volumes of ethanol imported from Brazil have demonstrated substantial variability since 2011; it is unclear whether the decreasing trend since 2011 will be maintained or will return to the higher values observed in earlier years.
3. Other Latin America includes: Ecuador, Argentina, Costa Rica, El Salvador, Guatemala, Jamaica, Nicaragua and Trinidad and Tobago

[97] USDA, Brazil Biofuels Annual 2017, https://gain.fas.usda.gov/Recent%20GAIN%20Publications/Biofuels%20Annual_Sao%20Paulo%20ATO_Brazil_9-15-2017.pdf.

In contrast, biodiesel imports have increased in recent years (see Table 7), reaching almost 700 million gallons in 2016. Imports from Argentina, which are likely soybean oil biodiesel, more than doubled from 2015 to 2016, reaching almost 450 million gallons. The second largest import country of origin in 2016 was Indonesia, where palm oil is the dominant feedstock. Although EPA has not approved a pathway for the production of palm oil biodiesel that would meet the minimum 20% lifecycle greenhouse gas reduction requirement, Indonesian biodiesel imports may include grandfathered volumes that are nevertheless eligible as conventional biofuel under the RFS program. Imports from Canada, which reached almost 100 million gallons in 2016, are likely from a combination of canola/rapeseed oil, soybean oil, and waste oils, such as used cooking oil and inedible tallow.[98]

Table 7. Annual U.S. biodiesel imports by country of origin (million gallons)[1]

	Argentina	Canada	EU	Indonesia	Other[2]	Total
2011	0	11	5	0	4	20
2012	0	18	10	0	8	36
2013	132	45	88	52	25	342
2014	52	71	8	59	3	192
2015	196	61	3	72	21	353
2016	444	98	25	102	24	693

1. https://www.eia.gov/dnav/pet/pet_move_impcus_a2_nus_EPOORDB_im0_mbbl_m.htm, 4/28/17.
2. Other: Australia, Korea, Panama, Singapore and Taiwan.

Over 200 million gallons of renewable diesel were imported from Singapore in both 2015 and 2016, as shown in Table 8. These volumes were likely drop-in renewable diesel produced through hydrotreating of fats and oils, including waste and vegetable oils. The increase in biodiesel and renewable diesel imports[99] could have resulted in direct and indirect land use changes and other associated environmental impacts in some of the trading nations.

[98] USDA, Canada Biofuels Annual 2017, https://gain.fas.usda.gov/Recent%20GAIN%20Publications/Biofuels%20Annual_Ottawa_Canada_8-9-2016.pdf.

[99] Renewable diesel and biodiesel, which differ chemically, are both included as 'Biomass-based diesel.' Non-ester renewable diesel is produced through hydrotreating, thermal conversion or biomass-to-liquid, and can be used in its pure form, or as an additive. Biodiesel (mono-alkyl esters) is produced using a transesterification process. For the RFS program implementation, EPA utilizes 'Equivalence Values' based on energy content (renewable diesel - 1.7 & biodiesel - 1.5) for determining RIN generation.

Table 8. Annual U.S. renewable diesel imports by country of origin (million gallons)[1]

	Aruba	Finland	Singapore
2012	2	14	9
2013	6	36	164
2014	0	9	111
2015	0	0	205
2016	0	0	223

1. https://www.eia.gov/dnav/pet/pet_move_impcus_a2_nus_EPOORDO_im0_mbbl_m.htm, 4/28/17.

3.7.2.2 Trends in Annual U.S. Exports

Corn exports reduced from about 61,000 metric tons in 2007 to 20,000 metric tons in 2012. However, since 2012 exports steadily increased to about 56,500 metric tons in 2016. Exports of brewers and distillers dregs and waste, sometimes known as distillers dried grains with solubles (DDGS), have also been on the rise since 2012 (see Figure 28). Soybean oilseed exports were similar, ranging between 35,000 to 41,000 metric tons during 2007-2012, but have increased since 2012 (see Figure 29).

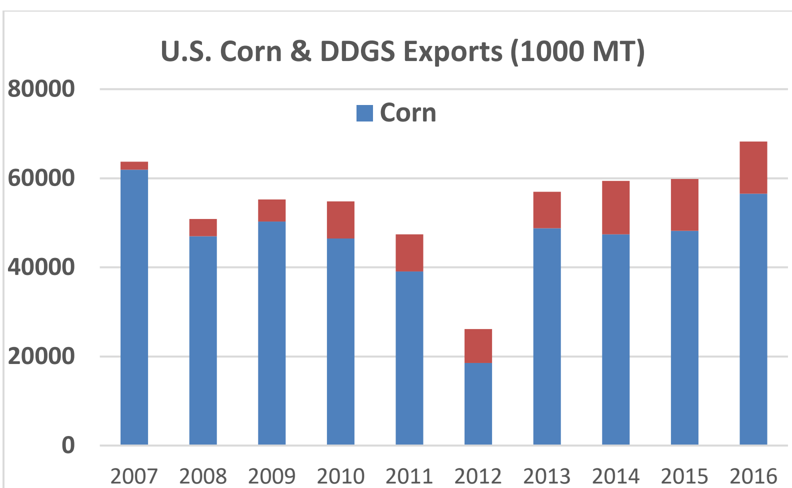

Figure 28 Trends in annual metric tons of U.S. exports of corn and brewers' and distillers' dregs and waste (DDGS).[92]

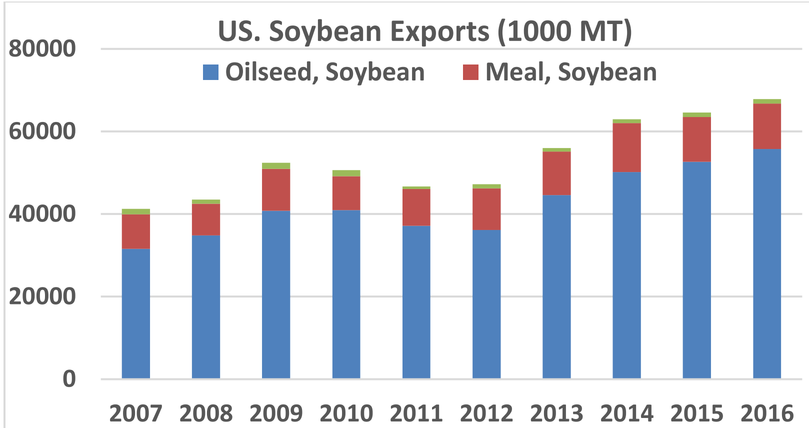

Figure 29 Trends in annual metric tons of U.S. exports of soybeans[100]

U.S. ethanol exports varied during 2010-2013, with a peak in 2011. Since 2013, exports have increased from 15 million barrels to 25 million barrels (see Figure 30). For biodiesel, exports increased until 2013 to 4.6 million barrels, then decreased to 2 million barrels in 2014, holding a similar trend until 2016 (see Figure 31).

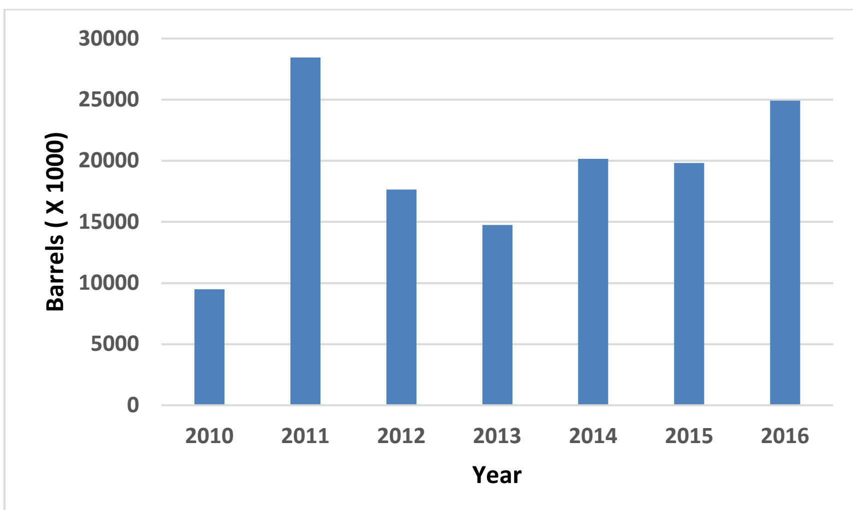

Figure 30 Trends in annual U.S. ethanol exports[101]

[100] Source USDA https://apps.fas.usda.gov/psdonline/app/index.html#/app/downloads?tabName=default 6/2/17

[101] Source EIA http://tonto.eia.gov/dnav/pet/hist/LeafHandler.ashx?n=PET&s=M_EPOOXE_EEX_NUS-Z00_MBBL&f=A 6/2/17

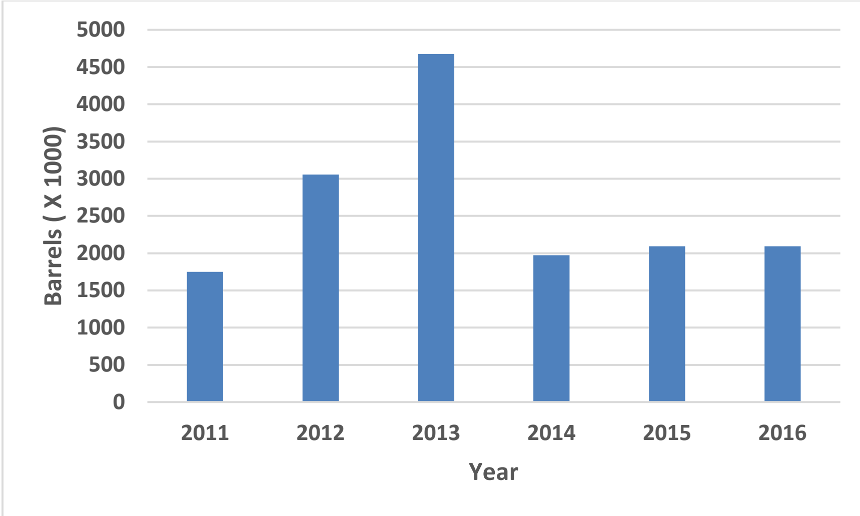

Figure 31 Trends in annual U.S. biodiesel exports[102]

Broadly, corn, soybean, ethanol, and biodiesel exports were the same or higher compared to 2007 or 2011, correlating with the increase in US ethanol and biodiesel production. A probable exception was the year 2012, which could be attributed to the drought conditions and yield decreases (see Chapter 2). These data differ from the projections in the 2011 Report, which expected declines in ethanol and biodiesel exports.

3.7.3 Changes in Drivers of International Impacts

Reports indicate that in the past decade market-mediated land use impacts (both direct and indirect land use changes) occurred due to demands for biofuel stocks. These changes probably resulted in decreased forest and pasture lands, crop intensification and multiple cropping, and depletion of global phosphorous reserves (Hein et al. 2012; Timilsina et al. 2012; Hertel et al. 2013; Langeveld et al. 2014; Tokgoz et al. 2014; Babcock 2015). As reported earlier in Section 2.5, forest loss is reported in countries exporting biofuels to the United States. Expansion and intensification of soybeans are observed in Argentina in relation to increased biofuel production, coinciding with loss of native grasslands (Solomon et al. 2015). The use of soybean as livestock feed is also reported as driving the expansion of soybeans into native grasslands (Modernel et al. 2016). In Indonesia, forest loss (driven in part by demand for biodiesel) and increased multiple cropping for palm oil are reported (Wicke et al. 2011; Langeveld et al. 2014). Market-mediated land use changes are reported, and it is likely that increased biofuel production

[102] Source http://www.eia.gov/dnav/pet/pet_move_expc_a_epoordb_eex_mbbl_a.htm 6/2/17

has contributed to these land use changes (as noted in Section 2.5). Quantification and causal attribution of these forest losses and other land use changes and environmental impacts due to biofuel production and renewable fuel standards remain uncertain and are an area for further research (see Section 2.5.2).

3.7.4 Potential for Future Changes in International Impacts

Biofuels made from more sustainable grasses or woody crops using higher-yield cellulosic technologies, or from waste biomass or biomass grown on degraded and abandoned agricultural lands, have been promoted as causing less environmental damage and having less impact on agricultural lands. If fuels from these feedstocks reach production at scales large enough to meet a substantial fraction of global fuel demand, the international environmental and natural resource impacts would be considerably less than those from current technologies. Global supply of such feedstocks remains only a small fraction of first-generation biofuels.

Efroymson et al. (2016) challenge the current practice of economic simulation models that incorporate commodity trade and coarse land cover data as causal pathways for land use change due to biofuel production, and they propose a comprehensive causal analysis framework. Beginning with the definition of the change that occurred, the causal analysis framework put forth by Efroymson et al. (2016) utilizes a strength-of-evidence approach that incorporates mechanistic plausibility of relationship, completeness of causal pathway, spatial co-occurrence, time order, analogous agents, simulation model results, and quantitative agent–response relationships. Other reports also indicate that complex analyses combining economic simulation models, place-based empirical studies, value chain analyses, and biophysical accounting could help infer causal mechanisms and quantification of such land use impacts (Hertel et al. 2013; Meyfroidt et al. 2013). Beyond causal analysis, Sanchez et al. (2012) argue that establishing a comparable and coordinated framework for estimating land use changes across the major biofuel trading nations could better inform policy outcomes.

3.7.5 Conclusions: International Impacts

- Since the 2011 Report, U.S. ethanol imports decreased, while biodiesel and renewable diesel imports increased, leading to potential land use change impacts in countries of origin. Exports of corn, DDGS, soybeans, and ethanol primarily increased or are similar in comparison with 2007 levels.
- Reports suggest that demands for biofuel feedstocks have led to market-mediated land use impacts (both direct and indirect land use changes) in the past decade.

- Cropland expansion and natural habitat loss (including forests) have been observed internationally, and it is likely that increased biofuel production has contributed to these land use changes.
- Quantification and causal attribution of land use change and international environmental impacts due to biofuel production remain uncertain and undetermined.

3.7.6 Research Needs: International Impacts

- Comprehensive causal analysis frameworks and coordinated frameworks for evaluating land use changes across biofuel trading nations may help our understanding of international land use change and environmental impacts.

4 Conclusions and Recommendations

4.1 Overarching Conclusions

The 2011 Report presented three overarching conclusions:

- Evidence to date from the scientific literature suggests that *current* environmental impacts from increased biofuels production and use associated with EISA 2007 are negative but limited in magnitude.
- Published scientific literature suggests a potential for both positive and negative environmental effects in the future.
- EISA goals for biofuels production can be achieved with minimal environmental impacts if existing conservation and best management practices are widely employed, concurrent with advances in technologies that facilitate the use of second-generation feedstocks.

Reports and data published since the 2011 Report have increased the confidence in the conclusions of that report. Research also generally confirms the expected environmental and resource conservation impacts of increased biofuel production and use, given the increased production of biofuels from corn grain and soybeans observed since the 2011 Report was published. There has been an increase in U.S. acreage planted with soybeans and a modest increase in U.S. acreage planted with corn since enactment of the EISA (see Figure 4), with strong indications that some of this increase is a consequence of increased biofuel production. There has not been a significant increase in cellulosic feedstocks (e.g., corn stover, perennial grasses, and woody biomass) since the 2011 Report was published. As a result, the environmental impacts continue to be primarily those associated with increased production of corn and soybeans, the associated conversion to fuels, and end use.

Since the 2011 Report, findings from the scientific literature and data from observations allows the conclusions of the 2011 Report to be reaffirmed, with qualification:

- Disregarding any effects that biofuels have on displacing other sources of transportation energy, evidence since 2011 indicates the specific environmental impacts listed in EISA Section 204 are negative. However, without assessing biofuels' displacement of other sources of transportation energy, there is insufficient evidence to support a conclusion on the overall direction or magnitude of effect.

- Literature published since 2011 supports the conclusion of the potential for positive and negative effects. Available information suggests, without accounting for the environmental effects of displacing other sources of transportation energy, the specific environmental impacts listed in EISA Section 204 are negative in comparison to the period prior to enactment of EISA.
- Evidence continues to support the conclusion that biofuel production and use could be achieved with reduced environmental impacts. The majority of biofuels continue to be produced from corn grain and soybeans, with associated impacts that are well understood. Cellulosic and other feedstocks remain a minimal contributor to total biofuel production.

4.2 Specific Conclusions

Conclusions regarding the environmental and resource conservation impacts for each of the sections are summarized below.

4.2.1 Land Use Change

- Biofuel feedstock production is responsible for some of the observed changes in land used for agriculture, but we cannot quantify with precision the amount of land with increased intensity of cultivation nor confidently estimate the portion of crop land expansion that is due to the market for biofuels.
- Recent research and anticipated updates to data are expected to improve our ability over the next three years to quantify the fraction of land use change attributed to biofuel feedstock production in the U.S.
- Evidence from multiple sources demonstrates an increase in actively managed cropland in the U.S. since the passage of EISA by roughly 4-7.8 million acres, depending upon the source.
- Much of this increase is likely occurring in the western and northern edges of the corn belt with reductions of pasture and grassland, but also through infilling of already agricultural areas.
- Thus, intensification likely dominates in already agricultural areas and extensification dominates in less agricultural areas.
  - Global cropland area has expanded since the year 2000, coinciding with the increase in U.S. biofuel production. During this period, the ratio of area harvested to arable land increased and crop yields increased significantly, due in large part to gains in total factor productivity.
  - Agricultural extensification and deforestation have been documented in countries that are major exporters of biofuels to the U.S., including Brazil, Argentina, and Indonesia.

- Cropland expansion and natural habitat loss (including forests) have been observed internationally during the implementation of the RFS program. It is likely that increased biofuel production has contributed to these land use changes, but significant uncertainty remains about the amount and type of land use changes that can be quantitatively attributed to U.S. biofuel consumption (see Box 3 on Attribution).
- Researchers have continued to update and refine economic models to estimate biofuel-induced land use changes.
- Due to inherent challenges, uncertainties are large, and progress reducing the sources of uncertainty has been limited.

4.2.2 Air Quality

- There is no new evidence that contradicts the conclusions of the 2011 Report concerning air quality. Those conclusions emphasized that life cycle emissions of NO_x, SO_x, CO, VOCs, NH_3, and particulate matter can be impacted at each stage of biofuel production, distribution, and usage. These impacts depend on feedstock type, land use change, and land management/cultivation practices and are therefore highly localized. The impacts associated with feedstock and fuel production and distribution are important to consider when evaluating the air quality impacts of biofuel production and use, along with those associated with fuel usage.
- Ethanol from corn grain has higher emissions across the life-cycle than ethanol from other feedstocks.
- Ethanol plants relying on coal have higher air pollutant emissions than plants relying on natural gas and other energy sources.
- The magnitude, timing, and location of all these emissions changes can have complex effects on the atmospheric concentrations of criteria pollutants (e.g., O_3 and $PM_{2.5}$) and air toxics, the deposition of these compounds, and subsequent impacts on human and ecosystem health.
- Ethanol increased NO_x emissions from light-duty vehicles certified to Federal Tier 2 Standards, likely occurring during times when the vehicle catalyst is not yet warmed up or air/fuel ratio is not perfectly controlled. However, only limited data exist on the impacts of biofuels on the tailpipe and evaporative emissions of light-duty Tier 3 vehicles and light-duty vehicles using advanced gasoline engine technologies to meet GHG emissions standards.

4.2.3 Water Quality

- The 2011 Report found that corn production intensification was associated with higher levels of erosion, chemical loadings to surface waters, and eutrophication.
- Modeling studies since the 2011 Report suggest that demand for biofuel feedstocks may contribute to harmful algal blooms, as recently observed in western Lake Erie, and to hypoxia, as observed in the northern Gulf of Mexico.
- Empirical studies documenting cropland extensification and crop switching to more corn suggest water quality impacts, but the magnitude of these changes is variable across the landscape and so may be detectable only in some regions.
- Implementation of conservation practices has been observed to result in a decrease of nitrogen, phosphorus, and soil erosion.
- Changes to future nitrogen and phosphorus loadings will depend on feedstock mix and crop management practices. Decreases in nitrogen and phosphorus loadings are possible should perennial feedstocks become dominant.
- Specific biofuel production scenarios expected to improve water quality may help decrease the water quality impact of predicted future extreme weather events.

4.2.4 Water Quantity

- As discussed in the 2011 Report, the irrigation of corn and soybeans grown for biofuels is the predominant water quantity impact. Water use for feedstock production is significantly greater than water use in the biofuel conversion process.
- There are some indications of increased water use in irrigated areas for corn between 2007 and 2012 and elevated rates of land use change to corn production in more arid Western states, including the Ogallala region. Adverse water availability impacts will most likely arise in already stressed aquifers and surface watersheds.
- Irrigation practices are dependent on a number of economic and agronomic factors that drive land management practices, making attribution of increased irrigation and water quantity to biofuels difficult.

4.2.5 Ecosystem Health and Biodiversity

- Loss of grasslands and wetlands is occurring in ecologically sensitive areas, including the Prairie Pothole Region.

- Loss of habitat and landscape simplification are associated with negative impacts to pollinators, birds, soil-dwelling organisms, and other ecosystem services, both in terrestrial and aquatic habitats.
- Increased fertilizer applications of nitrogen and phosphorus have known negative effects on aquatic biodiversity.

4.2.6 Soil Quality

- Corn-grain ethanol and soy biodiesel account for most of the biofuel volumes produced to date. As a result, almost all the soil quality impacts from biofuels, thus far, are from the production of the dominant conventional feedstocks.
- Conversion of grasslands to annual cropland typically negatively affects soil quality, with increases in erosion and the loss of soil nutrients and soil organic matter, including soil carbon. Impacts of this conversion can be partially mitigated – though not entirely – through the adoption of management practices such as conservation tillage.
- The soil quality impacts of converting from other crops to corn or soybeans are generally less than those of the conversion of grasslands. The production of corn on existing cropland can provide soil carbon benefits, although these benefits are outweighed on a per area basis by the negative effects of grassland conversion.
- Overall, these land use trends suggest that negative impacts to soil quality from biofuel feedstocks have increased since 2011, but this has not been quantified and the magnitude of effects depends predominantly on the relative areas of grasslands converted versus existing croplands attributable to biofuels.
- Corn stover is now being harvested at the commercial-scale in Iowa, and the scientific literature indicates this must be done carefully to avoid negatively affecting soil quality and crop yields.

4.2.7 Invasive Species

- Biofuels are primarily produced in the forms of bioethanol and biodiesel derived from food crops (i.e., non-invasive first generation biofuels – corn and soybeans). Hence current production of biofuel feedstocks poses little risk of invasion, consistent with findings in the 2011 Report.
- Weed risk assessments, part of the formal biofuel regulatory process, provide information on invasion risk and are designed to inform protective management of species and varieties that are predicted to be invasive.

- Increased cultivation of crops engineered for herbicide tolerance (e.g., glyphosate) and concomitant application of the herbicide has led to a widespread increase in the number of glyphosate-resistant weed species.
- Potentially invasive species approved as feedstocks require risk management actions under current RFS requirements. However, invasive species are not presently being used for commercial scale production of biofuels.

4.2.8 International Impacts

- Since the 2011 Report, U.S. ethanol imports decreased, while biodiesel and renewable diesel imports increased, leading to potential land use change impacts in countries of origin. Exports of corn, DDGS, soybeans, and ethanol primarily increased or are similar in comparison with 2007 levels.
- Reports suggest that demands for biofuel feedstocks have led to market-mediated land use impacts (both direct and indirect land use changes) in the past decade.
- Cropland expansion and natural habitat loss (including forests) have been observed internationally, and it is likely that increased biofuel production has contributed to these land use changes.
- Quantification and causal attribution of land use change and international environmental impacts due to biofuel production remain uncertain and undetermined.

4.3 Opportunities for Future Environmental Improvements

- Some cellulosic feedstock production scenarios are expected to reduce surface water nitrogen loadings, particularly following extreme weather events. (Water Quality)
- Priority should be placed on identifying effective strategies to manage withdrawals in “hot spots” (e.g., Ogallala aquifer) where high water demands and water stress are coinciding. (Water Quantity)
- While cellulosic feedstocks, such as perennial grasses, can provide environmental benefits for biodiversity and ecosystem services, their potential impact on streamflow within a watershed should be carefully considered. (Water Quantity)
- Planting perennial grasslands and replacing marginal croplands with perennial grasslands can enhance ecosystem services. (Ecosystems and Biodiversity)

- Increased use of effective conservation practices can have multiple benefits, from reduced stream sedimentation and nutrient runoff to protection of pollinator habitat. (Ecosystems and Biodiversity)
- Increased adoption of other technologies, such as time-release and other enhanced-efficiency fertilizers, alternative fertilizer placement methods, and precision agriculture, could further improve conservation practices in biofuel feedstock production. (Ecosystems and Biodiversity)
- Alternative biofuel feedstocks, such as perennial grasses and woody biomass, are not yet used at commercial scales. Studies have shown that these feedstocks can improve soil quality relative to current conditions, contingent on species grown and type of land converted (e.g., marginal, abandoned, or degraded lands). (Soil Quality)

4.4 Limitations

- This report does not include a comparative assessment of the impact of biofuels on the environment relative to the impacts of other transportation fuels or energy sources, including fossil fuels, for every environmental endpoint, limiting the ability of this report to draw conclusions regarding the comprehensive environmental impacts of biofuels.
- The environmental impacts discussed in this report are not constant across all locations due to local factors, such as the extent of land use change and local relationships between land use change and their associated direct and indirect impacts.
- We cannot now confidently quantify the fraction of increased land use change and associated environmental impacts due to changes in biofuel production.
- Numerous factors influence the markets for biofuels and thus the associated environmental impacts, including: regional considerations; scale and volume of future commercial biofuel operations; development of hybrid biofuel conversion processes; changes in vehicle technologies; and changes in agricultural practices due to biofuel production and implications for environmental impacts. Each of these, whether individually or in combination, will affect the ultimate environmental impacts associated with biofuel production and use.

4.5 Research Needs

- Research is needed to quantify changes in the intensity of cultivation on existing agricultural land. (Land Use Change)

- Research is also needed to more effectively connect changes in land use to the environmental impacts of concern. (Land Use Change)
- Comprehensive studies of the impacts of biofuels on the emissions from advanced light-duty vehicle technologies (Tier 3), similar in scope to studies cited in this report for light-duty Tier 2 vehicles, would improve the understanding of the potential for biofuel-specific pollutants and associated health impacts as new technologies enter the vehicle fleet. These studies should consider engine technologies phasing into use for compliance with current and future light-duty GHG standards, with a focus on vehicles compliant with the Federal Tier 3 or California LEV III criteria pollutant emissions standards currently under implementation. Such technologies would include engine downsizing with addition of turbocharging, gasoline direct injection, and non-traditional thermodynamic cycles such as Miller or Atkinson. (Air Quality)
- Additional research and analyses are needed to adequately understand the potential health effects of exposure to biofuels and emissions from vehicles using biofuels under real-world conditions, concentrations, and exposures including to susceptible human populations. It would be appropriate to study health effects in populations exposed to biodiesel and ethanol blends in "hotspots," such as fuel production sites, and those exposed to combustion products of biodiesel and ethanol blends, especially at high blend levels. Such studies could include drivers of vehicles utilizing those fuels.
- Updated modeling is needed to incorporate improved emissions estimates as laboratory, field, and other studies lead to a better understanding of biofuel-related emissions changes and associated changes in the magnitude and composition of pollutants on air quality, health, and attainment with ambient air quality standards. (Air Quality)
- Studies are needed of water quality impacts associated with leaks and/or spills from biofuel production facilities and storage tanks. Such work would address the effectiveness of existing leak detection and cleanup approaches to address releases to the environment and resulting contamination plumes. (Water Quality)
- Studies are needed to determine the extent to which increases in water consumption and withdrawals – due to changes in land use/management change – can be attributed to feedstock production for biofuels. In particular, studies should continue to explore increases in water demands that have occurred or are occurring along water-stressed areas, both for surface and groundwater. (Water Quantity)

- Research, both modeling and field work to verify modeling parameters, is needed to better understand future cellulosic feedstock water demands while assessing water quantity, water quality, and soil quality in an integrated manner. (Water Quantity)
- Studies that target the interactive effects of land use change and feedstock production could help identify impacts to specific organisms. (Ecosystem Health and Biodiversity)
- Research on the efficacy of methods to expand pollinator habitat in agricultural systems can improve understanding of appropriate methods and their potential tradeoffs for different agricultural areas. (Ecosystem Health and Biodiversity)
- Quantitative estimates of the cumulative soil quality effects are needed for the land use changes described in Section 2.4 and for the proportion attributable to biofuel feedstock production (this includes both the conversion of land to corn and soybeans and the management practices implemented). (Soil Quality)
- Methodological advancements for weed risk assessments and lessons from other industries (e.g., horticulture) should be incorporated to inform on potential invasiveness of biofuel feedstocks. (Invasive Species)
- Modeling and field work are needed to investigate the impacts of gene flow between novel feedstock varieties (genetically engineered, selectively bred, or a combination) and local natives. (Invasive Species)
- Comprehensive causal analysis frameworks and coordinated frameworks for evaluating land use changes across biofuel trading nations may help our understanding of international land use change and environmental impacts. (International Impacts)

4.6 Recommendations

- Additional research in coordination with other organizations (e.g., federal agencies, states, trade organizations) is recommended to better characterize land use change due to changes in biofuel feedstock production.
- Efforts at the federal level, as described by the Biomass Research and Development Board, to improve efficiencies and sustainability of processes across the biofuel supply chain should be continued and strengthened where possible.
- An ecosystem approach is recommended to evaluate environmental and natural resource impacts of biofuel production. Such an approach provides an integrative perspective that accounts for complex interactions of multiple stressors across different locations.

- Incorporating local information and perspectives will improve understanding of changes at local scales, which will enhance opportunities for improved information and will enable targeted responses to prevent and mitigate adverse impacts of biofuel production and use.
- Best management practices should be encouraged, incentivized, and otherwise expanded to promote conservation and sustainability in agricultural systems.

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Appendix A: Abbreviations and Glossary

Abbreviations

AMOX .... ammonia oxidation catalyst
bbl .... barrel
BOD .... biological oxygen demand
Bt .... *Bacillus thuringiensis*
CAFE .... corporate average fuel economy
CDL .... crop data layer
CDPF .... catalyzed diesel particulate filter
CH_4 .... methane
CNG .... compressed natural gas
CO .... carbon monoxide
CO2 .... carbon dioxide
CRP .... Conservation Reserve Program
DDGS .... distillers dregs and waste, or distiller's dried grain and solubles
DOC .... diesel oxidation catalyst
DOT .... U.S. Department of Transportation
DRP .... dissolved reactive phosphorus
E*XX* .... fuel blend of *XX* volume % ethanol and remainder gasoline
EISA .... Energy Independence and Security Act of 2007
EPIC .... Environmental Policy Integrated Climate (model)
FAO .... Food and Agriculture Organization (United Nations)
FAPRI .... Food and Agricultural Policy Research Institute (model)
FASOM .... Forest and Agricultural Sector Optimization Model
FFV .... Flexible-fuel vehicle
gal .... U.S. gallon
GHG(s) .... greenhouse gas(es)
HAB(s) .... harmful algal bloom(s)
HC .... hydrocarbon
HHDDE .... heavy-heavy-duty diesel engine
HT .... herbicide tolerant
LCA .... life cycle assessment
LCI .... life cycle inventory
LHDDE .... light-heavy-duty diesel engine
LNG .... liquefied natural gas
LUC .... land use change
MHDDE .... medium-heavy duty diesel engine
MOVES .... MOtor Vehicle Emissions Simulator (model)
MRB .... Mississippi River Basin
N .... nitrogen
NAIP .... National Agriculture Imagery Program (USDA)
NASS .... National Agricultural Statistics Service (USDA)

NEI..National Emission Inventory
NEWS...Nutrient Export from WaterSheds (model)
NH_3 ....ammonia
NMHC ....non-methane hydrocarbons
NMOG ....non-methane organic gas
NO_x ....oxides of nitrogen
NREL...National Renewable Energy Research Laboratory (DOE)
NRI..National Resources Inventory (USDA)
NTE...not to exceed (emissions)
P ....phosphorus
POLYSYS..Policy Analysis System (model)
PM...particulate matter
PM2.5..particulate matter with aerodynamic diameter less than 2.5 μm
RFS ....Renewable Fuel Standard
RFS2 ....revised Renewable Fuel Standard
RPA...Resource Planning Act
RIA..regulatory impact analysis
RIN..Renewable Identification Number
RON ....research octane number
RQ...risk quotient
SCR...selective catalytic reduction
SOM..soil organic matter
SOx ....sulfur oxides
SPARROW ....SPAtially Referenced Regressions On Watershed model
SWAT ....Soil and Water Assessment Tool
TP..total phosphorus
USDA...U.S. Department of Agriculture
USDA ERS ....U.S. Department of Agriculture Economic Research Service
US DOE (or DOE).............................U.S. Department of Energy
US EPA (or EPA) ..............................U.S. Environmental Protection Agency
USGS ....U.S. Geological Survey
VMT..vehicle-miles traveled
VOC ....volatile organic compounds
WRA ....weed risk assessment

Glossary

advanced biofuel: A renewable fuel, other than ethanol derived from corn starch, that has life cycle greenhouse gas emissions that are at least 50 percent less than life cycle GHG emissions from petroleum fuel. Cellulosic biofuels must achieve a 60 percent reduction in GHG to get credit for being "advanced."

biochar: the product of heating biomass in the absence of- or with limited air, with the resulting material rich in organic carbon (Lehmann et al. 2015). This material can be used as a soil amendment.

biodiesel (also known as "biomass-based diesel"): A renewable fuel produced through transesterification of organically derived oils and fats. May be used as a replacement for, or component of, diesel fuel. According to 40 CFR 80.1401, biodiesel means "a mono-alkyl ester that meets ASTM D6751 ('Standard Specification for Biodiesel Fuel Blend Stock (B100) for Middle Distillate Fuels')."

biodiversity: The variety and variability among living organisms and the ecological complexes in which they occur. Biodiversity can be defined as the number and relative frequency of different items, from complete ecosystems to the biochemical structures that are the molecular basis of heredity. Thus, the term encompasses ecosystems, species, and genes.

biofuel: Any fuel made from organic materials or their processing and conversion derivatives.

biofuel production: The process or processes involved in converting a feedstock into a consumer-ready biofuel.

biorefinery: A facility that converts biomass into fuels, heat, chemicals and other products using a variety of processes and equipment.

blendwall: The amount of ethanol that gasoline companies are permitted to blend into petroleum based fuel, current 10 percent (E10).

conservation tillage: Any tillage practice leaving at least 30% of the soil surface covered by crop residues; whereas conventional tillage leaves less than 15% of the ground covered by crop residues (Lal 1997). No-till management, a subset of conservation tillage, disturbs the soil only marginally by cutting a narrow planting strip.

corn stover: The stalks, leaves, husks, and cobs that are not removed from the fields when corn is harvested.

direct land use change: Land conversion that is directly related to the biofuel supply chain. An example of direct land use change would be the planting of biofuel feedstock on land that was previously native forest, to increase the supply of ethanol to export to the U.S.

double cropping: A form of agricultural intensification. Practice of growing two crops on the same piece of land during a single growing season.

ecosystem health: The ability of an ecosystem to maintain its metabolic activity level and internal structure and organization, and to resist external stress over time and space scales relevant to the ecosystem.

ethanol (also known as "bioethanol"): A colorless, flammable liquid produced by fermentation of sugars. Ethanol is generally blended with gasoline and used as a fuel oxygenate.

extensification: The expansion of agricultural land, like row crops, onto previously uncultivated land.

feedstock: In the context of biofuel, "feedstock" refers to a biomass-based material that is converted for use as a fuel or energy product.

hypoxia: The state of an aquatic ecosystem characterized by low dissolved oxygen levels (less than 2 to 3 parts per million) due to accelerated algal growth, decay of excess vegetation and algae, and reduced light penetration because of excessive nutrient levels (eutrophication). Low dissolved oxygen can reduce fish populations and species diversity in the affected area.

indirect land use change: land conversion that is a market-oriented response to changes in the supply and demand of goods that arise from increased production of biofuel feedstocks. An example of indirect land use change would be the clearing of foreign land to plant corn in response to an increase in global commodity prices caused by a decrease in U.S. corn exports.

intensification: Increased intensity of cultivation with no change in total agricultural land acreage

land use change: Conversion of land from one use or cover-type to another. Often human induced.

life cycle assessment: A comprehensive systems approach for measuring the inputs, outputs, and potential environmental impacts of a product or service over its life cycle, including resource extraction/generation, manufacturing/production, use, and end-of-life management.

renewable biofuel: A fuel produced from renewable biomass that is used to replace or reduce the use of fossil fuel.

renewable diesel: Diesel fuel derived from biomass, generally using a thermal depolymerization process, which meets the requirements of the American Society of Testing and Materials D975 or D396 standards.

row crop: A crop planted in rows wide enough to allow cultivators between the rows. Examples include corn, soybeans, peanuts, potatoes, sorghum, sugar beets, sunflowers, tobacco, vegetables, and cotton.

soil organic matter (SOM): "The organic fraction of the soil that includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil population" (Brady et al. 2000).

soil quality: "The capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation. In short, the capacity of the soil to function" (USDA-NRCS 2017).

water availability: In the context of this report, water availability refers to the amount of water that can be appropriated from surface water sources (e.g., rivers, streams, lakes) or groundwater sources (e.g., aquifers) for consumptive uses.

water quality: Water quality is a measure of the suitability of water for a particular use based on selected physical, chemical, and biological characteristics. It is most frequently measured by characteristics of the water such as temperature, dissolved oxygen, and pollutant levels, which are compared to numeric standards and guidelines to determine if the water is suitable for a particular use.

weed risk assessments: Formalized procedures for determining invasion risk. They are designed to predict invasive and non-invasive species/varieties and distinguish between them based on a set of questions about their history of invasiveness in other places, biological traits, and suitability for the environment into which they will be introduced.

Appendix B: Key Terms from Major Land Use Change Studies

This Appendix provides the abbreviated definitions of key terms from the four major federal land use studies discussed in Section 2.4. Full definitions are available in the source literature. Definitions are indented if they are a subset of a larger category.

USDA Major Uses of Land in the US, 2012 (MLU)

Cropland. Total cropland includes five components: cropland harvested, crop failure, cultivated summer fallow, cropland used only for pasture, and idle cropland.

> **Cropland used for crops.** Three of the cropland acreage components—cropland harvested, crop failure, and cultivated summer fallow—are collectively termed cropland used for crops, or the land used as an input to crop production.
>
> > **Cropland harvested.** Includes row crops and closely sown crops; hay and silage crops; tree fruits, small fruits, berries, and tree nuts; vegetables and melons; and miscellaneous other minor crops.
> >
> > **Crop failure.** Consists mainly of the acreage on which crops failed because of weather, insects, and diseases but does include some land not harvested due to lack of labor, low market prices, or other factors.
> >
> > **Cultivated summer fallow.** Refers to cropland in subhumid regions of the West that are cultivated for one or more seasons to control weeds and accumulate moisture before small grains are planted.
>
> **Cropland pasture.** Generally is considered to be in long-term crop rotation. This category includes acres of crops hogged or grazed but not harvested and some land used for pasture that could have been cropped without additional improvement.
>
> **Idle cropland.** Includes land in cover and soil-improvement crops and cropland on which no crops were planted. Some cropland is idle each year for various physical and economic reasons.

Grassland pasture and range. Grassland pasture and range encompass all open land used primarily for pasture and grazing, including shrub and brush land types of pasture; grazing land with sagebrush and scattered mesquite; and all tame and native grasses, legumes, and other forage used for pasture or grazing—regardless of ownership.

Forested land. As defined by the Forest Service, the 766 million acres of forested land in 2012 consist of "land at least 120 feet (37 meters) wide and at least 1 acre (0.4 hectare) in size with at least 10 percent cover (or equivalent stocking) by live trees, including land that formerly had such tree cover and that will be naturally or artificially regenerated."

> **Timberland.** Forestland that produces or is capable of producing crops (in excess of 20 cubic feet per acre per year) of industrial wood and not withdrawn from timber use by statute or administrative regulation.
>
> **Reserved forestland.** Forestland withdrawn from timber use through statute, administrative regulation, or designation without regard to productive status. Forested wilderness areas and parks are included in this category.
>
> **Other forestland.** Forestland other than timberland and productive reserved forestland. It includes available forestland, which is incapable of annually producing 20 cubic feet (1.4 cubic meters) per acre (0.4 hectare) of industrial wood under natural conditions because of adverse site conditions, such as sterile soils, dry climate, poor drainage, high elevation, steepness, or rockiness.

USDA 2012 Census of Agriculture

Total cropland. This category includes cropland harvested, other pasture and grazing land that could have been used for crops without additional improvements, cropland on which all crops failed or were abandoned, cropland in cultivated summer fallow, and cropland idle or used for cover crops or soil improvement but not harvested and not pastured or grazed.

> **Harvested cropland.** This category includes land from which crops were harvested and hay was cut, land used to grow short-rotation woody crops, Christmas trees, and land in orchards, groves, vineyards, berries, nurseries, and greenhouses.
>
> **Other pasture and grazing land that could have been used for crops without additional improvements.** This category includes 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.
>
> **Other cropland.** This includes all cropland other than harvested cropland or other pasture and grazing land that could have been used for crops without additional improvements. It includes

cropland idle, used for cover crops or soil improvement, cropland which all crops failed or were abandoned, and cropland in cultivated summer fallow.

Cropland idle or used for cover crops or soil improvement, but not harvested and not pastured or grazed. Cropland idle includes any other acreage which could have been used for crops without any additional improvement and which was not reported as cropland harvested, cropland on which all crops failed, cropland in summer fallow, or other pasture or grazing land that could have been used for crops without additional improvements.

Cropland on which all crops failed or were abandoned. No separate definition.

Cropland in cultivated summer fallow. No separate definition.

USDA 2012 National Resources Inventory

Cropland. A land cover/use category that includes areas used for the production of adapted crops for harvest. Two subcategories of cropland are recognized: cultivated and noncultivated.

Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hayland or pastureland that is in a rotation with row or close-grown crops.

Noncultivated cropland includes permanent hayland and horticultural cropland.

Hayland. A subcategory of cropland managed for the production of forage crops that are machine harvested. The crop may be grasses, legumes, or a combination of both. Hayland also includes land in set-aside or other short-term agricultural programs.

Horticultural cropland. A subcategory of cropland used for growing fruit, nut, berry, vineyard, and other bush fruit and similar crops. Nurseries and other ornamental plantings are included.

Land cover/use. A term that includes categories of land cover and categories of land use. Land cover is the vegetation or other kind of material that covers the land surface. Land use is the purpose of human activity on the land; it is usually, but not always, related to land cover.

Pastureland. A land cover/use category of land managed primarily for the production of introduced forage plants for livestock grazing. Pastureland cover may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture. Management usually consists of cultural treatments:

fertilization, weed control, reseeding or renovation, and control of grazing. For the NRI, includes land that has a vegetative cover of grasses, legumes, and/or forbs, regardless of whether or not it is being grazed by livestock.

Rangeland. A land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grasslike plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass, are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied. Grasslands, savannas, many wetlands, some deserts, and tundra are considered to be rangeland. Certain communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also included as rangeland.

Row crops. A subset of the land cover/use category cropland (subcategory, cultivated) comprising land in row crops, such as corn, soybeans, peanuts, potatoes, sorghum, sugar beets, sunflowers, tobacco, vegetables, and cotton.

USGS U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends, 1974-2012 (NWALT)

Production, Crops. Areas used for the production of crops, such as corn, soybeans, wheat, vegetables, or cotton, as well as perennial woody crops such as orchards and vineyards. Includes cultivated crops, row crops, small grains, and fallow fields. Identical definition to NLCD 2011 class 82.

Production, Pasture/Hay. Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Identical definition to NLCD 2011 class 81.

Production, Grazing Potential. Areas of good grazing potential beyond what is indicated by the NLCD. Information suggests the land could and has been used at least on a seasonal or occasional basis for animal grazing, including woodland pasture.

FAOSTAT Land Use Data

Arable and Permanent Crops. Arable land is the land under temporary agricultural crops (multiple-cropped areas are counted only once), temporary meadows for mowing or pasture, land under market and kitchen gardens and land temporarily fallow (less than five years). The abandoned land resulting from shifting cultivation is not included in this category. Data for “Arable land” are not meant to indicate the amount of land that is potentially cultivable. Permanent crops is the land cultivated with long-term crops which do not have to be replanted for several years (such as cocoa and coffee); land under trees and shrubs producing flowers, such as roses and jasmine; and nurseries (except those for forest trees, which should be classified under "forest"). Permanent meadows and pastures are excluded from land under permanent crops.

Area harvested. Data refer to the area from which a crop is gathered. Area harvested, therefore, excludes the area from which, although sown or planted, there was no harvest due to damage, failure, etc. It is usually net for temporary crops and some times gross for permanent crops. Net area differs from gross area insofar as the latter includes uncultivated patches, footpaths, ditches, headlands, shoulders, shelterbelts, etc. If the crop under consideration is harvested more than once during the year as a consequence of successive cropping (i.e., the same crop is sown or planted more than once in the same field during the year), the area is counted as many times as harvested. On the contrary, area harvested will be recorded only once in the case of successive gathering of the crop during the year from the same standing crops. With regard to mixed and associated crops, the area sown relating to each crop should be reported separately. When the mixture refers to particular crops, generally grains, it is recommended to treat the mixture as if it were a single crop; therefore, area sown is recorded only for the crop reported. Source: FAO Statistics Division

Forest area is the land spanning more than 0.5 hectares with trees higher than 5 metres and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use. Forest is determined both by the presence of trees and the absence of other predominant land uses. The trees should be able to reach a minimum height of 5 metres (m) in situ. Areas under reforestation that have not yet reached but are expected to reach a canopy cover of 10 percent and a tree height of 5 m are included, as are temporarily unstocked areas, resulting from human intervention or natural causes, which are expected to regenerate. Includes: areas with bamboo and palms provided that height and canopy cover criteria are met; forest roads, firebreaks and other small open areas; forest in national parks, nature reserves and other protected areas such as those of specific scientific, historical, cultural or spiritual interest; windbreaks, shelterbelts and corridors of trees with an area of more than 0.5 ha and width of more than 20 m; plantations primarily used for forestry or protective purposes, such as: rubber-wood plantations and cork, oak stands. Excludes: tree stands in agricultural production systems, for example in fruit plantations and agroforestry systems. The term also excludes trees in urban parks and gardens."

Permanent meadows and pastures is the land used permanently (for a period of five years or more) for herbaceous forage crops, either cultivated or naturally growing. A period of five years or more is used to differentiate between permanent and temporary meadows.

Attachment 2

UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF COLUMBIA

SIERRA CLUB Plaintiff, v. ANDREW WHEELER, in his official capacity as Acting Administrator, U.S. Environmental Protection Agency,[1] Defendant.	Case No. 1:17-cv-02174-APM

Declaration of Christopher Grundler

I, Christopher Grundler, declare that the following statements are true and correct to the best of my knowledge and belief and that they are based on my personal knowledge, or on information contained in the records of the United States Environmental Protection Agency (EPA), or on information supplied to me by EPA employees.

A. General Background

1. I am the Director of the Office of Transportation and Air Quality (OTAQ), Office of Air and Radiation (OAR) at the EPA, a position I have held since 2012. Prior to my current position, I was the Deputy Director of OTAQ, a position I held since 1995.
2. OTAQ's mission is to protect human health and the environment by reducing air pollution and greenhouse gas emissions from mobile sources and the fuels that power them. OTAQ's programs address emissions from the range of mobile sources: cars and

[1] Andrew Wheeler has been substituted for Scott Pruitt under Fed. R. Civ. P. 25(d).

light trucks, large trucks and buses, farm and construction equipment, lawn and garden equipment, marine engines, aircraft, and locomotives.

3. OTAQ's primary activities include: (a) assessing mobile source-related air quality problems and developing sophisticated modeling tools to develop solutions, measure results, and support emission inventories; (b) establishing national standards to reduce emissions from on-road and nonroad mobile sources of pollution; (c) implementing national mobile source standards; (d) developing fuel efficiency programs and technologies to reduce the emission of greenhouse gases from the transportation sector; and (e) researching, evaluating, and developing advanced technologies for controlling emissions, as well as developing new strategies for improving fuel efficiency.

B. Background on Assessing Air Quality Impacts of Motor Vehicles and Engines and their Fuels

4. Motor vehicles and engines contribute to air pollution such as ozone, particulate matter, and hazardous air pollutants. Motor vehicles and engines emit compounds such as nitrogen oxides, volatile organic compounds, particulate matter, and air toxics. These emissions react in the atmosphere to form additional pollutants, such as ozone.

5. Thus, understanding the air quality impacts of motor vehicles and engines involves estimating their emissions and how those emissions behave in the atmosphere.

6. Estimating the emissions from motor vehicles and engines requires a model that accounts for many variables, such as engine type, its emissions controls and performance, and properties of the fuel it uses (for example, levels of ethanol, vapor pressure, and distillation properties). Estimating how those emissions behave in the atmosphere to produce air pollution requires air quality modeling that accounts for emissions from all sources, ambient concentrations of other pollutants, meteorology, chemistry, and so on.

7. EPA uses the MOtor Vehicle Emission Simulator (MOVES) to estimate emissions at a county level. The MOVES model is used to create emission factor tables, which, in turn, are used by the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system to generate air quality model-ready files for input into a photochemical air quality model called the Community Multiscale Air Quality (CMAQ) model.

8. EPA's experience in assessing air quality impacts of renewable fuels[2] indicates that the magnitude, timing, and location of emissions changes can have complex effects on the atmospheric concentrations of air pollutants, and that these concentrations can be spatially variable. Detailed emissions and air quality modeling are needed to address this complexity. Without adequate time to perform this quantitative modeling, summarized below, the scope of an air quality assessment (and, by extension, what conclusions can be drawn from it) would be significantly different.

C. Analysis and Steps for the Anti-backsliding Study

9. As described in more detail below, EPA's current plan to conduct the anti-backsliding study required by 42 U.S.C. § 7545(v)(1)(A) requires significant work to (i) estimate county-level emissions for a scenario with required renewable fuel volumes and a scenario without required renewable fuel volumes; (ii) model ambient concentrations for each of these scenarios; and (iii) assess and interpret the results of that modeling in a study. I estimate that the most expeditious schedule under the circumstances for completing the anti-backsliding study is approximately 14 months from the date of the

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[2] See U.S. EPA, Renewable fuel standard program (RFS2) regulatory impact analysis (2010). https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P1006DXP.txt

Court's ruling on summary judgment. Paragraphs 12 to 21 describe this schedule. Because this is the most expeditious schedule for the steps described below, any shorter schedule ordered by the court would require EPA to reevaluate plans for the study in order meet the Court's deadline. If the Court orders EPA to complete the study in less than 14 months, we would need to determine how to meet the deadline and reevaluate the steps described below, which could affect what we do at each step. For that reason, EPA intends to initiate the steps in the schedule below once the Court issues its order setting forth a deadline for the completion of the anti-backsliding study.

10. The schedule considers to the extent possible the resources (both of EPA staff and contractors) that can be devoted to completing the study, and assumes that the resources (both personnel and computing resources) are available as soon as necessary for this project.

11. Devoting additional EPA staff to the study would not shorten the schedule described below, because the work involved requires very specialized skills and this schedule already assumes full involvement by EPA's subject matter experts. Furthermore, because quality assurance is necessary at each step of the analysis and before subsequent steps can begin, almost all the steps must occur sequentially. The time allotted for quality assurance is a critical aspect of ensuring an expeditious schedule for completion of the study; if errors are found late in the process, previous steps will need to be re-done.

Step one: estimate county-level emissions (5 months)

12. To give some context of the emission information we use, EPA develops estimates of county-level emissions as a normal part of our rulemaking and National Emission Inventory development. We do this work frequently and we have optimized our process

to produce accurate results under tight time constraints. The schedule provided here is based on this experience.

13. The analysis and steps required to develop emissions estimates are largely sequential. As described below, developing model inputs and doing model runs are performed sequentially in order to include time for quality assurance of modeling results and possible corrections to inputs (which requires re-running a simulation before proceeding to the next step). The steps that could be performed concurrently, such as documentation and parallel processing, have already been accounted for in the schedule.

14. *Develop County-Specific Information (10 weeks)*: To estimate county-level emissions, the EPA uses county-specific inputs. Those inputs come from detailed databases developed by EPA that contain information about the population of vehicles and equipment, and their activity, specific to each county. We group together counties with similar properties (such as fleet age distribution, inspection program, and fuel properties), and focus on a "representative county" in each group. We run the MOVES model on only the representative counties, which significantly reduces computation time. Two aspects of defining representative counties take significant amounts of time:

    a. Because fuel properties are an important factor in determining county groups, profiling fuel properties for every county is the first step in identifying the representative counties. This effort includes the analysis and creation of overall market shares for different blends of gasoline and ethanol, and for different blends of petroleum diesel and biodiesel. These market shares then need to be allocated geographically and fuel profiles must be specified in great detail by parameters that may vary with time, location and biofuel content. And because EPA has to compare emissions for scenarios with and without required renewable fuel volumes, it would need to profile fuel properties for both scenarios. This step is expected to take 6 weeks.

b. Next, EPA has to create county databases and identify representative counties. Many state and local agencies provide county-level MOVES inputs in the form of county databases. EPA screens all submitted data using several quality assurance (QA) scripts that analyze the individual tables in each county database and flag missing or unrealistic data values. EPA then reviews all potential errors, identifies errors that need to be addressed, and coordinates with the responsible state/local agency to clarify whether the data needs revision. After collecting all the county-specific model inputs and creating county databases, the representative counties are selected based on various factors. This step is expected to take 4 weeks.

15. *Generate MOVES Emission Factors for SMOKE for Both Scenarios (With and Without Required Renewable Fuel Volumes) (8 weeks)*. EPA runs MOVES for each representative county using two different fuels -- January fuels and July fuels. The composition of, and emissions attributable to, fuel sold in the summer can be substantially different from fuel sold in the winter due to additional regulatory requirements intended to reduce ozone pollution in warmer months. In addition, EPA runs MOVES for each fuel for a range of temperatures spanned by the represented county group. Typically, tens of thousands of these runs are needed to produce a nationwide inventory of emissions for one calendar year. The MOVES runs result in four emission factor tables for each representative county and fuel month: rate per distance, rate per vehicle, rate per hour, and rate per profile. Each table provides estimates of emission rates from different vehicle processes

at different temperatures and speeds. These emissions factors allow EPA to fully account for emissions from all emission processes for all temperatures and speeds for vehicles in that county. After the initial MOVES runs for all representative counties are completed, a series of QA steps are taken to identify which runs failed and need to be re-run.

16. Having additional contractor support for the county-level emission estimates (step one) is not likely to shorten the time needed to complete the study because (a) the work performed by EPA staff cannot be easily transferred to contractors without devoting significant time to train them; (b) even when new contract staff are properly trained, because they are inexperienced, EPA staff would need to thoroughly review the contractors' work, which might take longer than doing the work in-house; and (c) we are not aware of additional contractors who already possess the specialized skills needed to perform the work.

Step two: model ambient concentrations (5 months)

17. Once EPA has emissions-factor data, it prepares that data for the air-quality models, a process that is expected to take 6 weeks:
    a. First, EPA uses the SMOKE model to apply emission-factor data (generated from representative counties) to county-specific vehicle data, a method that produces vehicle emission rates for all counties. These calculations are done for every county, grid cell, and hour in the continental U.S. This step is expected to take 3 weeks.
    b. EPA then examines the results from all counties to look for anomalies, an important step to assure that relevant data is not missing or incorrect and that errors in post-processing have not occurred. To do this, we prepare tables, graphs,

and maps that enable us to see if some results stand out as unusually large or small. If we find odd results, we investigate to determine the cause. If corrections are needed, MOVES runs and/or inventory processing may have to be repeated. This step is expected to take 2 weeks.

c. EPA takes the county-specific vehicle emission rates and merges them with the inventories for the non-vehicle emission sources. This merging happens for every 12 km by 12 km grid cell and hour and there is a QA step to ensure that the merge is completed correctly. This step is expected to take 1 week.

18. *Model two scenarios (6 weeks).* EPA will utilize the CMAQ model to run the two emissions scenarios (with and without required renewable fuel volumes). CMAQ brings together three kinds of models: meteorological models to represent atmospheric and weather activities; emission models to represent man-made and naturally-occurring contributions to the atmosphere; and an air chemistry-transport model to predict the atmospheric fate of air pollutants under varying conditions. CMAQ will simulate the formation and fate of pollutants over the continental United States, using 12 km grid cells, for an entire year. Because of the complexity of these calculations, even assuming computing resources are available, each CMAQ run will take about 4 weeks. There can be some overlap so that once the first scenario is running, the second scenario can be run concurrently, so we expect this step to take 6 weeks.

19. *Process deliverables and check results (8 weeks).* The gridded hourly outputs of pollutant concentrations from each model run undergo extensive processing, including (1) extracting hourly grid-cell ozone, $PM_{2.5}$, and air toxic concentrations from model output files, (2) time-shifting model predictions from Greenwich Mean Time, which is

the native time zone used in air quality model runs, to local standard time, and (3) calculating appropriate pollutant concentrations (e.g., 8-hour ozone concentrations and 24-hour $PM_{2.5}$ concentrations). Once appropriate modeled concentrations of pollutants are available, then we compare those outputs (e.g., create difference values for each grid cell) and translate the information into visuals such as maps and tables, which can take significant time. QA happens at multiple places: on the initial raw outputs from each run, and on the processed concentration outputs from each run, and finally on the air quality concentration differences. Once modeling outputs are reviewed for accuracy, we may need to revise the visuals, for instance, adjusting concentration ranges presented in air quality difference maps.

20. Modeling ambient concentrations (step two) is the part of the anti-backsliding study that benefits most from contractor help. In fact, we expect that this step will be done mostly by contractors with EPA direction and oversight. In addition, we note that the modeling and post-processing that is performed requires use of specialized computing power resources that are shared across EPA.

Step three: assess and interpret results (4 months)

21. Finally, EPA would need to draft the anti-backsliding study, meaning we would need to document analysis methodologies, present and describe the results, and develop conclusions. The draft report will then need to be internally reviewed, including through internal peer consultation. Once comments are considered and modifications to the draft report are complete, the report would need internal agency review and clearance at the management level. We expect these steps to take approximately 4 months.

D. Analysis and Steps for Action under Section 7545(v)(2)

22. What the anti-backsliding study concludes about adverse air quality impacts will have a profound impact on what mitigation measures EPA considers, the types and complexity of supporting analyses that must be conducted, and the length of time needed to complete the regulatory process.

23. For example, if adverse air quality impacts are found, and if there are fuel controls that might mitigate those impacts, a regulatory response can vary a great deal based on what kind of air-quality impacts we are dealing with. A regulatory strategy to reduce emissions of volatile organic compounds (such as reducing fuel volatility) may look very different from a strategy to reduce emissions of particulate matter and air toxics (such as controlling the aromatic content of gasoline). The regulatory strategies we consider will determine the complexity of feasibility, cost, lead time, emissions, and other analyses, and these in turn will determine the time needed for rulemaking.

24. As a result, it is not currently possible to specify the appropriate deadline for completing such a regulation.

25. We have, however, tried to forecast a timeline to take the action required by 42 U.S.C. § 7545(v)(2). Based on our best estimate at this time, it could take a total of 23 to 44 months for EPA to propose a rule to implement mitigation measures and 13 to 26 months to finalize the rule. Alternatively, if we determine that no mitigation measures are necessary, we estimate that it could take 3 months if the anti-backsliding study finds no adverse air-quality impacts, and anywhere from 23 to 44 months if the study finds some air quality impacts but EPA concludes that no measures are necessary. Below we describe the many variables that contribute to the uncertainties in our estimate. Our estimate assumes that where analyses can be conducted in parallel with one another, we

will do so. To illustrate which steps can be conducted in parallel and which are sequential, we attach Exhibit A, which shows the time it takes to complete different steps of a proposed rulemaking based on the top-end estimates for each step. We do not illustrate the steps in finalizing the rule because those steps do not overlap.

Estimated timeline for proposed rulemaking (23-44 months)

26. Generate potential control options for further analysis (2 months). The initial step for OTAQ would be to review the results of the anti-backsliding study and identify a list of potential fuel controls which exist that might be able to offset any adverse air quality impacts of renewable fuels, including an assessment of the magnitude of fuel control necessary to offset any negative emission impacts of renewable fuels. This could involve a literature review and potentially new analysis of data on the effects of fuels on vehicle emissions.

27. Assess technological feasibility of potential control options and screen them by their relative viability (2 months). Once we have identified one or more control options, we would assess the technological feasibility of refinery and/or other fuel market changes for the identified potential fuel control options. This could involve assessment of current industry configurations and processes, based on existing literature, consultation with industry experts, and compliance information collected by EPA. The time and resources necessary for this assessment will vary depending on what the potential control options are; if there is little available information or industry experience for an option, we would need to collect more information, meaning it will take longer to assess than an option for which there is extensive information and experience.

28. Conduct detailed analyses of identified fuel control options, including fuel production and distribution system costs and feasibility analyses (at least 5-8 months). EPA would need to conduct detailed analysis to confirm the viability of the control options and to identify the lead time necessary for industry to implement them. In addition, the time needed to conduct these analyses will depend on how many control options need to be assessed and compared. Analysis to support each control option under consideration could take 5 months, and analyzing more than one option could take longer. The major tasks in analyzing each option are:

   a. *Refinery modeling (at least 4 months)*. One important aspect of the detailed analysis is refinery modeling, which is used to assess the capital cost, operating cost, and lead time, as well as to develop the scenario-specific fuel supplies for every county as inputs to emissions analysis. Refinery models are very sophisticated cost optimization models which project the changes that refiners would make to comply with a specified fuel change. Because of their sophisticated nature, refinery models require at least 4 months to set up, run and interpret the results, regardless of whether they are run in-house or contracted out. If existing refinery models have not been configured to evaluate the potential fuel control options, then we would need to update them. New refinery modeling may well be required in order to determine how refineries will be able to comply with the new requirements, and that in turn drives questions about the costs of the regulation and its impacts on small businesses, which may drive further analysis or regulatory changes.

b. *Assess distribution system impacts (if any) (0 -3 months in parallel with paragraph 28a)*. The gasoline market has transformed itself in recent years such that the product which leaves the refinery and is shipped by pipeline is generally not finished gasoline. Rather, refiners largely manufacture "blendstock" which is then blended with ethanol and additives at a terminal before being distributed to retail stations. As a result, some fuel changes can affect the operation of refineries, pipeline and other fuel distribution companies, terminals which store the fuels downstream of the refinery, and even retail outlets and consumers. It is not possible to predict now whether an assessment of the cost and feasibility of distribution systems would be necessary, or whether its complexity would necessitate contractor assistance. But if an assessment is needed, we believe this step would take 1 to 3 months, and the work can be conducted in parallel with the other detailed analyses described in paragraph 28a.

c. *Assess lead time/phase-in implementation feasibility of control options (1 - 4 months)*. The costs and lead time associated with the control options depends upon the capital expenditures, time necessary to design and construct refinery modifications and, if necessary, to modify the downstream distribution and retail infrastructure. Because this assessment considers the results of the refinery modeling, these two tasks cannot be performed in parallel. The assessment would also entail gathering information from vendors, construction companies, pipelines, terminals, retailers, and other industry experts. The complexity of this assessment depends upon the nature of the fuel change required by the control option and the associated actions that refineries and downstream parties would have to undertake in order to comply.

29. Assess emissions impacts of control options (6-24 months, conducted in parallel with paragraph 28c). The emissions impacts of control options would be estimated using the MOVES emissions model. This analysis differs from the one performed for the anti-backsliding study, because it will need to estimate different scenarios (e.g., different fuel properties) and different analysis years (e.g., one or more future years). Furthermore, even if the anti-backsliding study included a scenario that would be relevant for the regulatory analysis, it would have to be re-run to use the most up-to-date inputs and information related to MOVES, as detailed below.

    a. First, we would need to consider whether the model's underlying data and structure are adequate for estimating the impacts of a given control option. If so, the emissions impact analysis (including paragraph 29b) would take approximately 6 months. If key data gaps exist, we may need to develop a new modeling algorithm, or design and implement a new vehicle/engine emissions test program. Depending on their complexity, new test programs require at least 18 months to develop, contract out, implement, and analyze the results. Any subsequent changes to the structure of the model would require still more time for algorithm development, coding and testing. Thus, the emissions impact analysis could take 24 months or even longer.

    b. We would also need to assure that the standard model inputs are up to date. Periodic updates to MOVES incorporate the latest travel statistics and other data, but, depending on the focus of the proposed regulation and where MOVES is in its development cycle, we may update specific inputs particularly relevant to the analysis.

30. Assess air quality impacts of control options (5-7 months). The process for assessing air quality impacts using SMOKE and CMAQ is described above. But the air quality analysis for rulemaking would differ from that for the study because the emissions estimates that are input to the air quality modeling will be different; as described in paragraph 29, there will be different scenarios and analysis years. The time needed to complete this analysis would depend on how many scenarios are being analyzed. A simple analysis with only one analysis year and two scenarios (a reference scenario and a control scenario) would take approximately 5 months. However, if there are multiple control options and/or multiple future years that are being analyzed, more CMAQ modeling runs, and more time, would be required. Furthermore, the length of time needed to post-process the modeling runs and generate summary deliverables such as maps, charts, etc. would depend on how many pollutants were being analyzed and also how many environmental impacts were relevant (e.g., visibility, nutrient deposition).

31. Assess human health, environmental, and other impacts of proposed rule (5 weeks). Using the ambient air quality results of the CMAQ model (for each scenario) as an input, we would use the Environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE) to translate the modeled air concentration estimates into health effects incidence estimates and monetized benefits estimates. Further QA and post-processing of the BenMAP-CE output would be needed to generate an analysis of the primary suite of impacts as well as additional uncertainty, sensitivity, and supplemental analyses that reflect the current state of the science. BenMAP-CE processing time would

take approximately one week and QA/post-processing of BenMAP-CE output would take an additional two weeks. Documenting results and drafting supporting text would take approximately two weeks.

32. Conduct cost-benefit analysis (5 months, conducted partially in parallel with steps described in paragraphs 29 to 31). Cost-benefit analysis is required under E.O. 12866 for economically significant rulemakings and costs and benefits are key considerations for rulemakings pursuant to Title II of the Clean Air Act generally. The benefits generally come from the assessment of human health, environmental, and other impacts. The cost analysis requires the results of refinery modeling and distribution/retail system analysis and some can be done in parallel with the emissions analysis and air quality modeling described in paragraphs 29-31. Once the benefits and costs are available, the benefit-cost analysis itself will take a week.

33. Conduct economic-impact analysis (5 weeks, conducted partially in parallel with paragraphs 31 and 32). Economic-impact analysis assesses the effects of potential controls for key sectors and stakeholders and commonly includes assessment of employment impacts. For some sectors or stakeholders, qualitative assessment of impacts is sufficient and requires fewer resources. But for other sectors and depending on the magnitude of the effects, quantification of employment impacts may be necessary. Much of the model development, expected to take about a month, could overlap in time with the cost analysis; final calculations will take a week. Economic impact analyses of fuel changes also typically assess potential energy security and energy independence effects of the action. Quantification of energy security impacts of the various control options is

expected to take about a month, following the selection of control options and in parallel with other economic analyses.

34. Develop regulatory program design (2-6 months, in parallel with emissions impact analysis in paragraph 29). Once EPA settles on a control option, we would need to design a regulatory structure to implement that option, which would include both a compliance program and regulatory flexibilities. If the fuels regulation impacts refiners in similar ways as past rules, then this effort may be relatively straightforward – a 2-3 month process of determining how to best fold the new provisions in to the existing fuel regulations. However, if the new control imposes novel requirements on refineries, or if it involves regulations for downstream fuel distributors, then it will require a much more extensive regulatory development process. This process could involve consultation with external stakeholders and other EPA components (such as the Office of Air Quality Planning and Standards and the Office of Enforcement and Compliance Assurance).

35. Regulatory Flexibility Act analysis and review panel (8-10 months, largely in parallel with steps described in paragraphs 28-29 and 34). Under the Regulatory Flexibility Act, EPA would need to determine whether the proposed rule would have a significant impact on a substantial number of small entities, such as small businesses. Our Initial Regulatory Flexibility Analysis needs to include a description and estimate of the number of small entities to which the rule would apply and an analysis of the rule's potential impact on them. This is informed by the cost and feasibility analyses described above as well as additional unique analysis using information developed by the Small Business Administration, employment information, and other cost analyses. Depending on the results of the Initial Regulatory Flexibility Analysis, EPA may need to convene a Small

Business Advocacy Review Panel to consider regulatory options and flexibilities to help mitigate potential adverse effects on small businesses. This statutorily-prescribed process involves the Small Business Administration, the Office of Management and Budget (OMB), and representatives from the industry sectors that would potentially be affected by the rule, and includes convening of a federal panel and outreach meetings with small entity representatives, and ultimately a final report that summarizes the panel's recommendations. The time needed for this process depends on whether a panel must be convened and, if so, the complexity of the issues involved (all unknown at this time).

36. Other Executive Orders, including Paperwork Reduction Act (2 months; some partially in parallel with other steps, including paragraphs 31-33). There are a number of analyses and processes required to comply with the Paperwork Reduction Act and multiple Executive Orders, such as those related to energy supply, distribution or use (E.O. 13211); environmental justice (E.O. 12898); protection of children from environmental health risks (E.O. 13045); and consultation and coordination with tribal governments (E.O. 13175). The Paperwork Reduction Act requires EPA to identify the information collection requirements and estimate their burden, and submit an Information Collection Request to OMB for public comment and ultimately OMB approval.

37. Development, review, and signature of *Federal Register* proposed rule package (7-10 months, partially in parallel with other steps, including paragraphs 30-33, 36). Developing a proposed rule for the Federal Register involves preparing a notice that explains and supports the proposed regulatory action, including background information; the options and alternatives considered; and the legal, policy and technical bases for the action. There is also a regulatory impact analysis and other technical support documents

that explain the various technical analyses supporting the proposed action. Although the process of drafting the notice can be concurrent with some of the steps described above, significant portions of the notice cannot be drafted until EPA's senior management offers initial direction to the technical, policy, and legal staff and then makes policy decisions based on the analyses and options they present. Once the proposal package is drafted, staff and management in offices across the agency, including the Assistant Administrator of the Office of Air and Radiation and the Associate Administrator of the Office of Policy, will review the draft and provide comments, which must be addressed before the draft is sent for review and signature by the Administrator. Before the Administrator's signature, a standard part of the rulemaking process involves interagency review through OMB, which takes up to 90 days.

Final rule development (13-26 months)

38. Public comment period and hearing (75 days from proposal signature). After the proposal is signed, it needs to be published in the Federal Register. The time it takes for publication depends on the length of the proposal, its complexity, and factors within the Office of Federal Register. EPA is required to take public comment and provide an opportunity for a public hearing. There must be sufficient advance notice of the hearing in the Federal Register, and the record for the proposed action must be open for public comment for at least 30 days after the (last) public hearing. We assume that it takes the Office of Federal Register 30 days to publish the proposal, and that the proposal would provide 15 days' notice for a public hearing. That is, a hearing would be held 15 days after publication. If we close the comment period 30 days after the hearing, the public would have a total of 45 days from the date of publication to comment, and the total time

from signature of the proposal to the close of the comment period is 75 days. But for significant, complicated rules, stakeholders usually ask for a total of at least 60 (as opposed to 45) days to comment, meaning the total time from signature of the proposal to the close of public comment is at least 90 days.

39. Consider public comments (3 months). Once the comment period is closed, EPA must review the comments received and evaluate whether those comments warrant further analyses or affect the technical analysis or the policy decisions. While some comments can be reviewed during the comment period, most commenters make their submissions in the final few days of the comment period. As a result, review of the public comments received on the proposed action largely occurs after the close of the comment period. The length of time needed to consider public comments depends on the number and complexity of comments. (This step is separate from actually responding to comments in light of the Agency's ultimate decision, which takes place later in the process and is described in paragraph 41.)

40. Update or redo analyses based on public comments, additional or more recent information, etc. (up to 10 months). If EPA receives or becomes aware of new data to inform the technical analyses conducted for the proposal, or if EPA decides to consider alternative control options that were not previously analyzed, additional work would be needed. Conducting new emissions analysis and air quality modeling could take eight or more months, depending on the complexity of the changes that are needed and whether new refinery modeling is needed to provide inputs. Although some analyses could be conducted in parallel with the emissions and air quality modeling, other analyses, such as

the assessment of human health and environmental impacts, use results from the air quality modeling as inputs.

41. Development, review, and signature of *Federal Register* package (7-10 months). Once the updated analysis is complete, EPA will again need to identify issues that require decisions from senior management, identify options for addressing those issues, prepare written briefing materials outlining the issues and options, and brief senior management. EPA will also need to develop final rulemaking materials including a notice of final rulemaking, final regulatory text, and technical support documents detailing the policy analysis underlying the final rulemaking. In addition to these materials, the EPA will need to prepare a response-to-comment document detailing the EPA's response to all significant comments received on the notice of proposed rulemaking, including responses to various policy, legal, and technical issues raised in the comments. EPA anticipates that any rulemaking which significantly impacts refineries or the broader transportation fuels market will attract intense interest from stakeholders and close scrutiny of EPA's analyses. Even if EPA ultimately does not conclude it is necessary to redo analyses in response to comments, it will still have to invest substantial time in considering the merits of highly technical comments and then responding in writing to each of them. EPA will also need to develop a revised regulatory impact analysis evaluating the policy being finalized. Once the notice of final rulemaking and associated support documents are drafted, staff experts and management in offices across the agency, including the Assistant Administrator of the Office of Air and Radiation and the Associate Administrator of the Office of Policy, will review the draft and provide comments, which must be addressed before the draft is sent for review and signature by the Administrator.

The process of briefings, drafting, and internal review is expected to take at least 7 months after analyses are complete. Finally, before the Administrator signs the final rule, a standard part of the executive branch's rulemaking process involves interagency review through OMB, which takes up to 90 days.

Section 7545(v)(2)(B) determination

42. The length of time needed for a determination that mitigation measures are not necessary will largely depend on the results of the anti-backsliding study and the complexity of issues surrounding any adverse air quality impacts it identifies. If the anti-backsliding study identified no adverse air quality impact at all, then relatively little additional analysis may be necessary, and EPA would need about 3 months to draft a notice explaining its determination. However, if there is some adverse impact on air quality, and if that impact could be mitigated through fuel controls, EPA could potentially need to undertake the entire range of analyses described in paragraphs 26 to 33 to assess whether mitigation measures are necessary, as well as additional time for drafting and internal review of its determination. Indeed, depending on the results of the anti-backsliding study, one of the purposes of the analyses described in paragraphs 26-33 may be to determine whether mitigation measures are necessary, and EPA may undertake those analyses without knowing whether the outcome will be a notice of proposed rulemaking or a determination that such rulemaking is unnecessary. Again, without the anti-backsliding study's results, we do not know precisely how long these analyses would take. Our best estimate for completing a determination based on these analyses is 23 to 44 months.

E. Conclusion

43. Considering the extensive emissions and air quality modeling involved in the anti-backsliding study, EPA believes that 14 months from the date of the Court's ruling on summary judgment is the most expeditious deadline EPA can meet to complete the steps described in paragraphs 14-21.

44. Because the results of the anti-backsliding study are critical to EPA's decisions about how to proceed under 42 U.S.C. § 7545(v)(2) with respect to mitigation measures, it is not currently possible to specify the appropriate schedule to take that action. Based on our best estimate at this time, it could take a total of 23 to 44 months for EPA to propose a rule to implement mitigation measures and 13 to 26 months to finalize a rule. Alternatively, if we determine that no mitigation measures are necessary, we estimate that it could take anywhere from 3 months to 44 months, depending on the existence and degree of adverse air quality impacts, and the analyses needed to assess whether mitigation measures are necessary.

I declare under penalty of perjury that to the best of my knowledge and belief the foregoing is correct.

Executed this 10th day of August, 2018.

Christopher Grundler, Director
Office of Transportation and Air Quality
Office of Air and Radiation
United States Environmental Protection Agency

Exhibit A: Proposed Rulemaking Timeline