RURAL LAND CHANGE AND THE CAPACITY FOR ECOSYSTEM CONSERVATION AND SUSTAINABLE PRODUCTION IN NORTH AMERICA

EDITED BY: Alisa W. Coffin, Mark A. Drummond, David R. Huggins and Fardausi Akhter

PUBLISHED IN: Frontiers in Sustainable Food Systems and Frontiers in Environmental Science

frontiers Research Topics

ISSN 1664-8714
ISBN 978-2-88974-938-6
DOI 10.3389/978-2-88974-938-6

About Frontiers

Frontiers is more than just an open-access publisher of scholarly articles: it is a pioneering approach to the world of academia, radically improving the way scholarly research is managed. The grand vision of Frontiers is a world where all people have an equal opportunity to seek, share and generate knowledge. Frontiers provides immediate and permanent online open access to all its publications, but this alone is not enough to realize our grand goals.

Frontiers Journal Series

The Frontiers Journal Series is a multi-tier and interdisciplinary set of open-access, online journals, promising a paradigm shift from the current review, selection and dissemination processes in academic publishing. All Frontiers journals are driven by researchers for researchers; therefore, they constitute a service to the scholarly community. At the same time, the Frontiers Journal Series operates on a revolutionary invention, the tiered publishing system, initially addressing specific communities of scholars, and gradually climbing up to broader public understanding, thus serving the interests of the lay society, too.

Dedication to Quality

Each Frontiers article is a landmark of the highest quality, thanks to genuinely collaborative interactions between authors and review editors, who include some of the world's best academicians. Research must be certified by peers before entering a stream of knowledge that may eventually reach the public - and shape society; therefore, Frontiers only applies the most rigorous and unbiased reviews.
Frontiers revolutionizes research publishing by freely delivering the most outstanding research, evaluated with no bias from both the academic and social point of view. By applying the most advanced information technologies, Frontiers is catapulting scholarly publishing into a new generation.

What are Frontiers Research Topics?

Frontiers Research Topics are very popular trademarks of the Frontiers Journals Series: they are collections of at least ten articles, all centered on a particular subject. With their unique mix of varied contributions from Original Research to Review Articles, Frontiers Research Topics unify the most influential researchers, the latest key findings and historical advances in a hot research area! Find out more on how to host your own Frontiers Research Topic or contribute to one as an author by contacting the Frontiers Editorial Office: frontiersin.org/about/contact

RURAL LAND CHANGE AND THE CAPACITY FOR ECOSYSTEM CONSERVATION AND SUSTAINABLE PRODUCTION IN NORTH AMERICA

Topic Editors:
Alisa W. Coffin, Agricultural Research Service, United States Department of Agriculture, United States
Mark A. Drummond, United States Geological Survey (USGS), United States
David R. Huggins, Agricultural Research Service, United States Department of Agriculture, United States
Fardausi Akhter, Agriculture and Agri-Food Canada (AAFC), Canada

Citation: Coffin, A. W., Drummond, M. A., Huggins, D. R., Akhter, F., eds. (2022). Rural Land Change and the Capacity for Ecosystem Conservation and Sustainable Production in North America. Lausanne: Frontiers Media SA.
doi: 10.3389/978-2-88974-938-6

Table of Contents

frontiers in Environmental Science

EDITORIAL
published: 25 March 2022
doi: 10.3389/fenvs.2022.850424

Editorial: Rural Land Change and the Capacity for Ecosystem Conservation and Sustainable Production in North America

Alisa W. Coffin[1], Fardausi Akhter[2], Mark A. Drummond[3] and David R. Huggins[1]*

[1]United States Department of Agriculture, Agricultural Research Service (USDA-ARS), Washington, DC, United States, [2]Agriculture and Agri-Food Canada (AAFC), Ottawa, ON, Canada, [3]United States Geological Survey (USGS), Reston, VA, United States

Keywords: land-use change, landscape conservation, sustainability, agro-ecosystems, North America

Editorial on the Research Topic

Rural Land Change and the Capacity for Ecosystem Conservation and Sustainable Production in North America

Rural landscapes across the globe are vital to the production of food, timber, energy, and other resources for an increasing human population. They are also essential for sustaining ecosystem health for future generations. Accordingly, the challenge for humanity is to advance global production systems while also conserving and even enhancing ecosystem services (Rockström et al., 2017), and recognizing tradeoffs (Lark et al., 2020). The failure to meet this challenge is critical, pushing against planetary constraints of our biosphere, with cascading and potentially catastrophic repercussions to human well-being (Steffen et al., 2015). For societies to thrive, the capacity for ecosystem conservation must be enhanced, and rural landscapes are widely recognized as a key geography for this capacity.

Research on North American lands has examined trends in rural land cover (Sleeter et al., 2013), including urbanization (Brown et al., 2005; Sohl, 2016), woody encroachment (Bailey et al., 2010), 20th century cropping patterns (Sohl et al., 2016), and the periodic transitions of production forestry (Drummond et al., 2015). The dynamics of land change have been linked to multiple drivers associated with economics, policy, population, and climate (Napton et al., 2010; Drummond et al., 2012; Mcphee et al., 2021).

Emerging research increasingly emphasizes concepts that agriculture and nature can and should co-exist in ways that provide for people and healthy ecosystems (Kleinman et al., 2018; Spiegal et al., 2018; Mcphee et al., 2021). Developing a better understanding of human-environment dynamics in rural landscapes, including proximal and distant interactions (Liu et al., 2007), is critical.

The collection of papers in this research topic responded to this aim, identifying key aspects of rural landscapes in North America. The authors' approach ranged from broad examinations of national and regional trends, to more focused models addressing specific biophysical components of agroecosystems.

Contributions to this research topic included a pair of papers aimed at social and economic dimensions of agroecosystems. These include an improved framework for incorporating human well-being by Bentley Brymer et al., and advanced concepts of telecoupling (or "pericoupling") by Spiegal et al. to evaluate alternative strategies of beef production supply chains. Both papers

OPEN ACCESS

Edited and reviewed by:
André Mascarenhas, Humboldt University of Berlin, Germany

****Correspondence:***
Alisa W. Coffin
alisa.coffin@usda.gov

Specialty section:
This article was submitted to Land Use Dynamics, a section of the journal Frontiers in Environmental Science

Received: *07 January 2022*
Accepted: *25 February 2022*
Published: *25 March 2022*

Citation:
Coffin AW, Akhter F, Drummond MA and Huggins DR (2022) Editorial: Rural Land Change and the Capacity for Ecosystem Conservation and Sustainable Production in North America. Front. Environ. Sci. 10:850424. doi: 10.3389/fenvs.2022.850424

challenge researchers to consider the perspectives and perceptions of producers. They also push us to think about relationships occurring outside of our immediate geographies by incorporating deeper meanings of "community", on the one hand, and the complex linkages among alternative strategies of production, on the other.

National agricultural policy was addressed by Spangler et al., who found that policies described in United States Farm Bills have broadened in purpose and influence over time, favoring the expansion of commodity crop production and limiting support for diversification. Examining the finer details, Medina et al. explored farmer perspectives on federal conservation programs in Iowa, noting that, while limited, conservation programs played a role in incentivizing the adoption of conservation practices, a finding which supports research by others (Piñeiro et al., 2020). In addition to these, policy needs in Canada with regards to reducing chemical inputs to agroecosystems were noted by Banger et al., and Malaj et al.

Drivers, trends, and patterns of rural land use in North America were examined at regional and national scales. Goslee explored these issues in-depth, modeling the importance of climate, soils, and irrigation as drivers of crop diversity and change for the conterminous United States. Irrigation emerged as a key explanatory variable in models of crop diversity, suggesting that increases in irrigation could result in increased agricultural diversity. However, while biophysical drivers of change in crop diversity were less clear, Spangler et al., suggest that national policy is a key driver of broad trends in crop diversity. Although calculated differently, crop diversity trends showed similar results at broad national scales for both papers, with the highest levels of diversity found in California, the Great Lakes area, the Northern Great Plains, and the Southeast. In contrast, the lowest levels of diversity were found in the central regions of the United States

Regional landscape and land use patterns were the subjects of three studies in this collection. Analysis of regional trends in land use, irrigation, and streamflow by Yasarer et al. showed that low flow conditions in rivers of the Mississippi River Alluvial Plain have been significantly altered over the last fifty years. Drastic increases of irrigated cropland were associated with lower flows, increasing days with no flow, and declining groundwater levels. In the adjacent Southeastern Plains, an examination by Coffin et al. showed that the balance of tradeoffs among ecosystem services varies across the region and at multiple scales. Conservation indicators were stronger in Florida than other areas, with supporting services provided by larger embedded natural systems and low intensity working lands there. Galpern and Gavin also conducted multi-scale analyses in the Canadian Prairie Croplands, examining the distribution and variability of uncultivated areas within agricultural fields. Their work emphasized the importance of scale and the underlying environmental gradients for both understanding patterns of non-crop areas within agricultural fields, and determining potential areas for management. The importance of the intentional planning, design, and evaluation of natural systems in working lands naturally arises out of these studies as an exciting new area of research. To this end, Kröbel et al. demonstrated and tested a tool for shelterbelt components for the *Holos* model, a whole-farm model for evaluating carbon and other greenhouse gas budgets of alternative farm designs. Their work upgrades the model from an age-determined to a circumference-determined calculation to estimate the above- and below-ground carbon for field shelterbelts.

Nutrient management and chemical use were addressed in four studies that also incorporated modeling approaches. Each of these studies considered the subject at vastly different scales of analysis. At the farm level, Banger et al., showed that the returns accruing from environmentally optimal nitrogen (N) rates are significant but require a tradeoff in net farm income, which they opine could be offset by policies that compensate farmers for their economic loss. At a broader, regional level, Mezbahuddin et al. used *Ecosys* process-based modeling to simulate alternative N fertilizer management scenarios. Their predictions that spring banding in Alberta would lower N-species emissions and runoff were validated with empirical estimates, and demonstrated the value of the agroecosystem modeling approach. Across the southeastern United States, Coffin et al., summarized modeled N runoff from previous work. They found that watersheds in Georgia had lower levels of N runoff than those in Florida, pointing to the significant buffering capacity of riparian forested areas. At the national scale, Malaj et al., found that agrochemical use in Canada has increased rapidly and systematically, but these increases vary by region and by agrochemical type. Fertilizer increases were associated with increasing oilseeds and soybeans and decreasing cereal crops in the Prairie and Central cropland regions. More alarming, however, were the substantial increases in fungicides and insecticides in these areas.

This collection of papers points to lessons that enhance our understanding of how changes in rural lands affect the dual capacity for conservation and production. However, the complexity of evaluating the tradeoffs among ecosystem services that result from interacting suites of conservation practices requires long-term, convergent approaches to scientific research. In North America, working lands constitute one of the greatest opportunities to enhance regional and global capacity for ecosystem conservation.

AUTHOR CONTRIBUTIONS

AC, FA, MD and DH devised the concept for the editorial. AC drafted the manuscript. FA and MD provided editorial comments and revisions. AC finalized the manuscript and submitted.

ACKNOWLEDGMENTS

This research was a contribution from the Long-Term Agroecosystem Research (LTAR) network. LTAR is supported by the United States Department of Agriculture.

REFERENCES

Bailey, A. W., Mccartney, D., and Schellenberg, M. P. (2010). "Management of Canadian Prairie Rangeland," in AAFC No 10144. *Agriculture and Agri-Food Canada).* 1.

Brown, D. G., Johnson, K. M., Loveland, T. R., and Theobald, D. M. (2005). Rural Land-Use Trends in the Conterminous United States, 1950-2000. *Ecol. Appl.* 15, 1851–1863. doi:10.1890/03-5220

Drummond, M. A., Auch, R. F., Karstensen, K. A., Sayler, K. L., Taylor, J. L., and Loveland, T. R. (2012). Land Change Variability and Human-Environment Dynamics in the United States Great Plains. *Land Use Policy* 29, 710–723. doi:10.1016/j.landusepol.2011.11.007

Drummond, M. A., Stier, M. P., Auch, R. F., Taylor, J. L., Griffith, G. E., Riegle, J. L., et al. (2015). Assessing Landscape Change and Processes of Recurrence, Replacement, and Recovery in the Southeastern Coastal Plains, USA. *Environ. Manage.* 56, 1252–1271. doi:10.1007/s00267-015-0574-1

Kleinman, P. J. A., Spiegal, S., Rigby, J. R., Goslee, S. C., Baker, J. M., Bestelmeyer, B. T., et al. (2018). Advancing the Sustainability of US Agriculture through Long-Term Research. *J. Environ. Qual.* 47, 1412–1425. doi:10.2134/jeq2018.05.0171

Lark, T. J., Spawn, S. A., Bougie, M., and Gibbs, H. K. (2020). Cropland Expansion in the United States Produces Marginal Yields at High Costs to Wildlife. *Nat. Commun.* 11, 4295. doi:10.1038/s41467-020-18045-z

Liu, J., Dietz, T., Carpenter, S. R., Folke, C., Alberti, M., Redman, C. L., et al. (2007). Coupled Human and Natural Systems. *AMBIO: A J. Hum. Environ.* 36, 639–649. doi:10.1579/0044-7447(2007)36[639:chans]2.0.co;2

Mcphee, C., Bancerz, M., Mambrini-Doudet, M., Chrétien, F., Huyghe, C., and Gracia-Garza, J. (2021). The Defining Characteristics of Agroecosystem Living Labs. *Sustainability* 13, 1718. doi:10.3390/su13041718

Napton, D. E., Auch, R. F., Headley, R., and Taylor, J. L. (2010). Land Changes and Their Driving Forces in the Southeastern United States. *Reg. Environ. Change* 10, 37–53. doi:10.1007/s10113-009-0084-x

Piñeiro, V., Arias, J., Dürr, J., Elverdin, P., Ibáñez, A. M., Kinengyere, A., et al. (2020). A Scoping Review on Incentives for Adoption of Sustainable Agricultural Practices and Their Outcomes. *Nat. Sustainability* 3, 809–820.

Rockström, J., Williams, J., Daily, G., Noble, A., Matthews, N., Gordon, L., et al. (2017). Sustainable Intensification of Agriculture for Human prosperity and Global Sustainability. *Ambio* 46, 4–17. doi:10.1007/s13280-016-0793-6

Sleeter, B. M., Sohl, T. L., Loveland, T. R., Auch, R. F., Acevedo, W., Drummond, M. A., et al. (2013). Land-cover Change in the Conterminous United States from 1973 to 2000. *Glob. Environ. Change* 23, 733–748. doi:10.1016/j.gloenvcha.2013.03.006

Sohl, T., Reker, R., Bouchard, M., Sayler, K., Dornbierer, J., Wika, S., et al. (2016). Modeled Historical Land Use and Land Cover for the Conterminous United States. *J. Land Use Sci.* 11, 476–499. doi:10.1080/1747423x.2016.1147619

Sohl, T. (2016). "Southeastern Plains," in *Status and Trends of Land Change in the Eastern United States—1973 to 2000*. 1. Editors K. L. Sayler, W. Acevedo, and J. Taylor (Reston, VA: U.S. Geological Survey).

Spiegal, S., Bestelmeyer, B. T., Archer, D. W., Augustine, D. J., Boughton, E. H., Boughton, R. K., et al. (2018). Evaluating Strategies for Sustainable Intensification of US Agriculture through the Long-Term Agroecosystem Research Network. *Environ. Res. Lett.* 13, 034031. doi:10.1088/1748-9326/aaa779

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E.M., et al. (2015). Planetary Boundaries: Guiding Human Development on a Changing Planet. *Science* 347, 1259855. doi:10.1126/science.1259855

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's Note: All claims expressed in this article are solely those of the authors and do not necessarily represent those of their affiliated organizations, or those of the publisher, the editors and the reviewers. Any product that may be evaluated in this article, or claim that may be made by its manufacturer, is not guaranteed or endorsed by the publisher.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 31 March 2020
doi: 10.3389/fenvs.2020.00031

Assessing the Potential to Increase Landscape Complexity in Canadian Prairie Croplands: A Multi-Scale Analysis of Land Use Pattern

Paul Galpern and Michael P. Gavin*

Department of Biological Sciences, University of Calgary, Calgary, AB, Canada

OPEN ACCESS

Edited by:
Fardausi Akhter,
Agriculture and Agri-Food Canada (AAFC), Canada

Reviewed by:
Michael W. Strohbach,
Technische Universitat Braunschweig, Germany
Wenche E. Dramstad,
Norwegian Institute of Bioeconomy Research (NIBIO), Norway

***Correspondence:**
Paul Galpern
paul.galpern@ucalgary.ca

Specialty section:
This article was submitted to Land Use Dynamics, a section of the journal Frontiers in Environmental Science

Received: *16 September 2019*
Accepted: *03 March 2020*
Published: *31 March 2020*

Citation:
Galpern P and Gavin MP (2020) Assessing the Potential to Increase Landscape Complexity in Canadian Prairie Croplands: A Multi-Scale Analysis of Land Use Pattern. Front. Environ. Sci. 8:31. doi: 10.3389/fenvs.2020.00031

Increasing natural vegetation in agricultural landscapes can create habitat for beneficial organisms such as pollinators and the natural enemies of crop pests. Adding perennial vegetation can also support biodiversity conservation and climate change mitigation objectives. However, implementing such changes to agricultural land use across large geographic areas will require a strategic approach. This study examined the amount and distribution of uncultivated areas in Canadian prairie croplands, focusing on Alberta's agricultural zone (226,543 km^2). The aim was to identify locations in this region that have potential for increasing non-crop land cover within fields. This assessment was based on a multi-scale model of landscape complexity that described the distribution of land cover as a function of the distance from field centers. It is based on the assumption that the land cover in the field neighborhood is an informative index of how much non-crop area might realistically be maintained or restored in the field itself; i.e., because neighboring lands will reflect the local environmental conditions that support the growth and establishment of non-crop vegetation as well as the likelihood that crop growers will remove areas from production. The model identified variation across the region in land cover distribution, with regions at latitudes between 52°N and 55°N demonstrating the greatest contrasts in the amount of non-crop land between the field and the field neighborhood scale. These findings suggest that there remains capacity for land use decision-makers to optimize the distribution of non-crop land covers in ways the reduce the differences between these scales (i.e., to increase non-crop covers within fields to better represent the neighborhood proportions). Modeling also revealed scale-dependent patterns, such as field margins without crops (400–500 m from field centers) broadly distributed across the region, and evidence that gradients in moisture and temperature have interacted with land use decisions to shape the proximity of non-crop area to fields.

Keywords: ecological intensification, ecosystem services, landscape simplification, semi-natural habitat, perenniality, Canadian prairie, agroecosystems

INTRODUCTION

Increasing agricultural production to meet a growing global demand will require expansion and intensification of croplands (Godfray et al., 2010; Laurance et al., 2014; Egli et al., 2018), but this presents challenges for mitigating climate change and conserving biodiversity (Kleijn et al., 2009; Bustamante et al., 2014; Dalu et al., 2017; Karp et al., 2018). Ecological intensification of agriculture may offer a compromise by leveraging ecosystem services provided by organisms that boost yields while also minimizing impacts on natural systems (Bommarco et al., 2013).

One proposal is to promote non-crop vegetation within or near fields, which would reduce the simplification of agricultural landscapes caused by clearing and field expansion (Landis, 2017). Heterogeneity in non-crop land covers (hereafter landscape complexity), creates a greater number of off-field habitats providing opportunities for "spillover" into the crop (Birkhofer et al., 2018). The beneficial organisms that use these habitats may provide regulating services such as pest control and pollination to the surrounding crop. For example, complex agricultural landscapes have been associated with higher abundance and diversity of birds (Boesing et al., 2017), bats (Ancillotto et al., 2017), flower-visiting insects (Duarte et al., 2018) and the natural enemies of crop pests (Chaplin-Kramer et al., 2011), among other organisms.

There are further arguments for increasing landscape complexity. Restoring or augmenting semi-natural and other non-crop vegetation is an opportunity for carbon storage when trees, shrubs, and other perennial plants are permitted to grow (Smith, 2014; Lamb et al., 2016; Hungate et al., 2017; Williams et al., 2018). It may also support several other regulating ecosystem services through an increase in plant diversity (Asbjornsen et al., 2014), and conserve habitat for organisms that may not have direct benefits to agriculture (Phalan et al., 2011; Tscharntke et al., 2012).

However, meaningfully achieving these "win-win-wins" for agriculture, biodiversity and climate change mitigation will require that land use initiatives be implemented at broad geographic extents. Determining which parts of an agricultural region may be more or less amenable to improving landscape complexity can be used to target early efforts to those areas where there is greatest chance of success. For example, it may be easier to convince growers and land-owners to take land out of production in areas where there is already evidence of higher landscape complexity, but it is not uniformly distributed across all fields. In effect, the landscape context provides a measure of what improvements may be feasibly implemented.

This study examine landscape complexity in the Canadian Prairies. The region ranks among the world's largest contiguous agroecosystems, and has a cultivated footprint of more than 0.5 million km^2 (Agriculture and Agri-Food Canada, 2015). The focus is cropland in Alberta, which is distributed across most of the environmental and land use variability of these temperate grasslands (Ecological Stratification Working Group, 1995).

The primary aim is to identify which parts of this region have the highest potential for improving landscape complexity. That is, to identify where introducing additional non-crop land cover into fields and their surroundings may face the fewest challenges to implementation. For example, land conversion decisions may face resistance because natural vegetation is slow or costly to establish in certain regions, or because the land is highly-productive and value is placed on maximizing the area in production.

Finding such areas is done by building a spatial model of the distribution of non-crop land cover and how it varies with proximity to field centers. In this multi-scale approach, distributions of land cover at broad scales provide a target for the introduction of land cover within fields. Secondary aims of this study are to identify patterns in the distribution of these land covers, and how these may be associated with the broad environmental gradients that structure both vegetation and productivity.

DATA AND METHODS

Study Area

The geographic focus is Alberta's agricultural zone, one of the most intensively cropped regions in the world (Foley et al., 2005), and an area that represents 30% of Canada's cropland and

TABLE 1 | Data layers used to produce a binary (crop/non-crop) land cover map for Alberta, Canada at 30 m spatial resolution.

Data	Features	Citation
Agriculture and Agri-Food Canada Crop Inventory (2017)	Crop land covers (e.g., fields by crop type; pasture and hay lands)	Annual Space-Based Crop Inventory for Canada (2017). Center for Agroclimate, Geomatics and Earth Observation, Science and Technology Branch, Agriculture and Agri-Food Canada. https://open.canada.ca/data/en/dataset/ba2645d5-4458-414d-b196-6303ac06c1c9
Alberta Biodiversity Monitoring Institute Human Footprint Layer (2016)	Non-crop land covers (e.g., roads, rail corridors, urban, natural and disturbed vegetated surfaces)	Wall-to-Wall Human Footprint Inventory. (2016). Alberta Biodiversity Monitoring Institute. http://abmi.ca/home/data-analytics/da-top/da-product-overview/GIS-Land-Surface/HF-inventory.html
Alberta Merged Wetland Inventory	Non-crop land covers (wetlands)	Alberta Merged Wetland Inventory. (2017). Alberta Environment and Parks, Government of Alberta. https://maps.alberta.ca/genesis/rest/services/Alberta_Merged_Wetland_Inventory/Latest/MapServer/
OpenStreetMap (2017)	Non-crop land covers (roads; rail corridors)	OpenStreetMap contributors. (2017). https://planet.osm.org/

Input data sources were equal to, or coarser than, the original spatial data sources. The order of overlay of these layers is as listed below. Land cover classes of similar types were aggregated.

20% of its annual farm income (Statistics Canada, 2016; Alberta Agriculture and Forestry, 2017). The dominant land cover is cropland, primarily in a 3-years cereal grain, oilseed, and pulse rotation. Forage lands are a secondary land cover vegetated in both introduced and native perennial grasses that are used for pasture and hay. Other semi-natural areas not in crop production are found both within and adjacent to fields, and they vary in frequency, size and type spatially along environmental gradients. These include patches of perennial grasses, shrubs, and deciduous trees. Planted tree shelterbelts, grass and forb road margins and wetlands of different classes, often surrounded in perennial vegetation, are also common throughout most of the study area (Doherty et al., 2018).

Land Cover

To characterize the variation in land cover across the region, a composite raster map was produced at 30 m resolution by combining data from several published sources in the order listed (**Table 1**). The product included a variety of land cover classes which were then simplified thematically to crop and non-crop areas. Non-crop areas also included paved and other human-modified areas such as roads, buildings, farm yards and in-field oil and gas well pads as these areas are likely to be surrounded in perennial vegetation and therefore contribute to landscape complexity (Forman, 2009). The binary thematic resolution of the land cover map was necessary given that better resolved and accurate products do not exist for this large extent. While this restricts inference about the types of vegetation that may contribute to landscape complexity, it is a simplification that may improve interpretation by reframing land cover as a land use decision (i.e., for crop production or otherwise).

Measuring Landscape Complexity

Landscape complexity was measured from the land cover map using randomly-selected field centers as sampling locations following an algorithmic approach implemented in R. Probable fields were identified using a nineteenth-century land survey which consistently divided the province of Alberta into 259 ha (1 mile by 1 mile) sections (Larmour, 2005). As a consequence of the gridding of the region, fields typically are nested within a section, with the quarter section (square subdivisions; 805 m by 805 m) describing the boundaries of most fields. The field centers (**Figure 1**) were randomly selected from all quarter section centroids with the conditions: (1) that the quarter must contain at least 50% crop cover (to ensure it at least partly represents a crop field); and (2) that the centers be no closer than 2.5 km

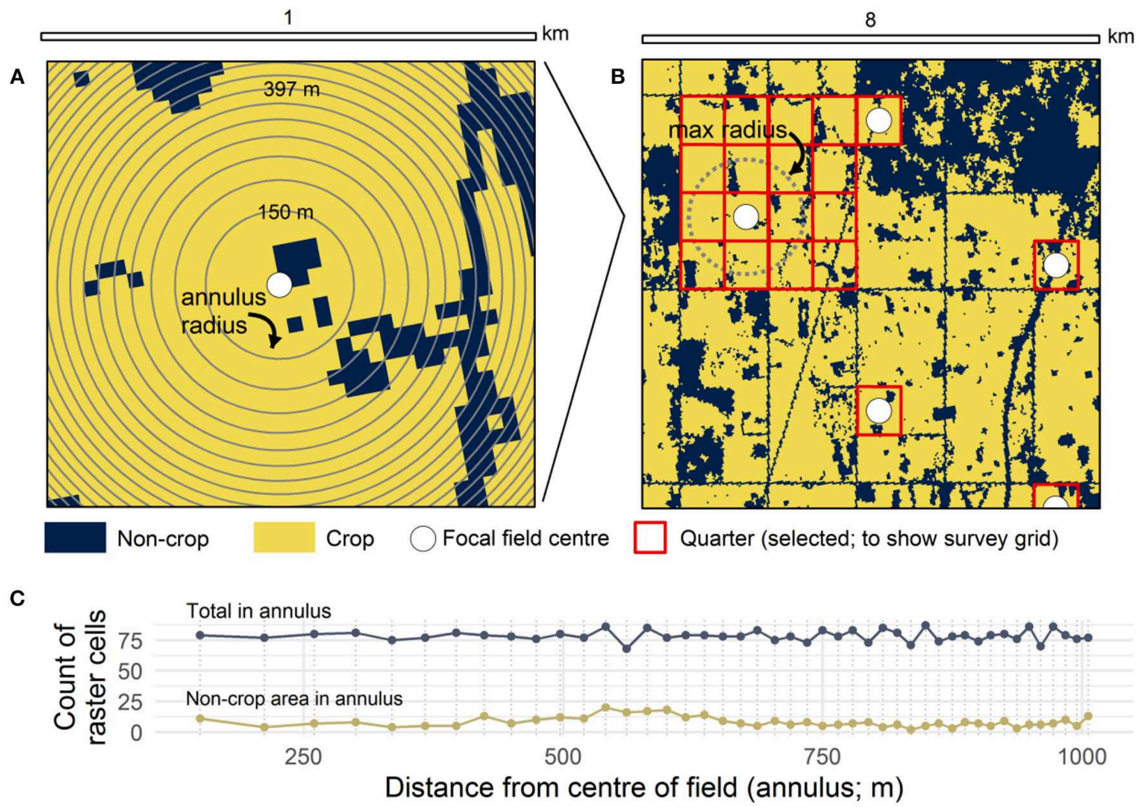


FIGURE 1 | Illustration of the multi-scale method for measuring the proximity of non-crop land covers to field centers. **(A)** Forty-five annuli of decreasing radii, but approximately equal area, are measured from each field center; **(B)** Field centers are positioned at the center of quarter sections, the spatial unit that describes most fields in the region, and spaced a minimum of 2.5 km apart; **(C)** Counts of non-crop area raster cells and the total number of cells in each annulus in A used to find proportional cover. Non-crop raster cells are counted in the smallest annulus in which they fall.

apart (to improve statistical independence). These conditions were implemented using an algorithm that iteratively tested a randomly-permuted list of field centers for inclusion until no more could be added.

Measuring landscape complexity requires assessing the variation in landscape structure. Metrics that have been used to describe landscape complexity are typically measured at a certain radius from a focal location, for example, by calculating the area of focal land covers, the proportional composition of crop, or the mean habitat patch size (e.g., Boesing et al., 2017). Because the distance from a field at which a non-crop land cover may have an effect is typically not known, this study uses

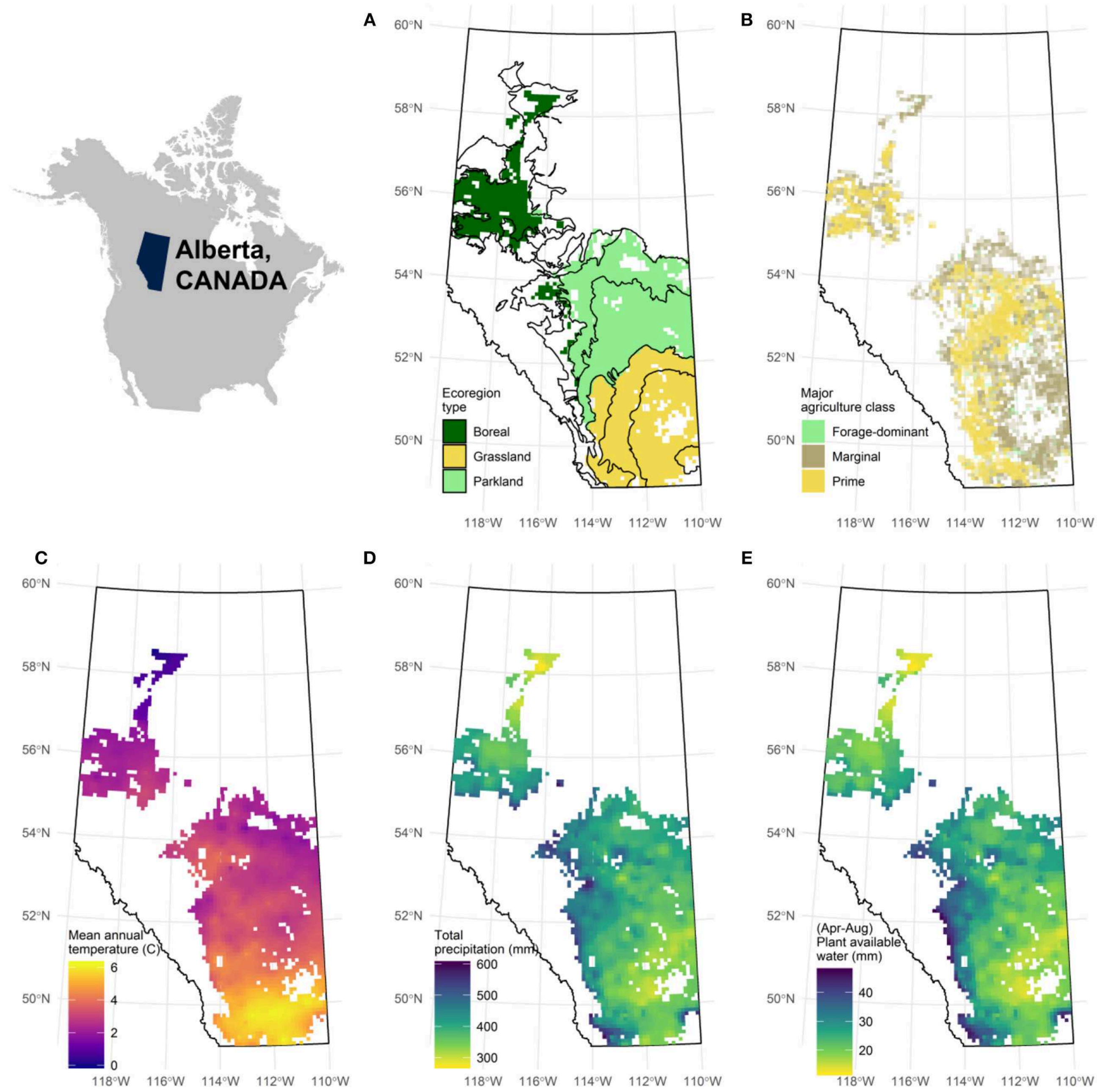


FIGURE 2 | Environmental gradients in Alberta, Canada used as covariates when modeling landscape complexity. A colored pixel on the map represents a survey township containing agricultural land (*N* = 2,430). **(A)** Agricultural land intersects with 12 ecoregions (polygons) which can be broadly grouped according to their dominant non-crop vegetation; **(B)** Prime and marginal agricultural lands as determined by the limitations associated with cultivation; **(C)** Mean annual temperature (9 years means; 2009–2017); **(D)** Total precipitation (9 years means; 2009–2017); **(E)** Plant available water in the top 120 cm of soil found as a daily mean between April and August (9 years means; 2009–2017).

an alternative approach that creates a multivariate index (or a "curve") integrating the amount of land cover over multiple distances. This landscape complexity curve is determined by finding the proportion of non-crop area in 45 annuli of increasing radii (150–1,006 m) each of which is pre-determined to cover approximately the same total area to minimize sampling bias associated with smaller annuli (**Figures 1A,C**). Pixels from the rasterized land cover map were recorded in the annulus with the smallest radius in which they fell.

Landscape complexity, as defined here, is therefore a multi-scale metric for the proximity of non-crop land covers to the crop and more generally as an index of the variation in non-crop area both within and in the vicinity of a probable crop field. Annuli with the smallest radii describe conditions within the field itself (150–400 m; "within-field scale"), intermediate radii describe the field margins (400–500 m; "field-margin scale") and larger radii which sample from multiple neighboring fields and semi-natural areas capture broader landscape variation (500–1,006 m; "neighborhood scale"). The neighborhood scale is intended to capture the local land cover conditions, and is applied in this study as an achievable upper target for restoring or augmenting non-crop land cover within the field itself. Because this scale samples neighboring fields it also measures how much of the landscape is in crop production.

Potential Environmental Drivers of Landscape Complexity

The study region encompasses several environmental gradients that influence both the dominant type of vegetation and primary productivity (e.g., temperature, moisture, latitude, and elevation; Ecological Stratification Working Group, 1995). These gradients also have the potential to affect the frequency of clearing and the rate of natural regrowth of vegetation, and may therefore interact with the behavior of land use decision-makers to shape the observed landscape complexity. Five covariates were chosen to test the importance of these environmental drivers (**Figure 2**). Mean annual temperature, total annual precipitation and plant available water in 120 cm of soil (April through August) were calculated as 9-years means (2009–2017) from an interpolated data product at the resolution of a survey township (36 sections; 93 km^2; Alberta Agriculture and Forestry, 2018). Also included were agricultural limitations for crop production, a classification chiefly based on the workability of the soil by mechanized agricultural equipment (Agriculture and Agri-Food Canada, 2012), and ecoregions, a well-established categorical assessment of contiguous areas with consistent climate characteristics, similar vegetation, soil types, and elevation (Ecological Stratification Working Group, 1995).

Modeling Landscape Complexity at Multiple Scales

Landscape complexity curves were modeled using a type of functional data analysis (FDA) known as function-on-scalar regression (Kokoszka and Reimherr, 2017; Wood, 2017). These models have functions (i.e., the landscape complexity curve) rather than the typical scalar values as response variables. A continuous function $Y(x)$, was estimated from discrete data (e.g., **Figure 1C**, light line) using the proportion of non-crop area at each annulus radius, x. The model had the general form,

$$Y(x) = \mu(x) + \alpha_{p(i)} + \alpha_{p(i)}(x) + \beta_q(q, x) + \beta_{r_1 r_2}(r_1, r_2, x) + \ldots + \epsilon(x)$$

TABLE 2 | Modeling of non-crop land cover variation.

	(a) Smooth terms edf[a]		(b) Parametric terms estimate	
Intercept (spline smooth and parametric)	9.90	*	−1.59	*
Northing × Easting (tensor product smooth)	444.36[b]	*		
Mean annual temperature (tensor product smooth)	37.96	*		
Total precipitation (tensor product smooth)	38.91	*		
Plant available water (tensor product smooth)	37.71	*		
Ecoregion effects (spline smooth and parametric)				
Clear hills upland	5.91	*	[c]	
Peace lowland	7.86	*	0.10	*
Mid-boreal uplands	5.97	*	0.36	*
Wabasca lowland	4.71	*	0.04	*
Western boreal	7.00	*	0.37	*
Western alberta upland	6.45	*	0.35	*
Boreal transition	7.49	*	0.35	*
Aspen parkland	0.51	*	0.22	*
Moist mixed grassland	7.65	*	0.22	*
Fescue grassland	6.81	*	0.22	*
Mixed grassland	6.14	*	0.29	*
Cypress upland	7.14	*	0.45	*
Agricultural limitation effects (spline smooth and parametric)				
Prime (None)	8.20	*	0.12	*
Prime (Moderate)	8.68	*	0.34	*
Marginal (Moderately severe)	7.65	*	0.52	*
Marginal (Severe)	2.83	*	0.54	*
Marginal (Unusable)	6.52	*	0.68	*
Mostly forage (Can improve)	8.00	*	0.72	*
Mostly forage (Cannot improve)	8.06	*	0.48	*
Unclassified	4.52	*	0.49	*
Organic soils	6.52	*	0.12	*

(a) *Significant ecoregion and agricultural limitation smooth terms ($P < 0.01$) imply that the function estimating continuous variation in non-crop land cover differed from zero, where higher edf values indicate greater non-linearity.* ***(b)*** *Significant parametric terms ($P < 0.01$) imply that the average amount of non-production land cover in a municipal district (i.e., averaged over all annulus distances) differed from the reference municipal district mean.*
**$P < 0.01$.*
[a] *Estimated degrees of freedom.*
[b] *Note that in a regression with a functional response, a single variable smooth is entered as a tensor product (i.e., two-dimensional smooth), resulting in a much higher estimated degrees of freedom.*
[c] *Reference ecoregion for parametric effects.*

where $\mu(x)$ is the intercept function (i.e., the mean landscape complexity curve), $\alpha_{p(i)}$ and $\alpha_{p(i)}(x)$ represent the constant effect and functional effect, respectively, of level i of categorical variable p, $\beta_q(q, x)$ reflects the functional effect of continuous variable q, $\beta_{r_1 r_2}(r_1, r_2, x)$ is the functional spatial effect given northing, r_1, and easting, r_2, and $\epsilon(x)$ is the residual function.

The model was fit using mgcv, a generalized additive modeling (GAM) package for R (Wood, 2017) using restricted maximum likelihood, by first setting up models with the refund package for R (Goldsmith et al., 2019) which provides optimized settings for FDA. The model assumed a Binomial data-generating process. Categorical constant effects were estimated parametrically, while categorical and continuous functional effects were estimated non-parametrically, with $\mu(x)$ and $\alpha_{p(i)}(x)$ functions entered as splines, and $\beta_q(q, x)$ or $\beta_{r_1 r_2}(r_1, r_2, x)$ functions modeled as two- or three-dimensional tensor products, respectively. Spatial coordinates were projected and included in the model to account for spatial autocorrelation and to model geographic patterns not represented in other covariates. The maximum number of knots that could be used for estimating spline functional terms was set at 11 as a balance between overfitting and modeling abrupt changes in non-crop area over annulus distance. All covariates were scaled and centered.

Interpretation of the model involved examining the coefficient functions $\alpha_{p(i)}(x)$ and $\beta_q(q, x)$ or generating predictions by excluding certain terms and systematically selecting input values to examine scenarios of interest. For example, predictions controlling for environmental covariates were generated by setting the relevant categorical functional or continuous functional terms to the zero function and entering categorical parametric coefficients as constants at the level with the most observations. Mapping was done by including the spatial term, $\beta_{r_1 r_2}(r_1, r_2, x)$, but not other environmental covariates, and generating predictions for the centroids of survey townships. The potential for increasing landscape complexity was assessed by finding the difference in the mean non-crop area prediction between neighborhood and within-field scales for the centroid of each survey township.

RESULTS

The data set represented 14,527 randomly-selected quarter sections with a mean crop area of 79%, and a mean distance from nearest neighbor of 2,891 m. These quarter sections occupied 2,430 survey townships with a mean of 6 quarter sections in each (min = 1, max = 14), collectively sampling from an agricultural study region of 226,543 km^2. Forty-five annuli ranging from 150 m to 1,006 m in radius were assessed for each quarter (mean area per annulus = 7.1 ha). Data from the sampled quarters and their surroundings covered 6% of the total area under study.

The model deviance explained was 32.2% ($R^2_{adj} = 0.355$) with all constant parametric coefficients for categorical terms significantly different from zero ($P < 0.01$; **Table 2a**) suggesting the average proportion of non-crop area differed by ecoregion and agricultural limitation class. All functional non-parametric coefficient functions significantly differed from zero ($P < 0.01$; **Table 2b**) demonstrating a non-linear relationship between non-crop area and the distance from field centers, and that the included covariates mediate this relationship. The spatial term summarizing the effect of unmodelled geographic variables also had an effect on the landscape complexity curve (Northing × Easting; **Table 2b**).

The average trend in the proximity of non-crop area to field centers is that inside fields (150–400 m) there tend to be lower proportions of non-crop area than in the surrounding landscape (500–1,006 m; **Figure 3**). Thus, focal crop fields were not typically surrounded on all sides by other crop fields (a situation that would produce no difference between these two scales). However, there is spatial variation across the region in the proximity of non-crop areas to fields, after controlling for environmental covariates. This pattern is evident in the geographic differences in the predicted amount of non-crop cover at the within-field scale (e.g., **Figure 4A**), and at the neighborhood scale (e.g., **Figures 4C,D**). The field margin scale (400–500 m; e.g., **Figure 4B**) is more conserved across the province.

Most ecoregions (**Figure 5**) and agricultural limitation classes (**Figure 6**) had a characteristic landscape complexity curve, with

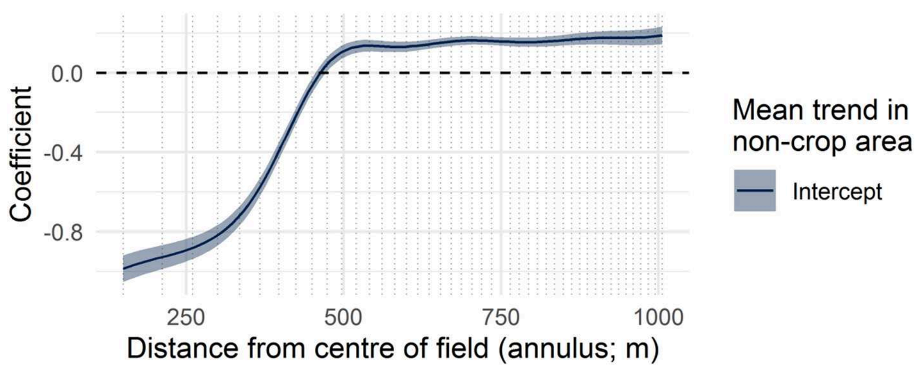


FIGURE 3 | The model intercept function (or mean landscape complexity curve) shown with two standard errors. The curve represents the estimated effect on the proportion of non-crop area and how it varies over distance from the center of an average field. Vertical dotted lines in this and subsequent figures indicate annulus radii at which fields were measured, providing support for the model.

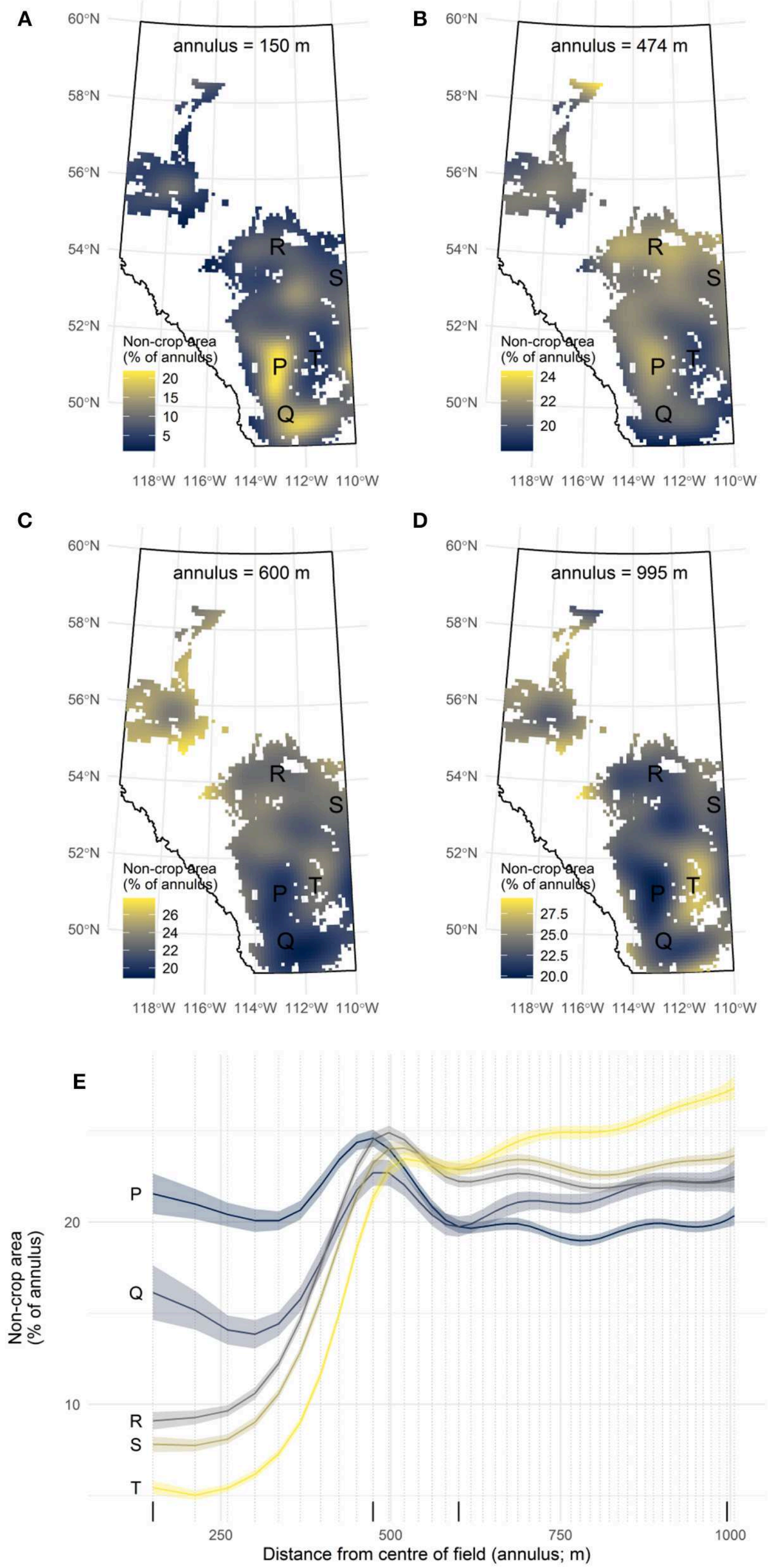


FIGURE 4 | Predictions from the model while controlling for all covariates except the spatial term. Non-crop area is shown at four different annuli to demonstrate: **(A)** the field scale; **(B)** the field margin scale; **(C,D)** the neighborhood scale. **(E)** Landscape complexity curves at four locations (*P, Q, R, S,* and *T*) represent predictions across all scales and are intended to illustrate how the entire curve varies over the study region.

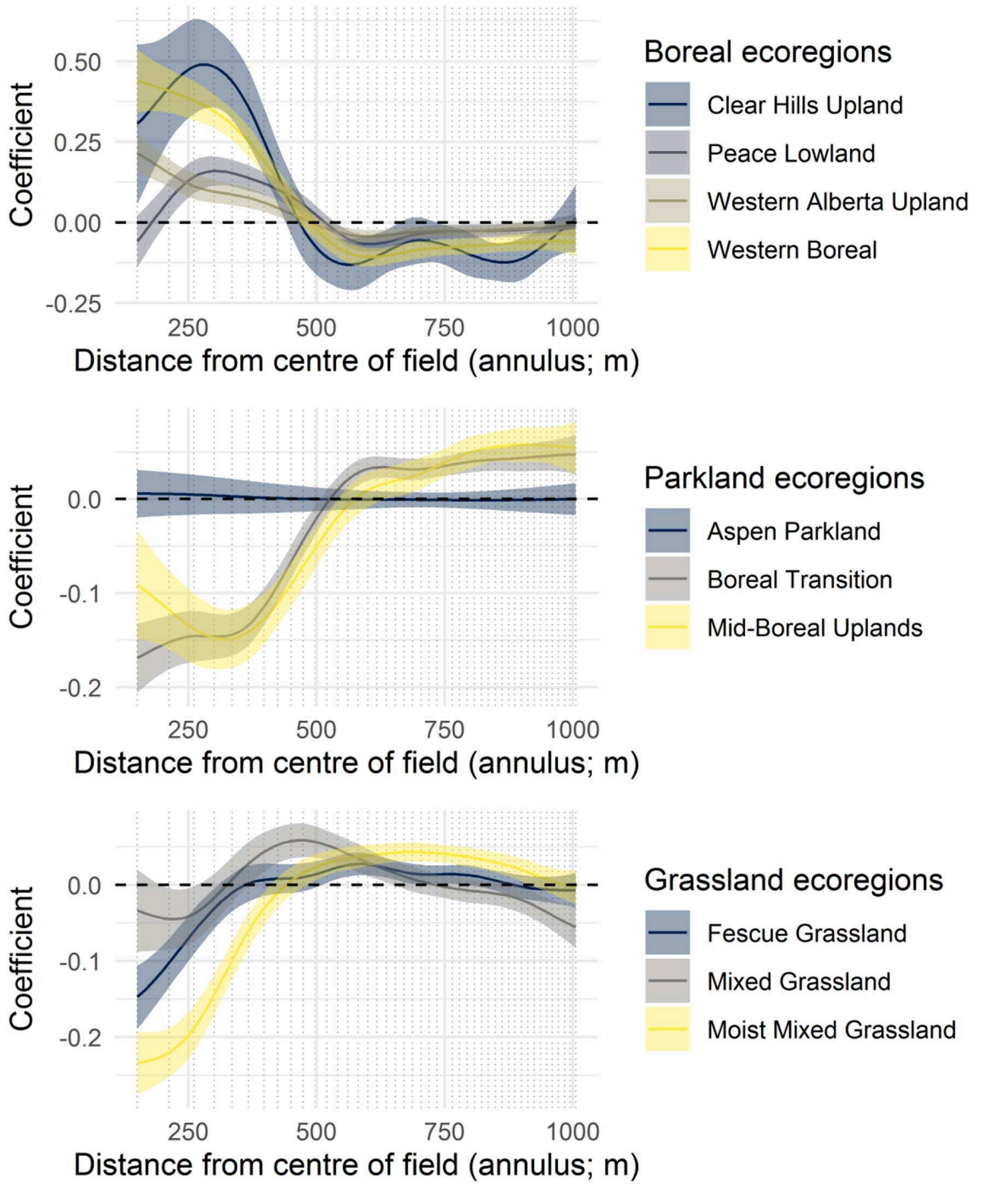


FIGURE 5 | The coefficient functions for twelve ecoregions, grouped by their dominant non-crop vegetation. Curves demonstrate deviations from the mean landscape complexity curve (**Figure 3**).

these coefficient function plots demonstrating deviations in the coefficient from the mean landscape complexity curve (i.e., from **Figure 1**). All climatic variables also exerted effects on the shape of the landscape complexity curves (**Figure 7**). Trends for scales larger than the field margin scale (400–1,006 m) were generally similar for temperature and moisture-related variables. At the within-field scale, higher temperatures, and drier soils resulted in less non-crop area (mean annual temperature, plant available water; **Figure 7**).

Mapping of a multi-scale index shows that there is geographic variation in the difference between the proportion of non-crop covers at the neighborhood and the within-field scales. Larger

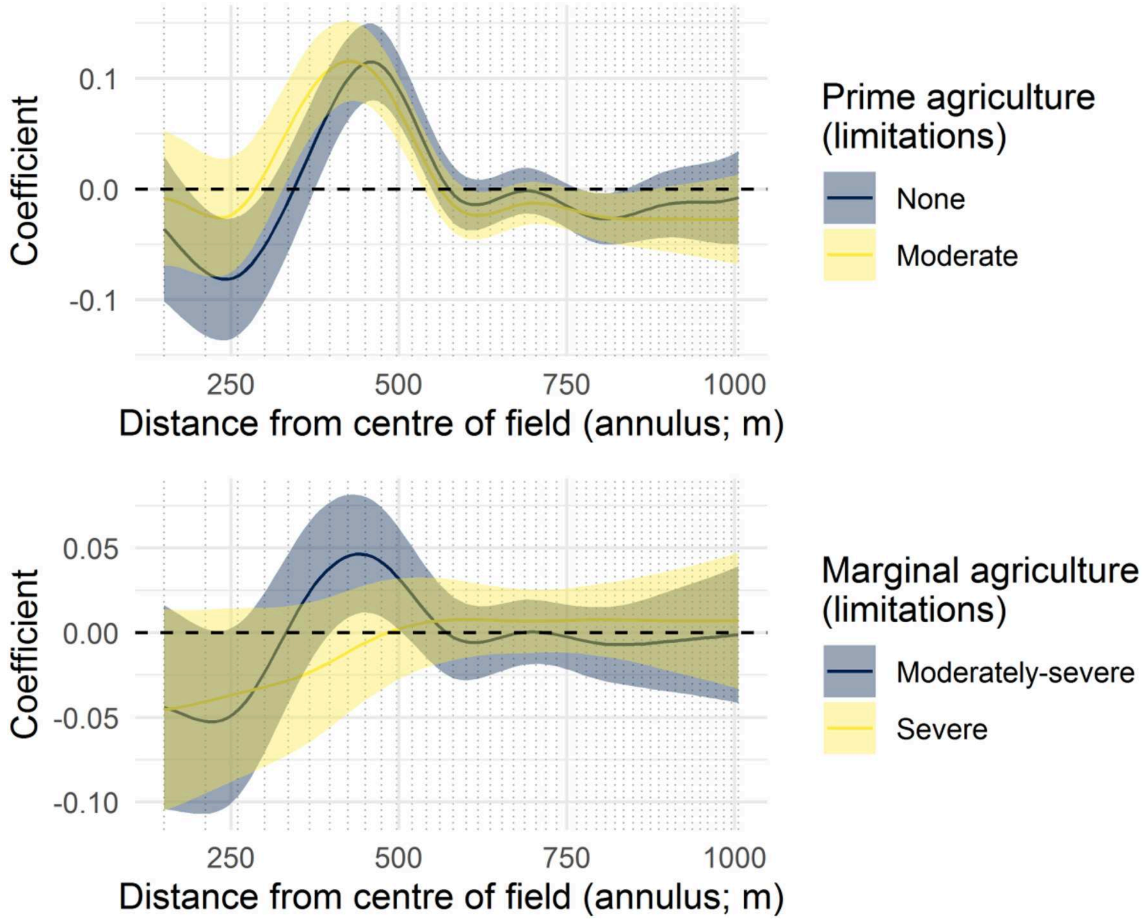


FIGURE 6 | The coefficient functions for four agricultural limitation classes. Curves demonstrate deviations from the mean landscape complexity curve (**Figure 3**).

differences (darker colors; **Figure 8**) indicate that fields in these townships have much less non-crop cover than their neighborhoods. Smaller differences (lighter colors; **Figure 8**) suggest fields are closer to the local optimum for non-crop areas.

DISCUSSION

This study demonstrated that there is considerable geographic variation in the proximity of non-crop areas to fields across Alberta's agricultural zone, and there remains capacity for growers and other land use decision makers to optimize this distribution. Analysis of landscape complexity curves revealed scale-dependent patterns in the proximity of non-crop areas to fields, such as the widespread occurrence of uncultivated field margins, and how land uses have been influenced by broad environmental gradients. These findings are considered in turn.

Potential for Changing Landscape Complexity

The capacity to introduce more non-crop land covers into fields, and therefore improve landscape complexity, can be seen by contrasting the proportions of non-crop area found within-fields to the neighborhood surrounding those fields (**Figure 8**). Non-crop area at the neighborhood scale can be understood as an estimate of the local potential for this quantity. Annuli at this scale sample from eight neighboring quarter sections, many of which may also be fields, effectively summarizing a region eight times the size of the focal field (**Figure 1**). The neighborhood scale, then, can be taken as a realistic potential proportion of non-crop area in the focal field because it captures neighboring land owners' willingness to allow those non-crop land covers to persist. It is also likely to be a correlate of the local expectation for crop productivity, given that neighboring growers would have an incentive to drain wetlands, clear trees and shrubs, remove perennial grasses, cultivate to fence lines or otherwise remove non-crop land covers from their fields, if that land could be used profitably for growing crops.

For example, the model suggests that within fields at location P (annulus = 150 m; **Figures 4A,E**) there is about 20% coverage in non-crop area, and there is also a similar coverage in the neighborhood (annulus = 995 m; **Figures 4D,E**). While 20% coverage suggests that non-crop covers are relatively common in fields, the more interesting observation is that there tend to be adjacent fields with similar non-crop proportions. This emerges

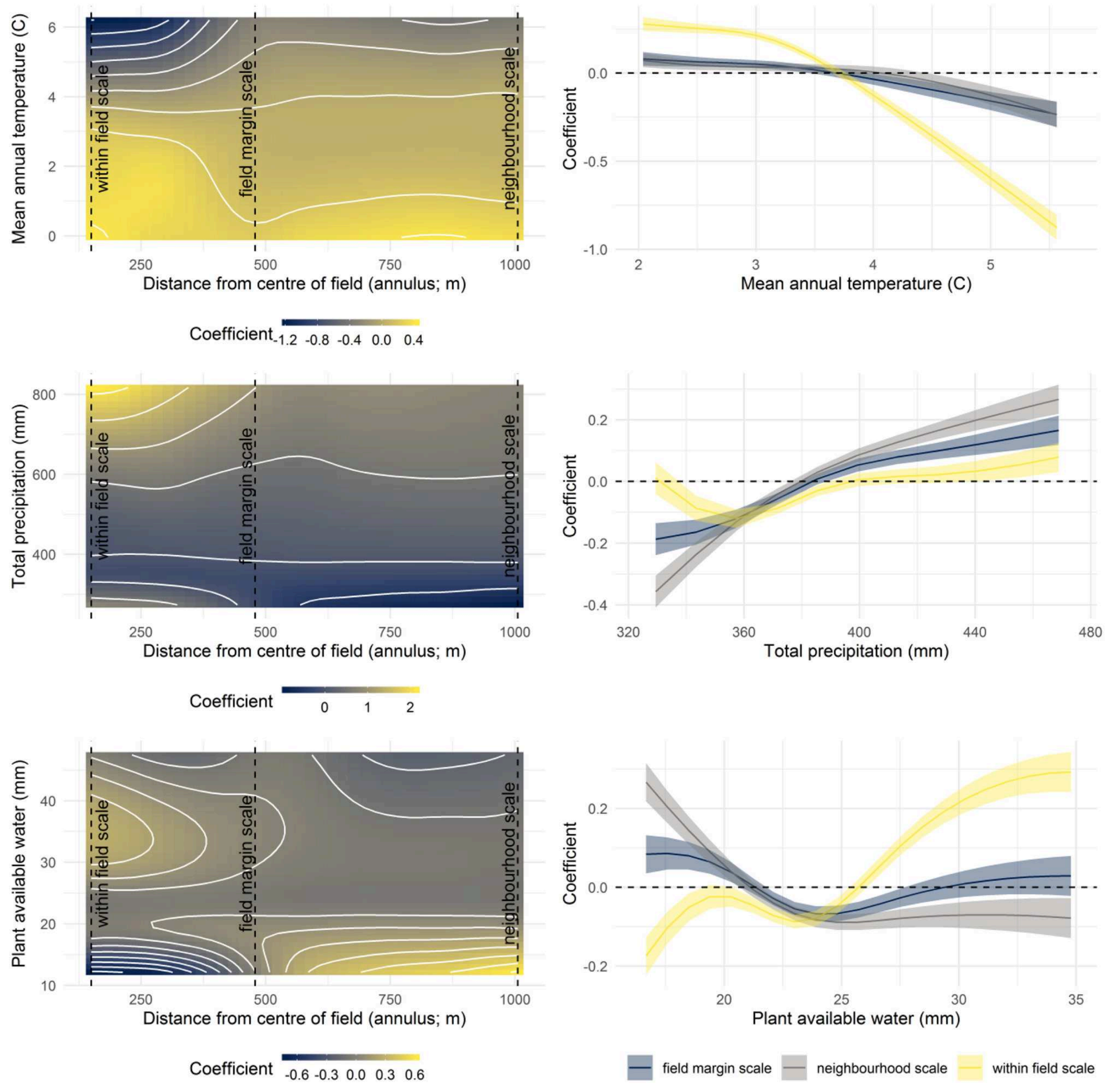


FIGURE 7 | (**Left**) Bivariate coefficient functions for continuous environmental variables, where coefficients give deviations from the mean landscape complexity curve (**Figure 3**). (**Right**) The effect of the environmental variable across 95% of the observed distribution of that variable shown at three selected scales.

as a flatter curve, and it is interpreted here as evidence that landscape complexity has reached its local potential; i.e., that land owners generally concur that 80% of the land should be in crops. The contrasting trend is evident at location *T*, where there is lower coverage of non-crop area within focal fields (about 5%) and higher coverage at the neighborhood scale (about 25%). Here fields have a low amount of non-crop area, but there is a large difference between the field and the neighborhood. This appears as a steeper curve, offering evidence that fields have less non-crop area than the local potential; i.e., that land owners differ markedly in how much land should be in crops.

Figure 8 (darker shades) therefore identifies parts of Alberta where agricultural land uses may be out of equilibrium with what the landscape can sustain, either because there may be more of each field in crop production than is optimal, or because there remains potential to further clear natural land covers for agriculture. The latter is probably the case in northern areas where twenty-first century clearing of boreal forest for

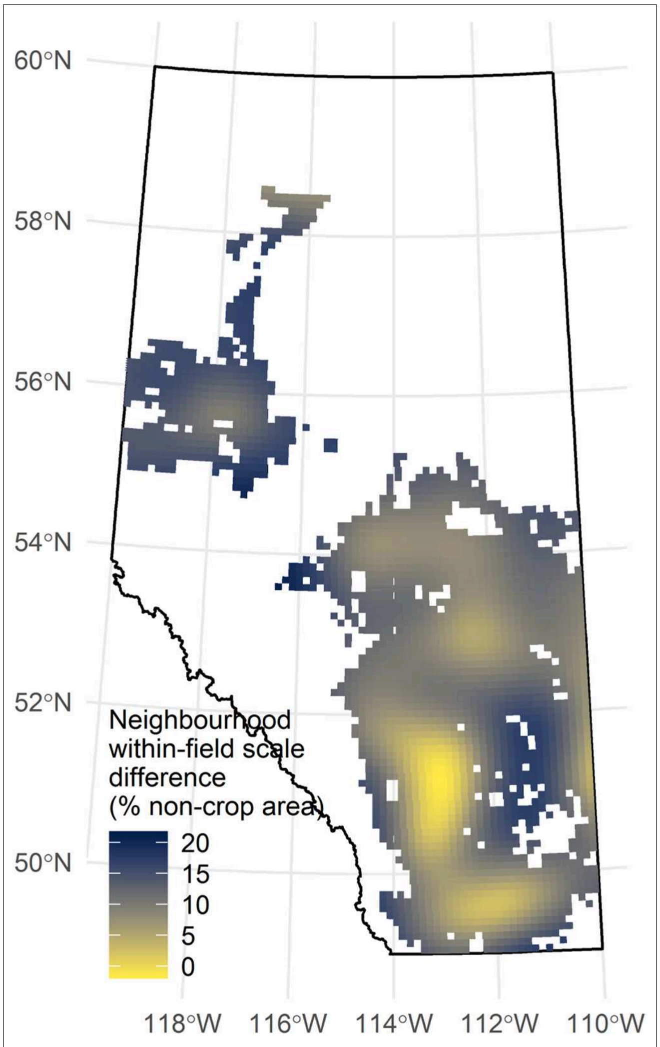


FIGURE 8 | The difference in the predicted non-crop area between the within-field (150–400 m) and neighborhood (500–1,006 m) scales for each of the 2,480 survey townships with agricultural land. Model predictions exclude all environmental covariates but include spatial terms.

agriculture is ongoing (north of 55°N; Bowen, 2002). However, the opportunity to clear more land is unlikely to be broadly important further south where agriculture has had an impact on the land since the late nineteenth century (Larmour, 2005).

The parts of the province that have the greatest difference at these two scales are those between 52°N and 55°N, after excluding the northern region and a region in south-central Alberta that is predominantly forage land (e.g., near point *T;* **Figure 4**). These latitudes are mostly in the parkland ecoregions, where prime agricultural land is abundant (**Figure 2**). There, growers may be able to leverage the collective wisdom of their neighbors to guide the return of non-crop areas to their fields. Or, in more concrete terms: marginal areas of fields that are currently cropped could be changed to non-crop vegetation; and forgoing production in these areas would mirror the decisions that neighbors have, on average, made regarding land use. This study, however, cannot provide any insight into how much non-crop land cover could be recovered. Rather it demonstrates that there are parts of the province where patterns in land use suggest the barriers doing so should be smaller than others.

Making the assumption that non-crop land covers in fields improve ecosystem services to agriculture (e.g., Rusch et al., 2016; Duarte et al., 2018; Vickruck et al., 2019), there is also the possibility that a small loss in crop area and therefore in farm returns, may be balanced by the improvement in the yields and profitability of crops on the remaining land. This can further reduce the risk associated with removing crop and replacing it with non-crop land covers.

Increasing non-crop land covers within fields could, in many parts of this region, be achieved by identifying marginal areas with relatively low productivity (e.g., measured in crop yield), removing them from production, and allowing other non-crop vegetation to re-establish. The proliferation of precision agricultural tools (e.g., yield monitors in harvesting equipment) should assist growers in identifying such sites (Mulla, 2013). These may be low spots in fields that have excess soil moisture or other poor soil conditions, or they may be near to other features that reduce productivity such as in the margins of wetlands where soils are poorly optimized for crop growth.

This study has not given any consideration to the class of non-crop cover and its association with landscape complexity, in part because this simplification aids interpretation. However, types of vegetation may differ in the ecosystem services they can provide to nearby crops, suggesting that regionally-targeted research on the benefits of establishing different vegetation classes is certainly appropriate. Equally, biasing re-establishment efforts toward perennial plants may better support carbon storage objectives (Hungate et al., 2017; Williams et al., 2018). However, the classes of vegetation already common at the neighborhood scale may be those that are the easiest to re-establish, and these may also be the species that naturally reclaim these sites without any intervention from land use decision-makers.

Patterns in the Proximity of Non-crop Area to Crops

Analysis of landscape complexity curves reveal field margins (400–500 m from field centers) as a common location for non-crop cover. These are evident as a peak at locations *P, Q, R, S,* and *T* (**Figure 4E**) at the scale that corresponds to the expected survey grid spacing. Geographically, the field margin effect is found throughout much of the region, with the notable exception of the extreme south of Alberta (annulus = 474 m; **Figure 4B**).

Field margins have been celebrated as a means to maintain non-crop areas in agricultural landscapes and to bring the ecosystem services they may provide close to fields with minimum loss of crop area (e.g., Marshall, 2002). In many cases these field margins are adjacent to roads or road allowances, which are public lands and are therefore at low risk of being changed to other land uses. Their widespread geographical distribution in Alberta represents an opportunity for regional policies that systematically promote their enhancement (e.g., by altering mowing regimes, or planting with native vegetation known to support ecosystem service provision; Kirmer et al., 2018).

Environmental Drivers of Landscape Complexity

Broad environmental gradients have played a role in shaping landscape complexity resulting in different patterns of non-crop area across scales. As might be expected, there is evidence that these gradients have influenced land use decisions, particularly the amount of clearing within fields and the density of fields at the neighborhood scale. Boreal ecoregions tended to have more non-crop area within fields (150–400 m; **Figure 5**) than the regional mean (**Figure 3**), perhaps reflecting the more recent clearing of these northern areas (Bowen, 2002) and the greater speed with which shrubs and trees can re-establish in the moisture and temperature regime of this part of the province. Grassland and parkland ecoregions registered at or less than the mean at the within field scale. The Mixed Grassland ecoregion had the most vegetation in field margins (400–500 m) while two ecoregions with significant tree cover (Mid-Boreal Uplands, and Boreal Transition) had the most non-crop area at neighborhood scales, reflecting the substantially lower amount of agricultural activity in these regions (**Figure 5**; Ecological Stratification Working Group, 1995).

Agricultural limitation classes demonstrated there has been more removal of vegetation within fields where there are fewer challenges to crop production in terms of the workability of the soil (**Figure 6**). Crop fields that have been successfully established in areas with severe limitations to agriculture suggest they are similar to the surrounding landscape in terms of non-crop area, and field margins are not distinguishable (e.g., marginal agriculture; **Figure 6**).

Temperature, precipitation and soil moisture variables enabled a directional assessment of how climatic variation is associated with landscape complexity patterns. Notably, locations with warmer temperatures and less soil moisture, conditions which favor prairie grassland species (Ecological Stratification Working Group, 1995), had less non-crop area within fields (**Figure 7**). The low frequency of woody vegetation in such dry prairie conditions means that clearing fields of all non-crop area is easier and less costly. Overall, the studies of environmental gradients indicate that landscape complexity is primarily under the control of land use decision-makers and not merely a result of local conditions, suggesting it is a matter of incentivizing these changes and not a problem simply of what the environment can sustain.

The multi-scale approach used in this study provided a flexible means to characterize landscape complexity as the proximity of non-crop features to crop fields. Interpretation of this model revealed that there is potential to increase landscape complexity by leveraging the natural potential in each region to support non-crop land cover and promoting practices that foster such vegetation, for example, on marginal or low-productivity sites within fields.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

PG and MG were involved in the writing of the manuscript. PG was primarily responsible for the completion and interpretation of analyses. MG compiled the composite land cover data product on which this analysis was based.

FUNDING

This research was supported by research grants from Natural Sciences and Engineering Research Council (NSERC).

REFERENCES

Agriculture and Agri-Food Canada (2012). *Land Capability for Agriculture*. Canada Land Inventory (CLI). Retrieved from: http://www.data.gc.ca

Agriculture and Agri-Food Canada (2015). *AAFC Annual Crop Inventory*. Government of Canada. Retrieved from: http://open.canada.ca/

Alberta Agriculture and Forestry (2017). *Agriculture Statistics Yearbook*.

Alberta Agriculture and Forestry (2018). *Alberta Climate Information Service (ACIS)*. Retrieved from: https://agriculture.alberta.ca/acis/

Ancillotto, L., Ariano, A., Nardone, V., Budinski, I., Rydell, J., and Russo, D. (2017). Effects of free-ranging cattle and landscape complexity on bat foraging: implications for bat conservation and livestock management. *Agric. Ecosyst. Environ.* 241, 54–61. doi: 10.1016/j.agee.2017.03.001

Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., et al. (2014). Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renewable Agric. Food Syst.* 29, 101–125. doi: 10.1017/S1742170512000385

Birkhofer, K., Andersson, G. K. S., Bengtsson, J., Bommarco, R., Dänhardt, J., Ekbom, B., et al. (2018). Relationships between multiple biodiversity components and ecosystem services along a landscape complexity gradient. *Biol. Conserv.* 218, 247–253. doi: 10.1016/j.biocon.2017.12.027

Boesing, A. L., Nichols, E., and Metzger, J. P. (2017). Effects of landscape structure on avian-mediated insect pest control services: a review. *Landsc. Ecol.* 32, 931–944. doi: 10.1007/s10980-017-0503-1

Bommarco, R., Kleijn, D., and Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends Ecol. Evol.* 28, 230–238. doi: 10.1016/j.tree.2012.10.012

Bowen, D. (2002). Agricultural expansion in Northern Alberta. *Geogr. Rev.* 92, 503–525. doi: 10.2307/4140932

Bustamante, M., Robledo-Abad, C., Harper, R., Mbow, C., Ravindranat, N. H., Sperling, F., et al. (2014). Co-benefits, trade-offs, barriers and policies for greenhouse gas mitigation in the agriculture, forestry and other land use (AFOLU) sector. *Glob. Change Biol.* 20, 3270–3290. doi: 10.1111/gcb.12591

Chaplin-Kramer, R., O'Rourke, M. E., Blitzer, E. J., and Kremen, C. (2011). A meta-analysis of crop pest and natural enemy response to landscape complexity. *Ecol. Lett.* 14, 922–932. doi: 10.1111/j.1461-0248.2011.01642.x

Dalu, T., Wasserman, R. J., and Dalu, M. T. B. (2017). Agricultural intensification and drought frequency increases may have landscape-level consequences for ephemeral ecosystems. *Glob. Change Biol.* 23, 983–985. doi: 10.1111/gcb.13549

Doherty, K. E., Howerter, D. W., Devries, J. H., and Walker, J. (2018). Prairie Pothole Region of North America. In *The Wetland Book,* eds C. Finlayson, G. R. Milton, R. C. Prentice, and N. C. Davidson (Dordrecht: Springer), 1–10.

Duarte, G. T., Santos, P. M., Cornelissen, T. G., Ribeiro, M. C., and Paglia, A. P. (2018). The effects of landscape patterns on ecosystem services: meta-analyses of landscape services. *Landscape Ecol.* 33, 1247–1257. doi: 10.1007/s10980-018-0673-5

Ecological Stratification Working Group (1995). *A National Ecological Framework for Canada.*

Egli, L., Meyer, C., Scherber, C., Kreft, H., and Tscharntke, T. (2018). Winners and losers of national and global efforts to reconcile agricultural intensification and biodiversity conservation. *Glob. Change Biol.* 24, 2212–2228. doi: 10.1111/gcb.14076

Foley, J. A., Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., et al. (2005). Global Consequences of Land Use. *Science*. 309, 570–574.

Forman, R. T. T. (2009). *Urban Ecology: Science of Cities.* Cambridge.

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., et al. (2010). Food security: the challenge of feeding 9 billion people. *Science* 327, 812–818. doi: 10.1126/science.1185383

Goldsmith, J., Scheipl, F., Huang, L., Wrobel, J., Gellar, J., Harezlak, J., et al. (2019). Refund: Regression with Functional Data. *R Package Version 0.1-17.* Retrieved from: https://cran.r-project.org/package=refund.

Hungate, B. A., Barbier, E. B., Ando, A. W., Marks, S. P., Reich, P. B., van Gestel, N., et al. (2017). The economic value of grassland species for carbon storage. *Science Adv.* 3:e1601880. doi: 10.1126/sciadv.1601880

Karp, D. S., Frishkoff, L. O., Echeverri, A., Zook, J., Juárez, P., and Chan, K. M. A. (2018). Agriculture erases climate-driven β-diversity in Neotropical bird communities. *Glob. Change Biol.* 24, 338–349. doi: 10.1111/gcb.13821

Kirmer, A., Rydgren, K., and Tischew, S. (2018). Smart management is key for successful diversification of field margins in highly productive farmland. *Agric. Ecosyst. Environ.* 251, 88–98. doi: 10.1016/j.agee.2017.09.028

Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E. D., Clough, Y., et al. (2009). On the relationship between farmland biodiversity and land-use intensity in Europe. *Proc. Biol. Sci.* 276, 903–909. doi: 10.1098/rspb.2008.1509

Kokoszka, P. and Reimherr, M. (2017). *Introduction to Functional Data Analysis.* New York, NY: CRC Press.

Lamb, A., Green, R., Bateman, I., Broadmeadow, M., Bruce, T., Burney, J., et al. (2016). The potential for land sparing to offset greenhouse gas emissions from agriculture. *Nature Clim. Change* 6, 488–492. doi: 10.1038/nclimate2910

Landis, D. A. (2017). Designing agricultural landscapes for biodiversity-based ecosystem services. *Basic Appl. Ecol.* 18, 1–12. doi: 10.1016/j.baae.2016.07.005

Larmour, J. (2005). *Laying Down the Lines: A History of Land Surveying in Alberta*. Brindle and Glass. Victoria, BC.

Laurance, W. F., Sayer, J., and Cassman, K. G. (2014). Agricultural expansion and its impacts on tropical nature. *Trends Ecol. Evol.* 29, 107–116. doi: 10.1016/j.tree.2013.12.001

Marshall, E. J. P. (2002). Introducing field margin ecology in Europe. *Agric. Ecosyst. Environ.* 89, 1–4. doi: 10.1016/S0167-8809(01)00314-0

Mulla, D. J. (2013). Twenty five years of remote sensing in precision agriculture: key advances and remaining knowledge gaps. *Biosyst. Eng.* 114, 358–371. doi: 10.1016/j.biosystemseng.2012.08.009

Phalan, B., Onial, M., Balmford, A., and Green, R. E. (2011). Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science* 333, 1289–1291. doi: 10.1126/science.1208742

Rusch, A., Chaplin-Kramer, R., Gardiner, M. M., Hawro, V., Holland, J., Landis, D., et al. (2016). Agricultural landscape simplification reduces natural pest control: a quantitative synthesis. *Agric. Ecosyst. Environ.* 221, 198–204. doi: 10.1016/j.agee.2016.01.039

Smith, P. (2014). Do grasslands act as a perpetual sink for carbon? *Glob. Change Biol.* 20, 2708–2711. doi: 10.1111/gcb.12561

Statistics Canada (2016). *Census of Agriculture*. Statistics Canada. Ottawa, ON.

Tscharntke, T., Tylianakis, J. M., Rand, T. A., Didham, R. K., Fahrig, L., Batáry, P., et al. (2012). Landscape moderation of biodiversity patterns and processes - eight hypotheses. *Biol. Rev.* 87, 661–685. doi: 10.1111/j.1469-185X.2011.00216.x

Vickruck, J. L., Best, L. R., Gavin, M. P., Devries, J. H., and Galpern, P. (2019). Pothole wetlands provide reservoir habitat for native bees in prairie croplands. *Biol. Conserv.* 232, 42–50. doi: 10.1016/j.biocon.2019.01.015

Williams, D. R., Phalan, B., Feniuk, C., Green, R. E., and Balmford, A. (2018). Carbon storage and land-use strategies in agricultural landscapes across three continents. *Curr. Biol.* 28, 2500–2505. doi: 10.1016/j.cub.2018.05.087

Wood, S. N. (2017). *Generalized Additive Models: an Introduction With R*. New York, NY: CRC Press.

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 21 April 2020
doi: 10.3389/fsufs.2020.00038

Social-Ecological Processes and Impacts Affect Individual and Social Well-Being in a Rural Western U.S. Landscape

Amanda L. Bentley Brymer[1,2]*, *David Toledo*[3], *Sheri Spiegal*[4], *Fred Pierson*[2], *Patrick E. Clark*[2] and *J. D. Wulfhorst*[1]

[1] Environmental Science Program, University of Idaho, Moscow, ID, United States, [2] USDA-ARS Northwest Watershed Research Center, Boise, ID, United States, [3] USDA-ARS Northern Great Plains Research Laboratory, Mandan, ND, United States, [4] USDA-ARS Jornada Experimental Range, Las Cruces, NM, United States

To achieve agroecosystem conservation strategies while balancing the needs of people who live and work across rural landscapes, it is critical to understand what people need to improve and sustain their quality of life and well-being. Research that is designed to connect social-ecological dynamics, landscape change, and human impacts to human well-being and ecosystem health is well-suited to inform land management strategies and decision-making for agricultural production policies. We asked livestock producers, public land and resource managers, recreation users, conservationists, and wilderness advocates who live and work among rural communities in southwestern Idaho to describe social-ecological conditions that support and degrade their well-being. Using grounded theory methodology, we analyzed semi-structured interviews to discover meanings of well-being and to understand how people experience changes to their quality of life in an arid rangelands context. Our findings support previous research that suggests well-being is experienced at both individual and community scales, with sense of well-being influenced by ecological, economic, and socio-cultural processes. Specifically, our findings illuminate the role of social interactions as processes that support agroecosystem conditions and functions to the benefit or detriment of human well-being and ecosystem health. Community is not just a geographic territory; it is a process of social interactions through which people build, improve, or damage relationships that support or degrade well-being. By integrating scholarship on social change processes, ecosystem services, and impacts to human well-being, we contribute an integrated framework with a comprehensive set of social-ecological concepts to be used as a common language and synthesis guide for agroecosystem researchers and practitioners. We discuss our findings in the context of the USDA Agricultural Research Service's national network for Long-Term Agroecosystem Research (LTAR). The LTAR network is charged with identifying strategies for sustainable intensification that support agricultural productivity, environmental quality, and rural well-being. Our research sheds

OPEN ACCESS

Edited by:
Mark A. Drummond, United States Geological Survey, United States

Reviewed by:
Benjamin Paul Bryant, Stanford University, United States
Alisher Mirzabaev, Center for Development Research (ZEF), Germany

***Correspondence:**
Amanda L. Bentley Brymer
abentleybrymer@uidaho.edu

Specialty section:
This article was submitted to Land, Livelihoods and Food Security, a section of the journal Frontiers in Sustainable Food Systems

Received: *04 October 2019*
Accepted: *19 March 2020*
Published: *21 April 2020*

Citation:
Bentley Brymer AL, Toledo D, Spiegal S, Pierson F, Clark PE and Wulfhorst JD (2020) Social-Ecological Processes and Impacts Affect Individual and Social Well-Being in a Rural Western U.S. Landscape. Front. Sustain. Food Syst. 4:38. doi: 10.3389/fsufs.2020.00038

light on the functions of agroecosystem stakeholders and rural communities beyond their adoption (or not) of new technologies and management practices. Future assessments of environmental change and impacts must adequately address social processes that, alongside ecological processes, affect well-being for rural communities and landscapes.

Keywords: individual well-being, social well-being, social change processes, ecosystem services, social impacts, agroecosystems, rangelands, rural landscapes

INTRODUCTION

In a globalized food-energy system, rural landscapes comprise space and resources for agricultural production, while also providing *place* and *purpose* for rural communities and people whose livelihoods are directly or indirectly dependent upon healthy, functioning agroecosystems. While global demands for nutritious food and fiber are increasing, agricultural producers and land managers are challenged to promote sustainable, functional, and productive agroecosystems while adapting to stressors and rapid rural landscape change. Recent calls for sustainable intensification focus on agricultural management practices that meet demands while reducing negative impacts to agroecosystems and to rural communities amid multiple environmental stressors (Robertson et al., 2008; Rockström et al., 2017; Spiegal et al., 2018). This emphasis on sustainable food systems represents a paradigm shift from agricultural research that focused primarily on productivity, profitability, and ecosystem health. Now the challenge is to conserve agroecosystems while balancing the needs of people who live and work across rural landscapes.

Emerging research on sustainable intensification, beyond questions of productivity and profitability, is poised to investigate how human well-being changes in response to dynamic social-ecological processes and drivers of land-use change. Recent insights call for frameworks that guide analyses of trade-offs and synergies among ecosystem services and between production and conservation as outcomes of sustainable intensification (Lescourret et al., 2015; Rockström et al., 2017; Spiegal et al., 2018). However, conceptual frameworks commonly employed to guide assessments of interactions and feedbacks among ecosystems and people tend to imply—intentionally or not—that ecosystems provide services to people while people impact ecosystems (Fish, 2011; Reyers et al., 2013). "If we look after the services, the framework implies, well-being will take care of itself," (Fish, 2011, p. 673). Indeed, people are beneficiaries of ecosystem services and often harness ecological processes to co-produce goods that are beneficial to human well-being, such as basic material needs (e.g., agricultural production of food and fiber). However, from a sociological perspective, the formation of benefits that support dimensions of well-being like "social relations" and "freedom of choice and action" does not emerge from ecological processes (Fish, 2011). While prominent ecosystem service frameworks acknowledge the influential relationships between institutions, anthropogenic drivers of change, and human well-being (e.g., Díaz et al., 2015, 2018), there remains a need to integrate social theory, concepts, and processes to better frame our investigations and to improve our interpretation and understanding of individuals' and communities' needs and responses to environmental changes. Vadrot et al. (2016, 2018) call for contributions from the social sciences and humanities to improve our understanding of social-ecological systems and how they relate to human well-being, human rights, equity, and justice. Specifically, there is room to improve our characterization of human well-being and the way ecosystems, people, and communities co-produce sustainable food systems (Huntsinger and Oviedo, 2014). Furthermore, there is a clear need for concepts and theory from disciplines within the social sciences, like rural sociology and social-psychology. Such scholarship will help frame and explain people and communities as functional parts of agroecosystems—not just as reactors to institutions and ecological processes, or impactors to nature.

To improve our collective understanding, we take a three-part approach. First, we review established frameworks for conceptualizing social and ecological processes, impacts, and human well-being that exist in related but separate literatures. Next, we present a qualitative, interdisciplinary methodology that integrates perspectives from ecology, agricultural productivity, and rural sociology to identify and clarify relationships among social-ecological processes and impacts to ecosystem health and individual and social well-being. We close with a discussion about research designed to assess feedbacks, trade-offs, and synergies among management practices, environmental changes, and well-being, and how such research is critical for agroecosystem management and conservation that sustains rural livelihoods and food security.

Human Well-Being and Existing Frameworks

The Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005) has numerous direct applications for questions related to environmental change and human well-being. The MA was designed, in part, to conceptualize and define well-being as a multivariate state comprising: (1) basic material for a good life, (2) health, (3) security, (4) social relations, and (5) freedom of choice and action. Similarly, quality of life is defined as a value-based, context-dependent state of material and non-material components that enable the achievement of a fulfilled human life (Díaz et al., 2015, 2018). The MA emphasized the need to think about ecosystem services in relation to human well-being to improve outcomes of planning for sustainable development. Arguably, an individual requires basic material needs, health, security, and the freedom and social relations to obtain and sustain those needs. Yet, society is more than an aggregation of individuals; it is communicative and interactive.

TABLE 1 | Dimensions of social well-being adapted from Wilkinson (1991).

Distributive justice	Recognition of the *fact* of human equality, actions to remove inequalities
Open communication	Efficient channels for sharing information, communicative interactions that are honest, complete, and authentic
Tolerance	Normative standard of respect; acceptance of differences and similarities
Collective action	Building social relationships, working together in pursuit of common interests
Communion	Willful entry into celebration of community, joyful response to relationships and shared purpose, purposive involvement

What does it mean for groups, communities, or nations of people who communicate and interact to also live well? Turning to scholarship in rural sociology, social well-being is a concept made distinct from, and dependent upon, individual well-being to denote human pursuits of social interactions and solidarity (Wilkinson, 1991). An interactional theory of community and social well-being explains the role of community as an organization of social life through which social interactions enable the expression and achievement of common needs, and as a process for mobilization toward solving common problems and improving common life (Wilkinson, 1991), as well as what it means for people to feel connected to the places where they live (Brehm et al., 2009). Social well-being through social interaction, consequently, is not an aggregate of individual sustenance needs. As human bodies we need food and shelter; as social beings we also interact to express and negotiate dynamic interests and goals. Wilkinson (1991) characterized dimensions of social well-being in rural North America (**Table 1**).

These dimensions of social well-being represent the proposition that the health of a rural community and thus its inhabitants depend (in part) on social interactions that—beyond meeting sustenance needs—support conditions that enable social cohesion and local solidarity (Wilkinson, 1991). In other words, community is more than an ecological unit or territory, and it is more than a network of people living in proximity and exchanging resources to meet daily needs. Taking the interactional view, community is a process of dynamic social interactions that support individual, social, and ecological well-being (Wilkinson, 1991). Moreover, understanding community as an interactional field of collective processes elaborates a framework to analyze a variety of "capitals" that may or may not exist within communities. Individual and social well-being comprise key components of social capital within the community capitals framework (Emery and Flora, 2006) and illustrate how individual experiences such as stress or anxiety can manifest as impacts to social well-being.

While recent contributions to systems scholarship conceptualize humans as co-producers and beneficiaries of ecosystem services, there is room to improve our understanding and characterization of social processes and their contributions to people and communities, like social interactions that generate and support social relations and social cohesion. Next, we review scholarship on social impacts and project appraisals that provide conceptualizations of social change processes and insights about the role of people and communities in pursuit of their own well-being.

Social Change Processes, Human Impacts, and Existing Frameworks

Social change processes are series of actions that trigger changes in the conditions and functions of a social system and may or may not cause social impacts, while a social impact is a physical or perceptual change experienced by humans as individuals and at higher levels of aggregation (Vanclay, 2002). To improve the assessment of proposed resource management projects and their impacts to people living and working in a project area, Slootweg et al. (2001) presented a function evaluation framework that is useful for identifying potential pathways of change from the project intervention to impacts. For example, severe restrictions of the sustainable use of biodiversity and ecosystems might result in the sell-off of agricultural lands, followed by rural-to-urban migration, leading to rural population decline and changed demographic structures (Vanclay, 2002). The impacts of such demographic changes might produce a negative experience for both migrating and remaining rural residents as community cohesion is disrupted, thus reducing social connections (Wulfhorst et al., 2006) and opportunities for bartering and market exchange (Toledo et al., 2018). **Table 2** elaborates categories and examples of social change processes and potential impacts conceptualized by Vanclay (2002).

The function evaluation framework (Slootweg et al., 2001) is useful for identifying pathways of influence between a social change process and impacts to human well-being. The conceptualization of social change processes and their impacts (Vanclay, 2002) is useful for categorizing and describing social processes and drivers of change to human conditions that may be experienced as positive or negative impacts to well-being for an individual, family, or community.

Here, we respond to the call for a common approach to understand how well-being can be achieved and sustained while pursuing the conservation and sustainable use of biodiversity and ecosystems (Díaz et al., 2015, 2018). In our view, a common approach to assess processes and impacts that affect human well-being does not preclude quantitative indicators, but initially, if not primarily requires a qualitative approach to data collection and analyses (Sayre, 2004). Unlike quantitative research, qualitative approaches to data collection and analysis are typically inductive processes through which researchers iterate between literature review, data collection, and analysis to discover meanings and derive explanations about the data (Locke, 2002; Patton, 2015). An investigation that explores meanings of well-being in a local context enables findings on the conceptualized relationships between ecosystems, people, and their communities to be grounded in the data, thus offering salient variables and dynamics for consideration in future agroecosystem research. Turning to a case study of rural landscapes in southwestern Idaho, USA, we analyzed semi-structured interviews with rangeland agroecosystem stakeholders to discover meanings of well-being and to understand how the

TABLE 2 | Adapted from Vanclay (2002).

SOCIAL CHANGE PROCESSES	
Category	**Examples**
Demographic processes	In-migration, out-migration, presence of newcomers, rural-to-urban migration, urban-to-rural migration
Economic processes	Conversion and diversification of economic activities, impoverishment, inflation, concentration of economic activity
Geographical processes	Conversion and diversification of land use, urban sprawl, urbanization, enhanced transportation, and rural accessibility
Institutional and Legal processes	Institutional globalization and centralization, decentralization, privatization
Emancipatory and empowerment processes	Democratization, marginalization and exclusion, capacity building
Sociocultural processes	Segregation, social disintegration, cultural differentiation
SOCIAL IMPACTS	
Category	**Example indicators**
Health and social well-being impacts	Mental health: feelings of stress, anxiety, apathy, and other psycho-social factors; nutrition: quality and adequacy of food supply; perceived health and fertility; death of self, family member, or community: loss of human and social capital
Live-ability impacts	Aesthetic quality: vistas, infrastructure; leisure and recreation opportunities; perceived and actual adequacy of housing, built infrastructure, social infrastructure; perceived and actual personal safety: crime and violence
Economic and material well-being impacts	Workload; standard of living: ability to obtain goods and services; opportunities for individual employment, income; economic prosperity and resilience of a community; property values; debt
Cultural impacts	Moral rules, beliefs, values; language; integrity: ability of a culture to persist
Family and community impacts	Family structure: stability; obligations to living elders and/or ancestors; sense of belonging; place attachment; perceived and actual community cohesion; perceived and actual inequity
Institutional, legal, political, and equity impacts	viability and integrity: capacity and competence of government agencies to perform tasks; access to legal procedures; participation in decision-making
Gender relations impacts	Gendered division of labor; equity of educational achievement

conditions that support their individual and social well-being are impacted by social-ecological processes and dynamic rural landscape change.

LTAR Network in The Great Basin—Exploring Framework Applications

Rangelands in the western United States comprise deserts, grasslands, shrublands, savannah, and a complex mosaic of municipalities, rural communities, privately-owned property, publicly administered lands, and multiple scales of governance. To guide research and impact assessments for conservation planning, rangelands have been conceptualized and analyzed as coupled human-natural systems, social-ecological systems, or complex adaptive systems (Walker and Janssen, 2002; Havstad et al., 2007; Brunson, 2012, 2014; Li and Li, 2012; Petursdottir et al., 2013). The sustainability or resilience of such systems can be explained by the co-evolutionary relationship between ecosystems, people, and management practices at multiple scales across time (Berkes and Folke, 1998). In the case of U.S. rangelands, relationships between ecosystems and people are commonly characterized by the biodiversity and ecological processes that contribute to cultural heritage, recreation, agricultural production, and livelihoods (e.g., forage production that supports grazing and livestock production; open space for recreational use). The sustainability of rangeland agroecosystems in the western U.S. is complicated by climate-vegetation dynamics (Bradley et al., 2016; Larson et al., 2017), wildland-urban interface dynamics (Liffmann et al., 2000; Li et al., 2019), local economy-community dynamics (Lewin et al., 2019), and federal grazing use policies on public lands that are perceived as inflexible in the face of dynamic environmental change (Brunson and Huntsinger, 2008; Kleinman et al., 2018).

Across the U.S., other agroecosystems in addition to rangelands face similar stressors and rapid transitions while attempting to satisfy demands for agricultural commodities, environmental quality, and rural prosperity and well-being. In a coordinated effort to assess and contrast conventional and aspirational strategies for sustainable intensification, the Agricultural Research Service (USDA-ARS) and collaborators are engaged in a Long-Term-Agroecosystem Research (LTAR) network with the goal of building a nationally-relevant knowledge base to ensure the sustainable provision of agricultural products and ecosystem services from agroecosystems, while acknowledging current and future effects of environmental trends, public policies, and emerging technologies (Bryant et al., 2015). By implementing multi- and inter-disciplinary investigations of agricultural production practices at 18 sites across the U.S. (**Figure 1**), the LTAR network provides a critical opportunity to understand rural prosperity and well-being in relation to ecosystem services and social change processes. We also expect that more empirical investigation of well-being will enable the overdue articulation of constructs like ’rural prosperity’ in need of better definition.

Our analysis uses the Great Basin ARS site within the LTAR network to investigate human well-being and the social-ecological processes and impacts that affect it. Our analysis examines the experiences of ex-urban and rural residents who live and work across a mosaic of public and privately-owned

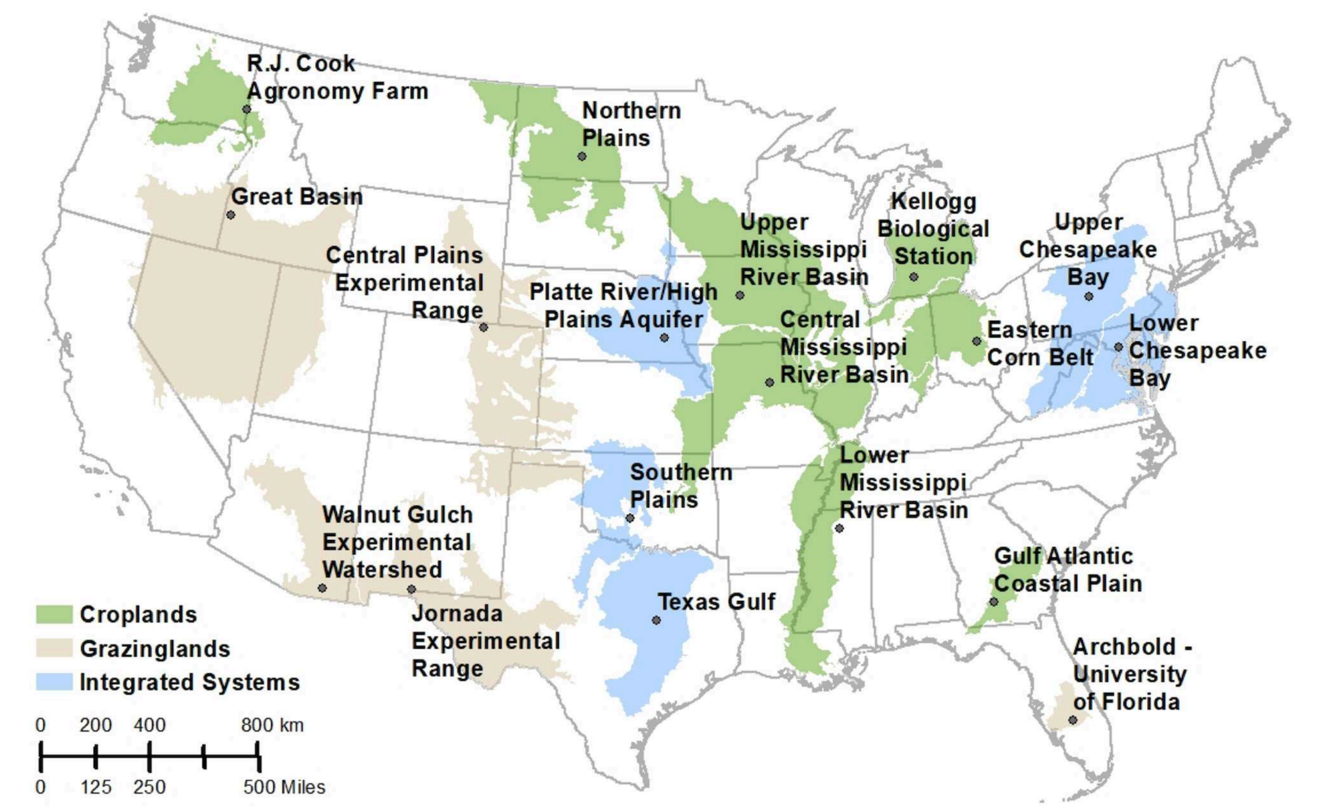


FIGURE 1 | Map of continental United States with USDA-ARS sites participating in the Long-Term Agroecosystems (LTAR) Network. Source: USDA-ARS.

rangelands in the Owyhee Mountains area of southwestern Idaho, USA. This region is part of the historic and current range of the Greater Sage-Grouse (*Centrocercus urophasianus*), a candidate species for endangered listing at the time of our research from 2013-2014. At the same time, public lands grazing allotments were up for permit renewal in Owyhee County, Idaho. As the federal agency responsible for administrating these public grazing allotments, the Bureau of Land Management (BLM) decides whether to renew a livestock producer's permit to use an allotment. Given the multiple use mandate for the BLM, its land management decisions impact multiple stakeholder groups, and the agency is often litigated. Concerns about federal regulations ranged from impacts on agricultural practices, livelihoods, and economic activity to impacts on recreational use of public spaces. This southwestern Idaho case of complex social-ecological dynamics provides a rich context in which to explore meanings and experiences of human well-being on a landscape with multiple land uses that include agricultural production and recreation among others.

METHODS

Data Collection

The sampling frame included people who depend on public lands for livelihoods (e.g., livestock producers, agency scientists, land managers), people whose livelihoods are related to public lands (e.g., attorneys, academics, county leadership) as well as those who engage in non-livelihood activities on public lands (e.g., non-governmental groups, hunters and other recreationists). Thirty three prospective interviewees were identified by key informants through snowball sampling, contacted via email, and asked to participate in one semi-structured interview in-person or by phone. Interviews were conducted between August 2013 and September 2014 with 29 people who live and work in the Owyhees and the metropolitan area surrounding Boise, Idaho, USA. The average interview length was 55 min. We followed ethical guidelines for working with people as research participants, and the University of Idaho Institutional Review Board approved our project #12-357.

Data Analysis

The purposes of this study were to understand how people living and working in a rangeland agroecosystem define their own well-being and to identify perceived drivers of change to well-being. We used a constructivist grounded theory building approach to analyze our data (Locke, 2002; Charmaz, 2006), following three key steps. First, using semi-structured interviews (**Supplementary Material**) and field notes, we coded with open and axial coding to denote interviewees' meanings of well-being, perceived social-ecological conditions that support well-being, and social-ecological drivers of change to the well-being of rangelands and rural communities in southwestern Idaho. We revised our codes while working through the data and the literature (Locke, 2002) and while

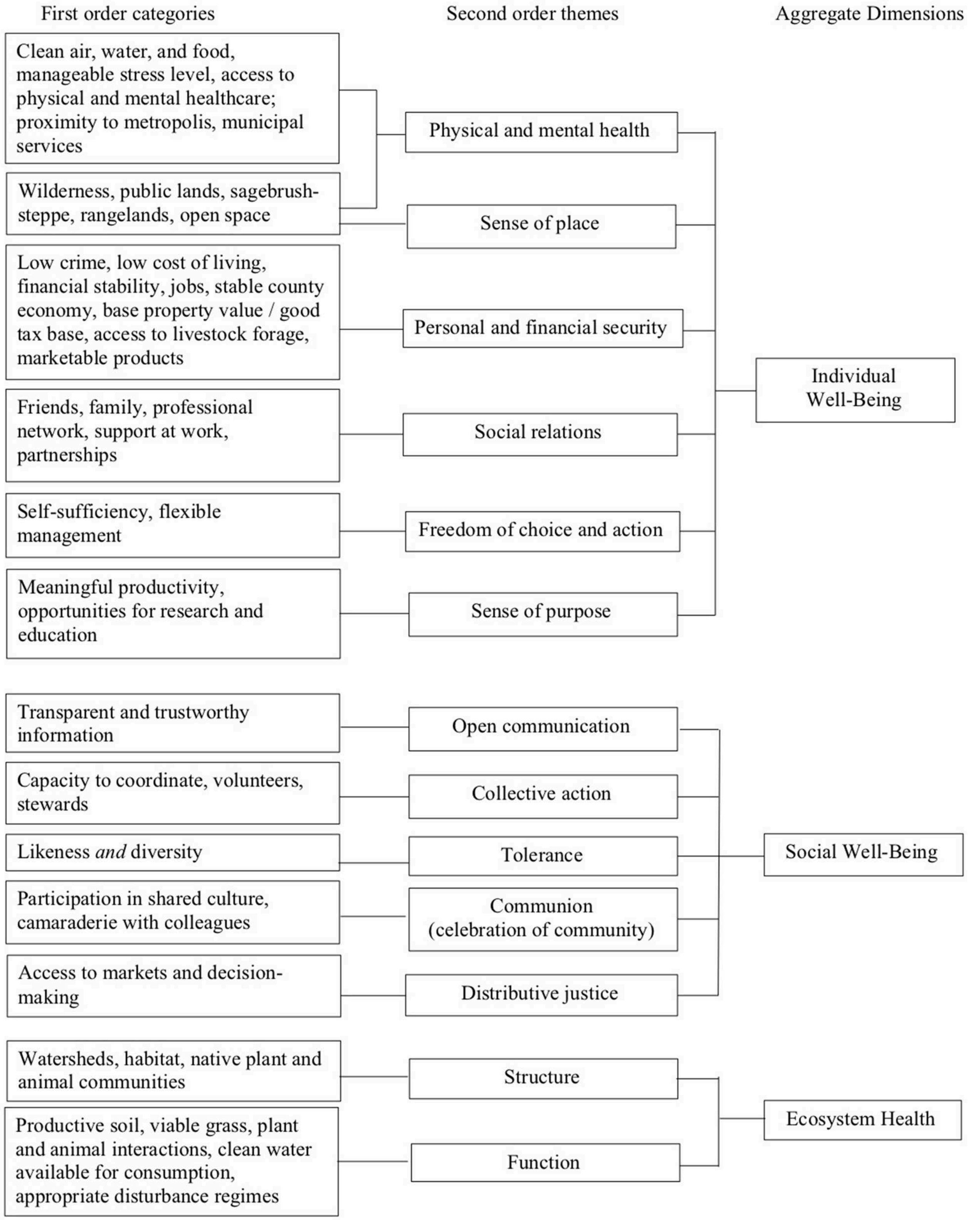


FIGURE 2 | Data Structure formed by iterative analysis to identify conditions and dimensions of human well-being and ecosystem health.

comparing data across participants throughout the analysis (Charmaz, 2006). With this analytical technique we identified first-order categories of social and ecological conditions. We then deduced second-order themes that labeled commonalities among first-order codes, continuing to compare concepts in the data and the literature, which subsequently enabled us to convert the second-order themes to aggregate dimensions (Locke, 2002). In this way, we iteratively examined the data and literature to determine conditions, dimensions, and scales of well-being. **Figure 2** outlines the first order categories, second order themes, and aggregate dimensions that represent the reported ecological and social conditions for individual and social well-being.

We repeated these analytical techniques to categorize interviewees' perceived drivers of change to well-being. **Figure 3** outlines the first order categories, second order themes, and aggregate dimensions that represent dynamic social change processes, including communal processes that are communicative and interactive.

We repeated these analytical techniques once more to categorize the positive and negative changes that interviewees experienced or perceived to result from social-ecological processes and change. **Figure 4** outlines the first order categories, second order themes, and aggregate dimensions that represent ecological and human impacts.

FINDINGS

Altogether, this iterative process of coding for categories and themes while comparing across interviews and previously

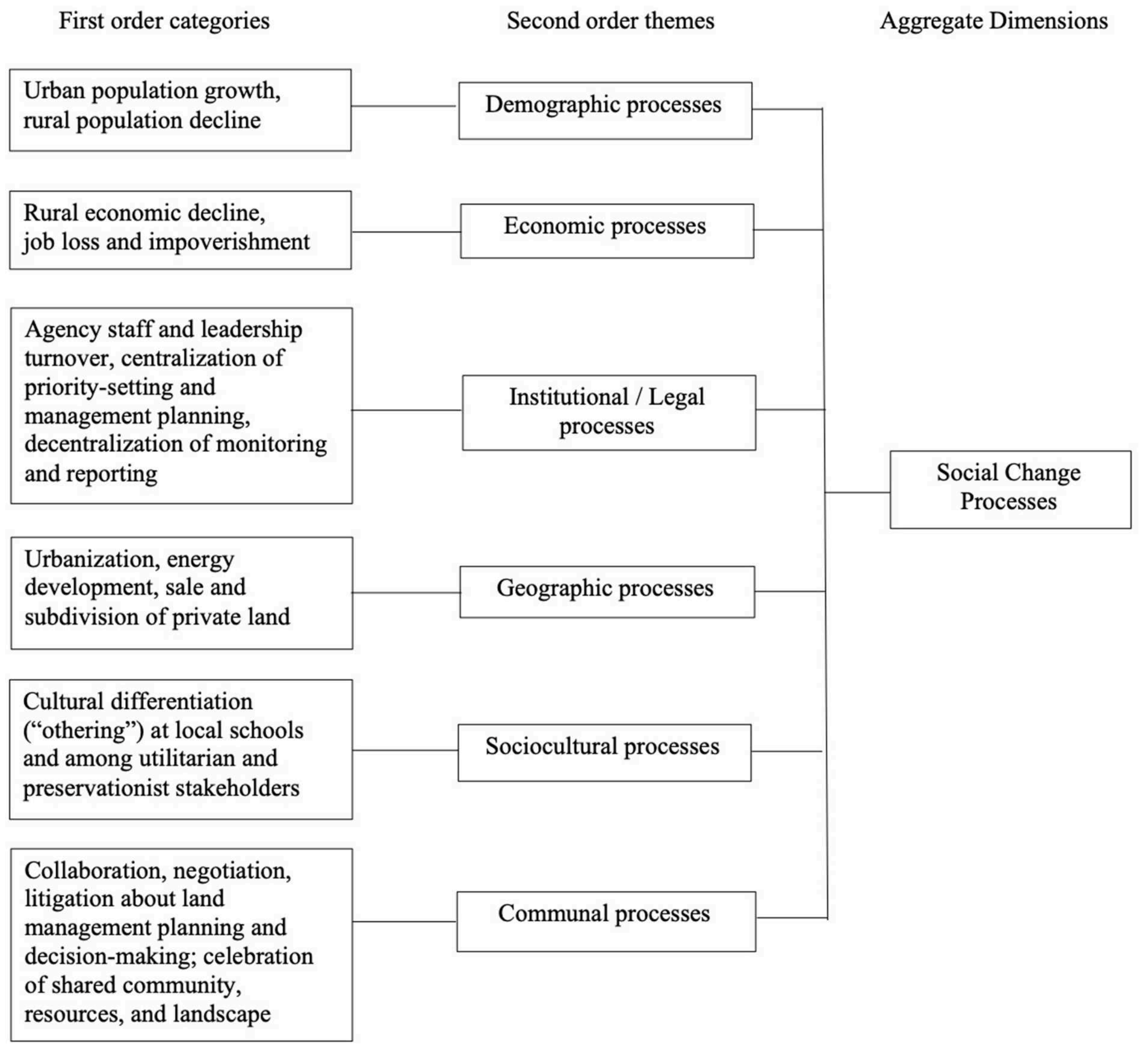


FIGURE 3 | Data structure formed by iterative analysis to identify social change processes.

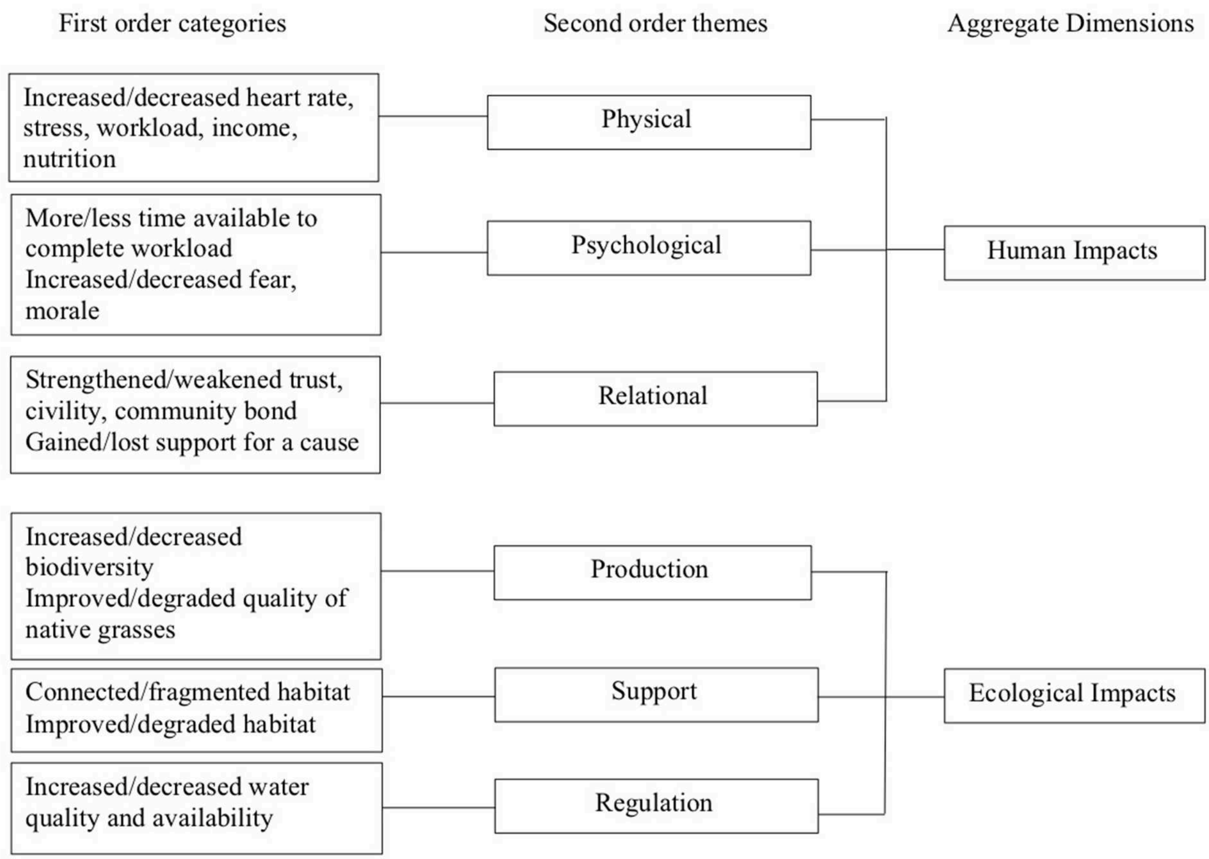


FIGURE 4 | Data Structure formed by iterative analysis to identify positive and negative change that was perceived and tangibly experienced by people and observed in a rangeland agroecosystem.

established literature revealed six aggregate dimensions that represent social-ecological processes and impacts affecting human well-being. In particular, our findings illuminate the functions of communal processes with impacts to physical, psychological, and relational conditions, thus affecting individual and social well-being. **Figure 5** illustrates the pathways between processes, impacts, and well-being by integrating our findings with a function evaluation heuristic that has demonstrated utility for assessment of social change processes and subsequent impacts to well-being (Slootweg et al., 2001). Our interviews revealed important dimensions of well-being that, while recognized in different domains of scholarship (Wilkinson, 1991; Millennium Ecosystem Assessment, 2005; Díaz et al., 2015, 2018), have yet to be integrated into standard frameworks for agroecosystem assessment and analysis. Additionally, our interviews revealed functions of communal processes of change that, coupled with previously conceptualized social processes and impacts (Vanclay, 2002), illuminate mechanisms through which people build or damage social relations upon which they partly depend for both individual and social well-being.

We first present findings on dimensions and scales of human well-being and ecosystem health. We then present findings on social change processes and perceived impacts to human well-being and ecosystem health.

Individual Well-Being

Early in our analysis it became clear that the range of conditions described as supportive of well-being aligned with previously established dimensions of well-being, including physical and mental health, personal and financial security, social relations, and freedom of choice and action (**Figure 2**). For example, a sense of well-being can be derived from the freedom and capacity to provide for oneself and one's family, as explained by a state agency range specialist: "It is a very comforting feeling to know that we can grow our own meat, grow our own produce, and almost be self-sufficient." It was common to hear descriptions of ecological and economic conditions in tight connection as people use their knowledge and skills to cultivate agroecosystems in the co-production of basic material needs that support physical and mental health and personal and financial security. Additionally, descriptions of social relations in terms of economic activity and exchange emerged as a pattern among interviewee responses. For example:

> …(Well-being is) dependent on your schools, your local businesses, your markets, where you sell your products… say if you grow hay, you've got extra hay. You sell to other farmers or ranchers, or if you've got corn you can sell, or purchase from those. There is a lot of ties amongst those, even like in

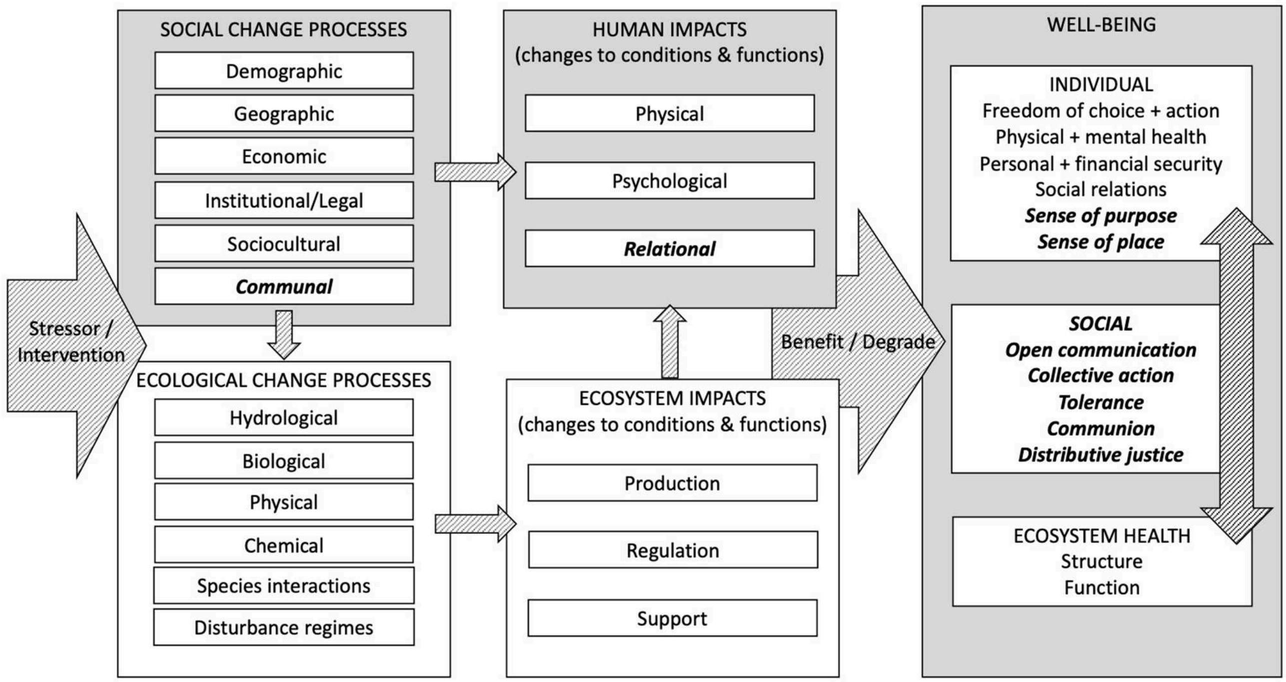


FIGURE 5 | Integrated framework that synthesizes social change processes, human impacts, and well-being (gray boxes) with impact pathway heuristics (Slootweg et al., 2001), ecological processes, and ecosystem services (i.e., nature's contributions to well-being, Díaz et al., 2018) to show how human well-being and ecosystem health benefit from (or are degraded by) both nature's and society's contributons. Concepts in bold italics are key dimensions grounded in our data and analysis that expand established frameworks commonly used to illustrate human-nature relationships and pathways of influence.

> my business. Even though it's small, there is certain crops that I need that I don't grow that I can purchase locally… and that connection economically contributes to the social understanding of how everybody is dependent on each other."—*Rancher and public lands permittee, Owhyee County*

Interestingly, interviewees reported conditions for their personal well-being that do not fit neatly within previously established dimensions of human well-being (Millennium Ecosystem Assessment, 2005). In our case, open landscapes and public lands managed for multiple uses were described as exemplary of social and ecological conditions requisite for well-being. In particular, a pattern of appreciation and attachment to open space emerged among many of our study participants. For example:

> "I want my kids to know the way of life that I value—I want them to have a good work ethic. If they choose not to live in a small town, that's fine. At the least, I want them to have a choice and a sense of freedom in the openness—not when everything's paved."—*Field office assistant manager, federal land management agency*

This attachment to open space was often coupled with a sense of belonging in publicly managed and accessible land and waterways. For example:

> "So, my well-being, as far as what I think, is the outdoors and kind of a balance of being able to see all types of wildlife, not just having to go to a national park. I think you can balance things out with agricultural, with the cattle industry, and be able to (sic) everybody live together type of thing. And big horn sheep is kind of something that everybody wants to see. If you float a river, if you go to Hills Canyon, if you float the middle fork of the Salmon, I mean when big horn sheep were up above the river I mean everybody stops and you know, takes the pictures and stuff… it's just beautiful country, you see a lot of wildlife, and beautiful clean water."—*Retired outdoor guide*

In addition to describing a sense of place as a condition of well-being, several interviewees described meaningful work and productivity as similarly important. For example:

> "…just putting in a good day's work and feeling like I'd actually completed something toward either conserving—enhancing conditions for wildlife in the area that I work at. In general terms, that's kind of what I look for (to feel fulfilled). It doesn't happen every day, but that's what I strive for."—*Biologist, federal land management agency*

Distinct from descriptions of basic material needs, health, and social relations, this desire for meaningful productivity aligns with the idea that a good quality of life can be assessed in terms of freedom of choice and action and a sense of purpose in action. A sense of purpose relates to one's own agency and action within a dynamic agroecosystem, and a sense of place relates to one's attachment and belonging to that system and the landscapes and communities within it.

Social Well-Being

As we continued to code and deduce second-order themes, we found that the previously established dimensions of well-being (Millennium Ecosystem Assessment, 2005) did not account for all well-being dimensions described by interviewees (**Figure 2**). These additional conditions for well-being reflect the communicative nature of society and the needs of people to interact toward common goals. The desire for access, transparency, and complete exchanges of trustworthy information emerged as a pattern among interviewees representing diverse stakeholder groups that are often in conflict over public lands planning and decision-making. For example:

> "On paper, this sounds really good because it's a collaborative group – these folks that were sworn enemies before, and they came up with this plan. They agreed on this plan and got the (agency) to implement it. Now me, I thought it really stunk. For one, their plan did not consider our land management plans that we had worked on with the whole public using all our resources issues. Another issue is, even though they said they were representing everybody, they weren't." – *Environmental advocate, non-governmental organization*

This quote exemplifies the common sentiment among our interviewees that barriers to open communication, like "back-door deal-making," tend to degrade well-being. Additionally, we found that access to decision-making and fairness of outcomes matter to a sense of procedural justice for many in this case (Lauer et al., 2017). Related, access to markets for exchange of basic and fundamental services (e.g., health care) and social needs (e.g., attention) align with notions of distributive justice emphasized as a core dimension of social well-being (Wilkinson, 1991). When interviewees reflected on decision-making processes for which they did have access and representation, some described a willingness to trade-off efficient channels of information sharing for the positive outcomes of committed, though time-consuming, open communication. For example:

> "… you've got to keep an open mind. I think in the long run cause - the meeting before - I was kind of down after it. I said, it don't look like it's going to go very good. But then after yesterday I could see a lot of positive movement. I think that's the way. It just takes time. You can't do it in four meetings.…it takes time…to get everything looked at, analyzed so that everybody is comfortable that yes, we did discuss it. And I may not agree, but I can see the reasons why maybe some of these things should be." – *Rancher & public lands permittee, Owyhee County*

This quote also exemplifies two additional dimensions of social well-being: collective action, through a long-term commitment to a collaborative process, and tolerance, with appreciation for diverse viewpoints. Given the public lands context of our study region, several interviewees described rich, nuanced experiences with collaborative processes. Some result in collective action toward a shared goal, while others may result in more conflict and polarization. For example:

> "…it's about finding that balance and what you can live with, too. Because some of our projects- it's a tolerance level. You think, I could have gone into it thinking I will never ever be a part of that. Then when it gets explained to you, you say okay, I have this much tolerance level to that. I can do that because it's important to your group." - *Rancher and public lands permittee, Owhyee County*

Our analysis also revealed interviewees' perceptions of opportunities to celebrate their shared culture as well as camaraderie with colleagues as necessary conditions for their sense of well-being. In particular, a sense of place in the back country of rural landscapes strongly aligned with ideas about communal celebration of shared space:

> "…look at the broad community of Opening Day. Fishermen who rallied together to plant along the Boise River. It's why many people decide to raise their families here. It is not only a shared family value, it's a great part of our community. You can have some cases with an explicit spiritual aspect as well when you talk about wilderness values, or if you talk about family camping events. One of the aspects we try to create out there, one of the values is solitude, but another integral value is community…going out with a bunch of friends elk hunting… a mountain biking trip down to the Middle Fork River." – *Conservation specialist, non-governmental organization*

This celebration of shared values among community members represents communion as a dimension of social well-being (Wilkinson, 1991). It also represents the distinction between rural landscapes as a functional space for recreation with benefits to individuals' physical and mental health, and rural landscapes as a functional space for celebration of shared values with benefits to community health and well-being. For example:

> "There's a gentleman that sold his ranch and moved to all private ground because he just got tired of always wondering what was going to go on. I just talked to him a couple weeks ago, he's an older gentleman, and he says there's a lot of things he misses about public lands ranching and there's some things that he doesn't. One of the things he misses is community bonding. So that was an interesting concept. Actually, ran into him at a funeral service and we had this conversation. Cause I always ask him, do you miss running on BLM, and he says there's certain things he misses about it, and he misses the people." - *Rancher and public lands permittee, Owhyee County*

By integrating social dimensions with individual dimensions, human well-being is conceptualized in a way that comprehensively represents how people experience changes to conditions and functions of their social-ecological system at multiple scales. Our analysis also revealed ecological conditions of rural landscapes that were perceived by interviewees to be necessary to sustain both human well-being and ecosystem health.

Ecosystem Health

Our analysis revealed descriptions of ecological conditions perceived to be necessary for ecological well-being (Wilkinson, 1991), i.e., ecosystem health in terms of structure (e.g., watersheds, habitat, native plant communities) and function (e.g., productive soil, water filtration). While such conditions were recognized for their importance to ecosystem health, several biophysical and ecological conditions were also described as beneficial for human well-being, including clean air, clean water, and open space for recreation:

> "[To be well, we need] livability, sustainability, clean air, clean drinking water. When you turn on your tap water, when you open your window, do you have a nice quality of life? The public land is more about the source of our drinking water, and also the wildlife and recreation opportunities, and also the sustainable management of our public lands. So... whether you like to hunt or fish out there, to not only preserve those opportunities, but that they improve over time." – *Conservation specialist, non-governmental organization*

Additionally, interviewees who self-identified as agricultural producers commonly described their dependence on ecological functions like forage production to support their livelihoods:

> "..we're very dependent on (the) ecological... whether it's climate, weather, it impacts the grasses that we depend on to graze my cattle..." *Rancher and public lands permittee, Owyhee County*

As exemplified by the quotes above, our analysis revealed common perceptions of production, regulation, and support functions as ecosystem services that, when impacted by drivers of environmental and landscape change, result in altered delivery of benefits (or detriments) to people and communities. Similarly, our analysis demonstrated the salience of social change processes and their beneficial/detrimental influence to individual, social, and ecological well-being in this western U.S. rural landscape context.

Social Change Processes

As we began to code interview transcripts for perceived social change processes, it became clear that processes of demographic change are a salient issue in the urban-rural interface surrounding Boise, Idaho (**Figure 3**). For example:

> "Recreation can be an issue, but it's generally in a smaller impact area, just primarily along the [Boise] Front, just because of the population explosion in the Treasure Valley. You talk to people who have been out here for a long time and you look at some graphs about OHV off-road vehicle use and things like that, and they've just gone off the charts in the last 20 years. So that has definitely been an issue that we're trying address both ecosystem-wise and wildlife-wise, and for the safety and well-being of the public, who are our customers, basically." – *Rangeland specialist, federal resource management agency*

This quote highlights the perception among interviewees that an increasing population is perceived to lead to an increase in recreational use of nearby public lands with potentially negative impacts to the physical health and safety of "the public," as well as potentially negative impacts to wildlife and ecosystem health. This finding also reveals a tension between the negative impacts perceived for some community members and the potentially positive impacts to physical, psychological, and relational conditions for those who engage in recreational activities like off-road vehicle use.

In addition to urban population growth, social change processes like urban sprawl, rural economic decline, and leadership turnover within public land management agencies were perceived to trigger human and ecosystem impacts. These and other reported phenomena represent geographic, economic, and institutional change processes, respectively, and align with previously conceptualized social change processes and their influence on biophysical change with subsequent impacts to ecosystems and to people (Vanclay, 2002). For example:

> "...economically, a lot of them [producers] aren't surviving, so they're selling off their ranches...when they sell them they turn into – a lot of them – suburban neighborhoods or those little subdivided ratchetted. So, there goes your open space because, granted, they're ranches, they're privately owned, but wildlife still uses those areas. So, then you're losing that, too, and it's a pretty rapid rate." – *Biologist, federal land management agency*

Our findings also reveal perceptions of vilification or "othering" as sociocultural change processes with negative impacts to well-being, as well as positive impacts from overcoming "othering." For example:

> "...we sat down with people who had been on the other side of lawsuits....we would meet twice a week the first year we met twice a week for all day, but we had lunch together every time. We didn't go our separate ways. We all went to the same place, and we had to sit by somebody we didn't know, and all of a sudden, your kids are reading the same books – right then Harry Potter was just out and so we get to talking, and I'll tell you the first day I had to sit by a guy...and we'd been on litigation and I was thinking, I don't want to sit by him. We got to talking, and all of the sudden we started talking about...camping things you can go look for different things to do... So, when you start having those conversations, all of the sudden you're not an organization. You're a person that has a wife, and kids, and feelings." - *Rancher and public lands permittee, Owyhee County*

This quote represents a process through which the perceived negative impacts of cultural differentiation were mitigated through communal processes. While the sociocultural change process aligns with previously established conceptualizations (Vanclay, 2002), the functions of these conversations as social interactions in relation to open communication and tolerance are important to distinguish and clarify. We noticed a common perception about public lands collaboration as a process that can trigger physical, psychological, and relational impacts. For some interviewees, these impacts are beneficial to well-being; to others, they are detrimental to freedom of choice and action and personal and financial security. For example:

> "I've heard of instances... where you go through the collaboration process, you feel that you've made compromises, addressed issues, and then once the decision's been issued, you still get appealed from those people who have been sitting across from you at the compromise table, the collaboration table. So that, I would think, would be extremely frustrating... that you spend all this time in collaboration, and then, because of these polarized viewpoints, if they still have not gotten exactly everything they want, then they are still going to appeal regardless." – *Biologist, federal land management agency*

As described above, coordinated activity for conservation planning and decision-making represent social interactions through communal processes that may or may not support collective action and open communication in relation to social well-being. There was a sense among a few interviewees that the tone of social interactions is important with respect to its impact on psychological and relational conditions. For example:

> "What bothers me is sometimes the lack of civility in public conversations about things... In our national conversation, which does then affect some of the other values we cherish, say ecological values, the lack of civility means we're not moving toward resolution. We're fighting, and that bothers me." – *Public lands researcher, academic institution*

As we focused our analysis on communal processes, we found examples of human conditions and functions that were perceived and felt to change as a result of collaborative or litigious experiences. For example:

> "We end up doing a lot of this reactive work because of litigation, then we end up not being able to get out to the field... It affects your work satisfaction... Some people... handle stress differently than others. I've seen some people about near have a meltdown." – *Public affairs specialist, federal land management agency*

While this quote exemplifies perceived negative impacts to mental health from participating in litigation, our last example quote illustrates the view shared among most, though not all of our interviewees regarding social interactions through a communal process like collaboration for public lands management:

> "Well you end up everybody having a voice, and then trying to figure out a solution. And it's a success when you do solve the problem, and everybody feels they were a part of it. And that gives kind of a personal attachment to the whole management even if you're just a small part of it." – *Range specialist, federal natural resource management agency*

These findings inform our thinking about how dynamic social processes drive changes to ecosystem and human conditions with beneficial and/or detrimental effects to ecosystem health and individual and social well-being.

DISCUSSION

Using a grounded theory methodology to explore meanings of well-being in a case of democratically governed public rangelands in the western U.S., our findings present evidence in support of a multi-scale characterization of human well-being. We asked people what they need to be well and what social-ecological processes threaten or support those needs. Our analysis revealed a similar theme among public rangelands stakeholders in southwestern Idaho, regardless of stakeholder group affiliation or self-reported identity: open space, clean air, clean water, productive soil, and resilient plants and animals are critical conditions of rangeland agroecosystems that contribute to human well-being. These findings align with scholarship that defines and categorizes ecosystem services (de Groot et al., 2002) and with scholarship on western U.S. rangelands-specific ecosystem services (Havstad et al., 2007; Brunson, 2014; Huntsinger and Oviedo, 2014; Bentley Brymer et al., 2016).

Individual and Social Well-Being

In addition to perceptions of ecosystem services and conditions necessary for ecosystem health, our interviewees described desirable conditions relating to several dimensions of individual well-being, including physical and mental health, personal and financial security, social relations, freedom of choice and action, sense of place, and sense of purpose. We highlight the latter two dimensions because, for rural people and communities in our case, sense of place and purpose are tightly wrapped up in resource-based livelihoods and management of agroecosystems and rural landscapes. Sense of place theory and tools for analysis provide fruitful directions for elaborating well-being and for understanding individuals' and communities' capacity to adapt to environmental change (Masterson et al., 2017). While a sense of place relates to one's attachment, meanings, and belonging to that system and the landscapes and communities within it, a sense of purpose relates to one's own agency and action within a place. Such meaningful productivity aligns with the idea that a good quality of life can be assessed in terms of freedom of choice and action and a sense of purpose in action. By integrating sense of place and sense of purpose, research that is designed to address questions about rural landscape change and impacts to quality of life will benefit from a more comprehensive conceptualization of individual well-being.

Our interviewees also described conditions relating to several dimensions of social well-being, including tolerance, open communication, distributive justice, collective action, and communion (i.e., celebration of community, purposive involvement). Interestingly, the interactional nature of social well-being was illuminated by rich descriptions of the positive and negative impacts to a community's opportunities for collective action, usually driven by social interactions through communal processes like collaborative resource management and public lands litigation. Collective, or coordinated actions play a critical role in building social capital. Social capital is important because it can provide access to other forms of capital such as financial capital, and it improves a community's ability to cope with change by providing access to innovative solutions and by mitigating perceived risk (Adger, 2003; Olsson et al., 2004; Wagner and Fernandez-Gimenez, 2008). As members of a community mobilize for collective action, social capital can be considered an interactional platform that supports improvements to well-being, especially during times of crisis (Woolcock and Narayan, 2000). In contrast, a breakdown in social capital and collective action has been shown to lead to ecological degradation and unregulated use of resources (Mallon, 1983; Wagner and Fernandez-Gimenez, 2008). As biophysical and social conditions change, landowners and managers must learn how to continually adapt to new conditions to sustain their well-being. Critically, learning is contingent on the development of trust among collaborators, suggesting the need for social processes that develop relationships and trust over time (Wilmer et al., 2018).

We gained nuanced descriptions of social relations as indicators of well-being, and their conceptualization in relation to collective action as a dimension of social well-being warrants further discussion. Social relations represent the connections or ties that a person has to others in her community for mutual benefit and cooperation (Coleman, 1990) and are important factors for individuals' physical and mental health (Thoits, 2011). Additionally, the strength or weakness of social ties influence the power dynamics within a community (Agrawal and Gibson, 2001). In the context of environmental governance and agroecosystem management, such power dynamics often manifest in decision-making settings that are increasingly designed as deliberative processes through which citizens can debate their concerns, improve their dialogue, and learn (Daniels and Walker, 1996, 2001). The outcome of such interactions for planning and decision-making are often driven by participants who have access, standing, and power in the process (Senecah, 2004; Dawson et al., 2017). Those who do not have access, standing, or power in the process are not fully *well* because they are cut-off from the mechanism through which they might influence their own well-being. With respect to social well-being, social relations are the building blocks of collective actions that build trust and social capital. In other words, collective action depends upon the strength of social relations. Our analysis also revealed processes that impact social relations and other dimensions of well-being; in particular, the role of communal processes is elaborated.

Social Interactions Through Communal Processes

Our findings align with scholarship on community as a process of social interactions that weave a social fabric comprising connectivity, cohesion, and cooperative opportunity with other people (e.g., Wilkinson, 1991; Wulfhorst et al., 2006; Toledo et al., 2018). The health of a rural community and its inhabitants depend (in part) on social interactions that - beyond meeting sustenance needs - support conditions that enable community cohesion and local solidarity (Wilkinson, 1991). In other words, community is more than an ecological unit or territory, and it is more than a network of people living in proximity and exchanging resources to meet daily needs. Taking the interactional view, community is a process of dynamic social interactions that support individual, social, and ecological well-being (Wilkinson, 1991). The nature and function of such communal processes appears to be distinct from economic, sociocultural, and other social change processes that have been conceptualized as impactful to a person's physical and psychological conditions (Vanclay, 2002). For example, while economic change processes represent shifts in local industry activity and opportunity that may impact an individual's employment, and while sociocultural change processes represent differentiation or concentration of culture and identity that may impact opportunities for communion (i.e., celebration of shared culture), communal processes represent the development or disruption of relationships, shared purpose, and community. Interviewee descriptions of collaborative and litigious interactions illuminate the influence of such communal processes on basic material needs, mental health, and open communication, to the benefit or degradation of human well-being and ecosystem health. Interestingly, even the fear of an adversarial interaction such as fighting in court over public lands management can indirectly impact ecosystem health. In the context of agroecosystems, a breakdown in communication and community may block the implementation of a new grazing or cropland management practice designed to balance and sustain productivity, ecosystem health, and rural well-being. In other words, social processes directly impact people and indirectly impact ecosystems (Slootweg et al., 2001). Therefore, agroecosystem research that aims to identify management practices that support rural well-being must adequately address the social change processes - including communal processes - that impact it.

Our findings illustrate the pathways of influence between social change processes, impacts to physical, psychological, and relational conditions and functions, and perceived benefit or degradation to dimensions of well-being. While methods for assessing impacts to local economies and social structures resulting from changes to public lands management practices in rangeland agroecosystems have been reviewed (Bentley Brymer et al., 2018) and implemented (Lewin et al., 2019), findings presented here highlight communal processes as a potentially new concept for social-ecological impact assessment.

IMPLICATIONS AND CONCLUSION

As a newly formed network with a goal of maintaining productive landscapes, long-term environmental stewardship, and well-being, the LTAR network can learn from these findings. As the network evolves, there needs to be a clear understanding of what conditions of well-being are meaningful to partners and stakeholders within and across LTAR sites. Existing LTAR efforts to define, support, and achieve "rural prosperity" (see Kleinman et al., 2018) can reconcile with our finding that human well-being is experienced at individual and community scales. Also, there is a need to understand pathways to achieve and sustain a good quality of life for rural communities in different agroecosystems - including the role of communal processes -

while sustaining ecosystem services in the co-production of food and fiber.

For instance, LTAR network scientists recently developed a conceptual model to represent regional-scale agroecosystems in terms of interactions among agriculture, the environment, the economy, and society, and used that model to synthesize the multiple dimensions of the LTAR Common Experiment across 18 network sites (Spiegal et al., 2018, **Figure 3**). The model centers on agricultural producers and their decision-making about selecting an agricultural production system suitable for a given agricultural region. In the model, feedback loops mediated by profitability, environmental effects, societal factors, and policy can reinforce "business as usual" or motivate producers to adopt an alternative production system. Comparing outcomes of the widespread adoption of alternative production systems is at the heart of the LTAR Common Experiment, and the explicit integration of communal processes and social well-being into current thinking - and into network conceptual models such as the one used to synthesize the LTAR Common Experiment - will help LTAR to implement the Common Experiment in a way that effectively addresses current and future challenges of coupled human and natural systems in agricultural regions.

As human agency and social dynamics are considered alongside ecological dynamics and ecosystem services, future research will be guided toward more effective interdisciplinary integration. Our research sheds light on the role of agroecosystem stakeholders and rural communities beyond their adoption of new technologies and management practices. Furthermore, we recognize that interdisciplinary approaches to human dimensions of agroecosystem research are more than a means to understand (barriers to) adoption and ecosystem impacts. Our findings illuminate human well-being beyond dimensions of health and financial security and across individual and community scales. By utilizing this expanded conceptual framework to guide interdisciplinary integration, LTAR collaborators will be better equipped to identify, describe, and understand social-ecological dynamics as directly impactful to rural communities and their well-being, and thus to the sustainable intensification and conservation of agroecosystems. Beyond LTAR, future assessments of human-nature relationships and environmental change will more adequately address social change processes and impacts that - along with ecosystem services – contribute to human well-being and to sustainable food systems.

DATA AVAILABILITY STATEMENT

The qualitative data published here are protected by human assurances protocol at the University of Idaho and are not currently available. Please contact the corresponding author with inquires.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by University of Idaho Institutional Review Board. The participants provided their written informed consent to participate in this study. Written informed consent was obtained from the individual(s) for the publication of any potentially identifiable images or data included in this article.

AUTHOR CONTRIBUTIONS

AB, DT, SS, FP, PC, and JW contributed to the conceptualization and drafting of this manuscript. AB and JW designed the interview protocol, AB collected the data, and AB, JW, and DT analyzed the data.

FUNDING

We thank the National Science Foundation IGERT program (Award0903479) for financial support in the early development of this research. The Northwest Watershed Research Center provided funds for open access publication fees through the Great Basin Annual Grass agreement with the University of Idaho, Fund 222989.

ACKNOWLEDGMENTS

This research was a contribution from the Long-Term Agroecosystem Research (LTAR) network. LTAR is supported by the United States Department of Agriculture.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.00038/full#supplementary-material

REFERENCES

Adger, W. (2003). Social capital, collective action, and adaptation to climate change. *Econo. Geograph.* 79, 387–404. doi: 10.1111/j.1944-8287.2003.tb00220.x

Agrawal, A., and Gibson, C. C. (Eds.). (2001). *Communities and the Environment: Ethnicity, Gender, and the State in Community-Based Conservation*. Piscataway, NY: Rutgers University Press.

Bentley Brymer, A. L., Holbrook, J., Niemeyer, R., Suazo, A., Wulfhorst, J. D., Vierling, K. B., et al. (2016). A social-ecological impact assessment for public lands management: application of a conceptual and methodological framework. *Ecol. Soc.* 21:9. doi: 10.5751/ES-08569-210309

Bentley Brymer, A. L., Taylor, D. T., Wulfhorst, J. D., Torell, L. A., Rimbey, N. R., and Tanaka, J. A. (2018). Economic and social impact assessment of ranching on public lands: a guide to concepts, methods, and applications. *J. Rangeland App.* 4, 1–16.

Berkes, F., and Folke, C. (Eds.). (1998). *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience*. Cambridge: Cambridge University Press.

Bradley, B. A., Curtis, C. A., and Chambers, J. C. (2016). "Bromus response to climate and projected changes with climate change," in *Exotic Brome-Grasses in Arid and Semiarid Ecosystems of the Western US, Springer Series on Environmental Management,* eds M. J. Germino, et al. (Cham: Springer International Publishing), 257–274.

Brehm, J. M., Eisenhauer, B. W., and Krannich, R. S. (2009). Dimensions of community attachment and their relationship to well-being in the amenity-rich rural west. *Rural Sociol.* 69, 405–429. doi: 10.1526/0036011041730545

Brunson, M. W., and Huntsinger, L. (2008). Ranching as a conservation strategy: can old ranchers save the new West? *Rangeland Ecol. Manag.* 61, 137–147. doi: 10.2111/07-063.1

Brunson, W. M. (2012). The elusive promise of social-ecological approaches to rangeland management. *Rangeland Ecol. Manag.* 65, 632–637. doi: 10.2111/REM-D-11-00117.1

Brunson, W. M. (2014). Unwanted no more: land use, ecosystem services, and opportunities for resilience in human-influenced shrublands. *Rangelands* 36, 5–11. doi: 10.2111/RANGELANDS-D-13-00064.1

Bryant, R. B., Hapeman, C. J., Havstad, K. M., Heilman, P., Kleinman, J., McCarty, G. W., et al. (2015). *Long-Term Agroecosystem Research Network Shared Research Strategy.* Available online at: https://www.ars.usda.gov/ARSUserFiles/np211/LTAR_SRS_Final_29Sept2015.pdf (accessed March 7, 2019).

Charmaz, K. (2006). *Constructing Grounded Theory: A Practical Guide Through Qualitative Analysis*. Thousand Oaks, CA: Sage.

Coleman, J. (1990). *Foundations of Social Theory*. Cambridge, MA: Harvard University Press.

Daniels, S. E., and Walker, G. B. (1996). Collaborative learning: improving public deliberation in ecosystem-based management. *Environ. Impact Assess. Rev.* 16, 71–102. doi: 10.1016/0195-9255(96)00003-0

Daniels, S. E., and Walker, G. B. (2001). *Working Through Environmental Policy Conflict: The Collaborative Learning Approach*. Westport, CT: Praeger.

Dawson, N. M., Grogan, K., Martin, A., Mertz, O., Pasgaard, M., and Rasmussen, L. V. (2017). Environmental justice research shows the importance of social feedbacks in ecosystem service trade-offs. *Ecol. Soc.* 22:12. doi: 10.5751/ES-09481-220312

de Groot, R. S., Wilson, M. A., and Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* 41, 393–408. doi: 10.1016/S0921-8009(02)00089-7

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., et al. (2015). The IPBES conceptual framework—connecting nature and people. *Curr. Opin. Environ. Sustain.* 14, 1–16. doi: 10.1016/j.cosust.2014.11.002

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., et al. (2018). Assessing nature's contributions to people. *Science* 359, 270–272. doi: 10.1126/science.aap8826

Emery, M., and Flora, C. (2006). Spiraling-up: mapping community transformation with community capitals framework. *J. Commun. Dev. Soc.* 37, 19–35. doi: 10.1080/15575330609490152

Fish, D. R. (2011). Environmental decision making and an ecosystems approach: some challenges from the perspective of social science. *Prog. Phys. Geograp.* 35, 671–680. doi: 10.1177/0309133311420941

Havstad, K. M., Peters, D. P., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., et al. (2007). Ecological services to and from rangelands of the United States. *Ecol. Econ.* 64, 261–268. doi: 10.1016/j.ecolecon.2007.08.005

Huntsinger, L., and Oviedo, J. (2014). Ecosystem services are social-ecological services in a traditional pastoral system: the case of California's mediterranean rangelands *Ecol. Soc.* 19:8. doi: 10.5751/ES-06143-190108

Kleinman, P. J. A., Spiegal, S., Rigby, J. R., Goslee, S., Baker, J., Bestelmeyer, B. T., et al. (2018). Advancing sustainable intensification of U.S. agriculture through long-term research. *J. Environ. Q.* 47, 1412–1425. doi: 10.2134/jeq2018.05.0171

Larson, C. D., Lehnhoff, E. A., and Rew, L. J. (2017). A warmer and drier climate in the northern sagebrush biome does not promote cheatgrass invasion or change in its response to fire. *Oecologia* 185, 736–774. doi: 10.1007/s00442-017-3976-3

Lauer, F. I., Metcalf, A. L., Metcalf, E. C., and Mohr, J. J. (2017). Public engagement in social ecological systems management: an application of social justice theory. *Soc. Nat. Res.* 31, 4–20. doi: 10.1080/08941920.2017.1364456

Lescourret, F., Magda, D., Richard, G., Adam-Blondon, A. F., Bardy, M., Baudry, J., et al. (2015). A social–ecological approach to managing multiple agroecosystem services. *Curr. Opin. Environ. Sustain.* 14, 68–75 doi: 10.1016/j.cosust.2015.04.001

Lewin, P. A., Wulfhorst, J. D., Rimbey, N. R., and Jensen, K. S. (2019). Implications of declining animal unit permits on public land: an integrated social and economic impact analysis. *West. Econ. Forum.* 17, 82–93. doi: 10.22004/ag.econ.28731

Li, W., and Li, Y. (2012). Managing rangeland as a complex system: How government interventions decouple social systems from ecological systems. *Ecol. Soc.* 17:9. doi: 10.5751/ES-04531-170109

Li, Y., Westlund, H., and Liu, Y. (2019). Why some rural areas decline while some others not: an overview of rural evolution in the world. *J. Rural Stud.* 68, 135–143. doi: 10.1016/j.jrurstud.2019.03.003

Liffmann, R. H., Huntsinger, L., and Forero, C L. (2000). To ranch or not to ranch: home on the, urban range? *J. Range Manag.* 53, 362–370. doi: 10.2307/4003745

Locke, K. (2002). "The grounded theory approach to qualitative research," in *Measuring and Analyzing Behavior in Organizations: Advances in Measurement and Data Analysis,* eds F. Drasgow and N. Schmitt (San Francisco, CA: Jossey-Bass), 17–43.

Mallon, F. (1983). *The Defense of Community in Peru's Central Highlands: Peasant Struggle and Capitalist Transition 1860 – 1940.* Princeton, NJ: Princeton University Press. doi: 10.1515/9781400856046

Masterson, V. A., Stedman, R. C., Enqvist, J., Tengo, M., Giusti, M., Wahl, D., et al. (2017). The contribution of sense of place to social-ecological systems research: a review and research agenda. *Ecol. Soc.* 22:49. doi: 10.5751/ES-08872-220149

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-Being: Biodiversity Synthesis.* Washington, DC: Island Press.

Olsson, P., Folke, C., and Hahn, T. (2004). Social-ecological transformation for ecosystem management: the development of adaptive co-management of a wetland landscape in southern Sweden. *Ecol. Soc.* 9:2. doi: 10.5751/ES-00683-090402

Patton, Q. M. (2015). *Qualitative Research & Evaluation Methods*, *4th Edn.* Thousand Oaks, CA: Sage Publications.

Petursdottir, T., Arnalds, O., Baker, S. C., Montanarella, L., and Aradottir, A. (2013). A social–ecological system approach to analyze stakeholders' interactions within a large-scale rangeland restoration program. *Ecol. Soc.* 18:29. doi: 10.5751/ES-05399-180229

Reyers, B., Biggs, R., Cumming, G. S., Elmqvist, T., Hejnowicz, A. P., and Polasky, S. (2013). Getting the measure of ecosystem services: a social-ecological approach. *Front. Ecol. Environ.* 11, 268–273. doi: 10.1890/120144

Robertson, G. P., Allen, V. G., Boody, G., Boose, E. R., Creamer, N. G., Drinkwater, L. E., et al. (2008). Long-term agricultural research: a research, education, extension imperative. *BioScience* 58, 640–645. doi: 10.1641/B580711

Rockström, J., Williams, J., Daily, G., Noble, A., Matthews, N., Gordon, L., et al. (2017). Sustainable intensification of agriculture for human prosperity and global sustainability. *Ambio* 46, 4–17. doi: 10.1007/s13280-016-0793-6

Sayre, F. N. (2004). The need for qualitative research to understand ranch management. *J. Range Manag.* 57, 668–674. doi: 10.2307/4004026

Senecah, L. S. (2004). "The trinity of voice: the role of practical theory in planning and evaluating the effectiveness of environmental participatory processes," in *Communication and Public Participation in Environmental Decision Making,* eds S. Depoe, J. W. Delicath, and F. M. Aepli Eisenbeer (Albany, NY: SUNY Press), 13–33.

Slootweg, R., Vanclay, F., and van Schooten, M. (2001). Function evaluation as a framework for the integration of social and environmental impact assessment. *Impact Assess. Project Apprai.* 19, 19–28. doi: 10.3152/147154601781767186

Spiegal, S., Bestelmeyer, B. T., Archer, D. W., Augustine, D. J., Boughton, E. H., Boughton, R. K., et al. (2018). Evaluating strategies for sustainable intensification of US agriculture through the long-term agroecosystem research network. *Environ. Res. Lett.* 13:034031. doi: 10.1088/1748-9326/aaa779

Thoits, P. A. (2011). Mechanisms linking social ties and support to physical and mental health. *J. Health Soc. Behav.* 52, 145–161. doi: 10.1177/0022146510395592

Toledo, D., Briceño, T., and Ospina, G. (2018). Ecosystem service valuation framework applied to a legal case in the Anchicaya region of Colombia. *Ecosyst. Serv.* 29, 352–359. doi: 10.1016/j.ecoser.2017.02.022

Vadrot, A. B., Jetzkowitz, J., and Stringer, L. C. (2016). Social sciences: IPBES disciplinary gaps still gaping. *Nature* 530:160. doi: 10.1038/530160b

Vadrot, A. B., Rankovic, A., Lapeyre, R., Aubert, P. M., and Laurans, Y. (2018). Why are social sciences and humanities needed in the works of IPBES? A systematic review of the literature. *Innovat. Eur. J. Soc. Sci. Res.* 31, S78–S100. doi: 10.1080/13511610.2018.1443799

Vanclay, F. (2002). Conceptualising social impacts. *Environ. Impact Assess. Rev.* 22, 183–211. doi: 10.1016/S0195-9255(01)00105-6

Wagner, C. L., and Fernandez-Gimenez, M. E. (2008). Does community-based collaborative resource management increase social capital? *Soci. Nat. Res.* 21, 324–344. doi: 10.1080/08941920701864344

Walker, B. H., and Janssen, M. A. (2002). Rangelands, pastoralists and governments: interlinked systems of people and nature. *Philos. Transact. R. Soc. B* 357, 719–725. doi: 10.1098/rstb.2001.0984

Wilkinson, P. K. (1991). *The Community in Rural America*. Westport, CT: Greenwood Publishing Group.

Wilmer, H., Derner, J. D., Fernandez-Gimenez, M. E., Briske, D. D., Augustine, D. J., Porensky, L. M., et al. (2018). Collaborative adaptive rangeland management fosters management-science partnerships. *Rangeland Ecol. Manag.* 71, 646–657. doi: 10.1016/j.rama.2017.07.008

Woolcock, M., and Narayan, D. (2000). Social capital: implications for development theory, research, and policy. *World Bank Res. Obser.* 15, 225–249. doi: 10.1093/wbro/15.2.225

Wulfhorst, J. D., Rimbey, N., and Darden, T. (2006). Sharing the rangelands, competing for sense of place. *Am. Behav. Sci.* 50:166. doi: 10.1177/0002764206290631

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 27 May 2020
doi: 10.3389/fsufs.2020.00075

Drivers of Agricultural Diversity in the Contiguous United States

*Sarah C. Goslee**

USDA ARS Pasture Systems and Watershed Management Research Unit, University Park, PA, United States

The spatial heterogeneity of vegetation types on a landscape has been linked to multiple ecosystem functions, including habitat for wildlife and pollinators, water cycling, human aesthetic values, and nutrient cycling. Although agricultural land uses are sometimes combined into a single unit when quantifying landscape heterogeneity, diverse cropping systems are a valuable alternative to near-monocultural croplands and contribute more strongly to ecosystem service provision, including services such as pest regulation and carbon sequestration that are of direct interest for agriculture. The USDA Cropland Data Layer was used to characterize crop diversity across the contiguous US for 2008–2018. Percentage of each crop type, along with non-crop uses such as forest and development, were calculated for each 4 km PRISM climate data grid cell. To better understand the drivers of crop diversity, Random Forest modeling was used to assess the importance of climate, soils, and irrigation for patterns of crop effective richness for the contiguous United States, stratified by USDA Land Resource Region. The models explained 57–89% of the variation in maximum crop diversity, with irrigation being by far the most important explanatory variable in regions where it was employed. The drivers of change from 2008 to 2018 were less clear. Random Forest models explained only 20–60% of the change in agricultural diversity over the 11-year period; both soil and climate properties were important, with no clear dominant drivers. Potential crop effective richness was greater than actual across the entire region studied, but substantial increases would require irrigation. Major changes in agricultural systems and infrastructure may be necessary to increase agricultural diversity at large spatial extents, and declining availability of water for irrigation could threaten the agricultural systems that are now most diverse.

Keywords: agricultural diversity, Cropland Data Layer, ecosystem services, irrigation, Random Forest

OPEN ACCESS

Edited by:
Fardausi Akhter,
Agriculture and Agri-Food Canada (AAFC), Canada

Reviewed by:
Sotirios Archontoulis,
Iowa State University, United States
Mauro Centritto,
Institute for Sustainable Plant Protection (CNR), Italy

Correspondence:
Sarah C. Goslee
sarah.goslee@usda.gov

Specialty section:
This article was submitted to Agroecology and Ecosystem Services, a section of the journal Frontiers in Sustainable Food Systems

Received: *12 September 2019*
Accepted: *28 April 2020*
Published: *27 May 2020*

Citation:
Goslee SC (2020) Drivers of Agricultural Diversity in the Contiguous United States. Front. Sustain. Food Syst. 4:75. doi: 10.3389/fsufs.2020.00075

INTRODUCTION

Multifunctional managed landscapes are necessary for the maintenance of the ecosystem services that sustain both humans and their environment. Not all landscapes are equal: some uses and configurations are more effective at maintaining ecosystem services than others. Large areas of a single land cover, core areas, provide habitat for species that cannot be found in more disturbed areas. Increasing human population requires increasing agricultural productivity without compromising the ecosystem; such developments will require detailed understanding of the positive and negative ecological consequences of agricultural management decisions (Bommarco et al., 2013). Manipulation of crop diversity within existing agricultural areas may offer a pathway for improving ecosystem service provision in agroecosystems without compromising food security.

Agriculture is a major component of landscapes in the United States, where 35% of the land surface is used for cropland and pasture (USDA, 2019), and fills the crucial ecosystem service of providing food, fiber and fuel. Both total agricultural area and agricultural diversity within that area are determinants of ecosystem service provision. At a global scale, separate factors drive agricultural diversity and agricultural expansion (Martin et al., 2019); the same is likely true at smaller scales.

More diverse agricultural landscapes have been demonstrated to provide environmental benefits (Altieri, 1999). Choice of crop identity and spatial and temporal configuration contribute to agricultural diversity. Spatial diversity improves habitat for birds, wildlife, and pollinators (Jerrentrup et al., 2017). Temporal diversity in the form of multi-species rotations and cover crops has been shown to increase soil carbon storage and improve nutrient cycling (McDaniel et al., 2014; Spawn et al., 2019). Agricultural diversity also benefits the farmer, by buffering unexpected events and potentially reducing revenue variability, and may reduce agrochemical usage, including pesticides and fertilizers (Di Falco and Perrings, 2005).

Globally, crop diversity has decreased with increased reliance on a few dominant commodity crops, even as the total crop richness has increased (Khoury et al., 2014; Martin et al., 2019). About 90% of the calories consumed globally are provided by 20 crop species; this reliance on only a few dominant crops may threaten food security at national and global scales (Khoury et al., 2014; Massawe et al., 2016). In the central U.S., monoculture cropping has increased (Plourde et al., 2013). Production costs, existence of markets, and subsidies and crop insurance programs all contribute to the maintenance of monocultural crops (Bowman and Zilberman, 2013). Contrary to ecological theory, diverse crop portfolios do not lead to revenue stability when high market prices and agricultural subsidies promote monocultures of specific crops (Di Falco and Perrings, 2005; Weigel et al., 2018).

Crop selection is heavily dependent on market prices (Weigel et al., 2018), infrastructure, and landscape history. Nonetheless, agricultural decisions are embedded in a biophysical template which constrains the choices available. Crop-specific requirements for temperature and water availability determine the palette of crop species which may be selected. Irrigation augments water availability, but water and temperature experienced by crops are predominantly functions of climate and soil properties.

Crop selection decisions made at field and farm scales have consequences for ecosystem service provision at those same scales, and also at landscape scales. While some ecosystem services, such as soil erosion and nutrient cycling, are primarily determined by field-scale conditions, others, including wildlife habitat and pollinator suitability, are relevant at larger scales. For instance, bees are known to forage within a 3–5 km radius (Kennedy et al., 2013).

Previous studies of crop diversity have summarized patterns of change over longer timescales using USDA National Census of Agriculture data, but have not attempted to relate those patterns to quantitative environmental variables because the county-scale nature of that dataset makes it difficult to do so (e.g., Aguilar et al., 2015; Hijmans et al., 2016). The USDA Cropland Data Layer (CDL) provides 30-m resolution gridded agricultural land cover data for the contiguous United States for 2008–2018, for major agricultural crops/land covers (Boryan et al., 2011). This dataset offers the richest available information about the spatial distribution of commercially-important crops, and forms the basis for potential assessments of agricultural diversity at a variety of scales.

The objectives of this analysis were to identify the relationships between biophysical variables representing temperature and water availability and the agricultural area and crop diversity across the contiguous United States from 2008 to 2018. Characterizing the drivers of agricultural land use patterns at this scale will enable better regional understanding of potential ecosystem service provision, both under current conditions and given expected changes in climate. Specifically, the maximum diversity and area across the 11-year period were modeled for each USDA Land Resource Region (LRR) using machine learning techniques, as were the changes in diversity and area from 2008–2013 to 2013–2018, and across the entire timespan. The maximum value model for the full US was used to predict the potential agricultural diversity across the region, by identifying areas that are similar and dissimilar to current areas of high diversity.

METHODS

Assembling a complex dataset comprising agricultural land cover, climate, soils properties, and irrigation data at continental scale necessarily requires consideration of trade-offs and arbitrary choices. Data sources are provided at different spatial and temporal scales, and merging them effectively is a complex affair. Care must be taken at all steps to preserve the attributes most relevant to the questions posed, while recognizing that no perfect solutions (yet) exist. For this study, the guiding principle was to aggregate all datasets to the coarsest spatial resolution dataset, the 4 km PRISM daily climate data (PRISM Climate Group, 2018), using procedures appropriate for each type of data.

To facilitate analysis and interpretation, the 20 LRRs, an agriculturally-based regionalization, were used to organize the analyses (**Table 1**; USDA, 2006). Given the diversity of climates and agricultural practices in the US, any implicit assumption that important drivers are consistent across the entire continent is flawed. Dividing the analysis based on predetermined agricultural regions allows the identification of regional patterns in determinants of agricultural diversity and area. However, for characterizing potential diversity, the full contiguous US was modeled and used for prediction. Extrapolating from a model trained on a single region limits the predictions to only those practices currently found within that region. A model trained on the contiguous US makes it possible to extrapolate agricultural potentials across regions, a more interesting analysis.

Agricultural Diversity

The USDA Cropland Data Layer provides spatially-referenced area data for major crops (Boryan et al., 2011). Aggregating the 30 m data to a coarser spatial scale reduced reliance on

TABLE 1 | USDA Land Resource Regions (USDA, 2006).

LRR	Name	Description
A	NW Forest	Northwestern Forest, forage, and Specialty crop region
B	NW Wheat	Northwestern Wheat and range region
C	California	California subtropical fruit, truck, and specialty crop region
D	West Range	Western Range and irrigated region
E	Rocky Mtn	Rocky Mountain range and forest region
F	N G Plains	Northern Great plains spring wheat region
G	W G Plains	Western Great plains range and irrigated region
H	C G Plains	Central Great plains winter wheat and range region
I	SW Plateaus	Southwest Plateaus and plains range and cotton region
J	SW Prairies	Southwestern Prairies cotton and forage region
K	N Lake St	Northern Lake States forest and forage region
L	Lake St	Lake State fruit, truck crop, and dairy region
M	C Grains	Central feed Grains and Livestock region
N	EC Farming	East and Central Farming and forest region
O	MS Delta	Mississippi Delta cotton and feed grains region
P	S Atlantic	South Atlantic and Gulf slope cash crops, Forest, & Livestock region
R	Northeast	Northeastern forage and forest region
S	N Atlantic	Northern Atlantic slope diversified farming region
T	Atlantic	Atlantic and Gulf Coast lowland forest and crop region
U	Florida	Florida subtropical fruit, truck crop, and range region

The regions and dominant crops are shown in ***Figure 1***.

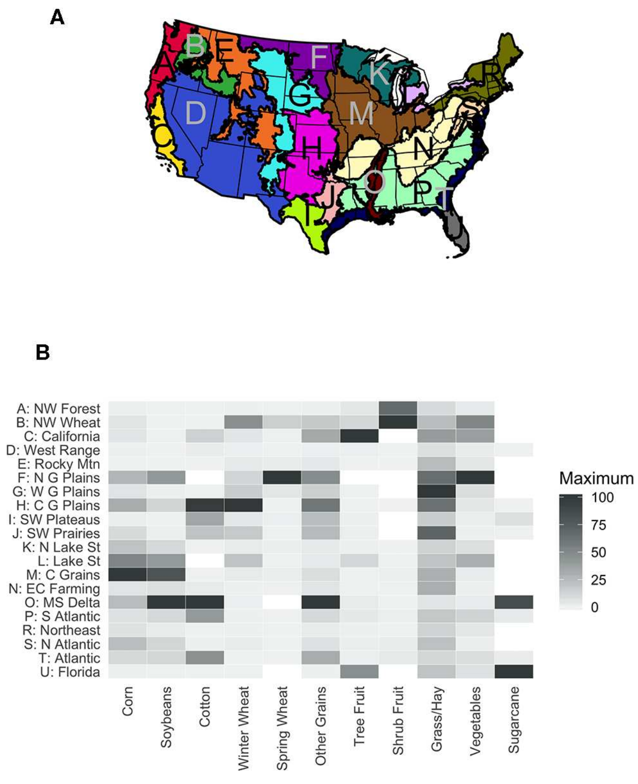


FIGURE 1 | The boundaries of each of the twenty USDA Land Resource Regions **(A)** and the dominant crop area in each from the 2018 Cropland Data Layer **(B)**. The full names of each region are given in **Table 1**.

pixel-scale accuracy, and integrated over crop rotation patterns in space rather than in time. Aggregated proportional areas were used to calculate both the percentage of each grid cell in agricultural land cover, and the percentage of grid cell area in each CDL-identified crop. This dataset was not developed for tracking change over time, and may also be inaccurate in its representation of field boundaries and sub-pixel areas (Reitsma et al., 2016; Lark et al., 2017). Because each year is classified independently of previous years, comparing two time points without considering the intervening years may lead to erroneous results. Aggregating to a 4 km grid cell smoothed over many of these issues, because individual CDL pixels were not being compared, and any uncertainty on pixel boundaries is far smaller than the overall area of interest. It would possibly still be inappropriate to investigate a single grid cell, but this analysis of the CDL provided an effective overview of trends at the continental scale.

For the purposes of this study, double-cropped areas were counted as their own individual crop category. Thus, double-cropped winter wheat and soybeans was counted as distinct from winter wheat alone. The contribution of double-cropping to crop diversity is clearly higher than that of a single crop, although this may not be the most effective adjustment. Pasture/hay was also counted as a crop type since it is an agricultural land cover. No attempt was made to propagate error in crop identification to the diversity measurements.

Defining diversity is a second complex issue, one that has spawned an extensive literature (e.g., Devictor et al., 2010). Taxonomic diversity? Functional diversity? Phylogenetic diversity? Structural diversity? For the purposes of this analysis, agricultural diversity has been defined as the crop effective richness (CER), calculated from the Shannon diversity of percentage area of each CDL-identified crop within a 4 km grid cell. Effective richness increases interpretability of information-theoretic indices such as the Shannon diversity by expressing them as the number of equally-abundant species that would have the same diversity index (Jost, 2006). Effective crop richness retains both richness and evenness components, but expresses the combined value in terms of number of crop species, an intuitively familiar metric. Note however that the lowest possible value of CER when used with Shannon diversity is 1, rather than 0. In this study, use of CER brings the implicit assumption that it is possible to raise at least one crop species everywhere in the region studied, even if there is currently no agriculture conducted there.

Environmental Data

The potential driver variables were chosen to represent temperature and water availability, basic requirements for plant growth. The PRISM daily gridded climate data formed the basis of the dataset (PRISM Climate Group, 2018). Derived variables representing aspects of precipitation and temperature relevant for crops were calculated for the period 2001–2015; selected variables follow Wang et al. (2017), and include a subset of BIOCLIM indices such as precipitation during the warmest quarter, maximum and minimum monthly temperatures, and temperature seasonality (Busby, 1991), and additional agronomic variables such as growing degree days (**Table 2**).

Soils properties were calculated as weighted mean values of the top 100 cm from two separate gridded digital reanalysis products. Texture and chemical properties came from a 100 m US soils dataset (Ramcharan et al., 2018), while soil depth and probability of a restrictive layer were aggregated from the global SoilGrids250 product (Hengl et al., 2017). Spatially-referenced gridded irrigation data were aggregated from the MODIS-derived 2012 gridded 1 km irrigated agriculture layer (Pervez and Brown, 2010; Brown and Pervez, 2014) to produce percentage of grid cell area that was irrigated. This was referenced to the 2012 total agricultural area for each cell to produce percentage of agricultural area that was irrigated; the resulting value was trimmed at 0 and 100% to reduce data inconsistencies.

All datasets were projected into Albers Equal Area and aggregated to the PRISM grid using GRASS GIS [cite]. Fewer than 1,000 grid cells did not have complete data for all climate, soils, CDL, and irrigation layers, resulting in 475,605 grid cells for analysis. All analyses were conducted in R 3.6.0 (R Core Team, 2019). The packages sp 1.3-1 (Pebesma and Bivand, 2005; Bivand et al., 2013) and ggplot2 3.2.0 (Wickham, 2016) were essential for display of results.

TABLE 2 | Climate and soils variables describing aspects of temperature and water availability relevant for crop productivity.

Name	Description	Source
T ff d	Maximum frost-free consecutive days (basis−2.2C; 10th%)	PRISM 2001-2015
T GDD	Growing degree days, basis (C; 10th%)	PRISM 2001–2015
P annual	Annual precipitation (mm; 10th%)	PRISM 2001–2015
T min	BIOCLIM 6: Minimum temperature of coldest month (10th%)	PRISM 2001–2015
P driest	BIOCLIM 17: Precipitation of driest quarter (10th%)	PRISM 2001–2015
P coldest	BIOCLIM 19: Precipitation of coldest quarter (10th%)	PRISM 2001–2015
T wettest	BIOCLIM 8: Mean temperature of wettest quarter (10th%)	PRISM 2001–2015
P dry d	Maximum consecutive days with < 2.5mm of precipitation (10th%)	PRISM 2001–2015
T range d	BIOCLIM 2: Mean diurnal temperature range (10th%)	PRISM 2001–2015
T isotherm	BIOCLIM 3: Isothermality (90th%)	PRISM 2001–2015
T seasonal	BIOCLIM 4: Temperature seasonality (90th%)	PRISM 2001–2015
T max	BIOCLIM 5: Maximum temperature of warmest month (90th%)	PRISM 2001–2015
T range yr	BIOCLIM 7: Temperature annual range (90th%)	PRISM 2001–2015
P wettest	BIOCLIM 13: Precipitation of wettest month (90th%)	PRISM 2001–2015
P seasonal	BIOCLIM 15: Precipitation seasonality (90th%)	PRISM 2001–2015
P warmest	BIOCLIM 18: Precipitation of warmest quarter (90th%)	PRISM 2001–2015
T driest	BIOCLIM 9: Mean temperature of driest quarter (90th%)	PRISM 2001–2015
BD	Soil bulk density	100 m US Soil Grids
Clay	Soil clay content (%)	100 m US Soil Grids
Sand	Soil sand content (%)	100 m US Soil Grids
SOC	Soil organic carbon	100 m US Soil Grids
Restrictive	Probability of restrictive layer (%)	SoilGrids250
Irrig	Irrigated area, 2012 (%)	MODIS
Soil depth	Maximum soil depth (cm)	SoilGrids250

PRISM-derived variables are 10th or 90th percentile over 2001–2015.

Statistical Methods

The core of the analysis is the machine learning method Random Forest (RF), a flexible tree-based regression approach. The fast implementation in the ranger 0.11.2 package (Wright and Ziegler, 2017) provides sophisticated tools for assessing variable importance. While RF models are empirical, rather than process- or theory-based, the shape of the relationship between the dependent variable and each independent variable was assessed using partial dependence plots (pdp 0.7.0 package in R; Greenwell, 2017). Preliminary testing using five-fold cross validation demonstrated that for this dataset, 1,000 trees was adequate, and that increasing the number of variables per tree beyond the default did not produce enough improvement in model fit to justify the large increase in runtime. Impurity, the variance of the regression responses, was used to assess variable importance within the forest, and a permutation test with 100 permutations was used to identify potentially important variables at $p < 0.1$.

Like most regression-based methods, RF models analyze the mean value across all samples at a particular level of an independent variable (or within a node, for tree models such as this). For modeling the potential values of variables where the minimum value is unconstrained (there can be zero agricultural diversity at any point, regardless of site characteristics), quantile RF enables the analysis of maximum values (quantregForest 1.3-7; Meinshausen, 2017). For this study, the 90th quantile was used for prediction. This ability to predict quantiles, not just means, good capability to assess importance of individual predictor variables, and the general familiarity of ecologists with Random Forest models all contributed to the selection of RF rather than another machine learning model for this study.

Three sets of models were constructed. The first was for the maximum value of CER for each grid cell over the 11 years of CDL data available. The maximum value within a grid cell was used, to reduce the effect of interannual fluctuations due to factors other than biophysical potentials. Each LRR was modeled separately. The second set of models described the overall change in CER from 2008 to 2011, again stratified by LRR. Finally, models of maximum CER were constructed for the contiguous US, treating irrigated and rainfed areas separately. The full US was used to parameterize the models to ensure that predicted CER values were not constrained only to those already present in the LRR, but extrapolated across the entire set of possibilities found in the dataset.

RESULTS

The relative abundance of different groups of dominant crops in the 2018 CDL highlights the regional patterns of agriculture in the contiguous US, and sets the stage for more detailed analysis of crop diversity (**Figure 1**). The values are scaled to the maximum percentage abundance for each crop, so for instance the greatest percentage area of corn was planted in the Central Feed Grains region, while the greatest area of cotton production was in the Mississippi Delta and Central Great Plains regions. Agriculture dominates the Great Plains regions, Mississippi Delta, California and the Northwest Wheat LRRs (**Figure 1A**). Large regions of arid rangeland in the Southwest and forest in the northern and eastern US have patchy or no agricultural areas (**Figure 2A**). Irrigation in 2012 was most common in the Mississippi Delta, California, and Western Range regions, as well as portions of the

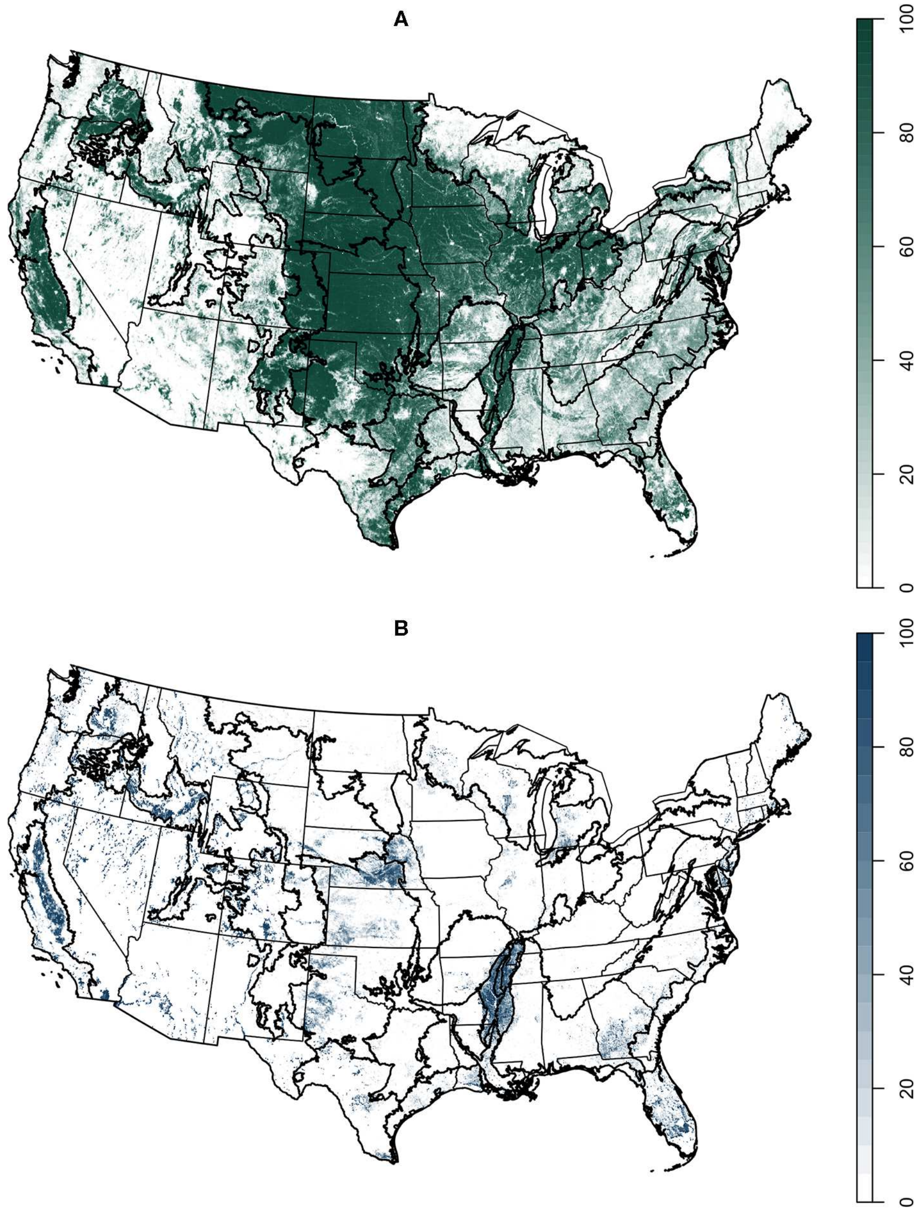


FIGURE 2 | (A) Maximum percentage of agricultural land, 2008–2018. **(B)** Irrigated agricultural area, 2012.

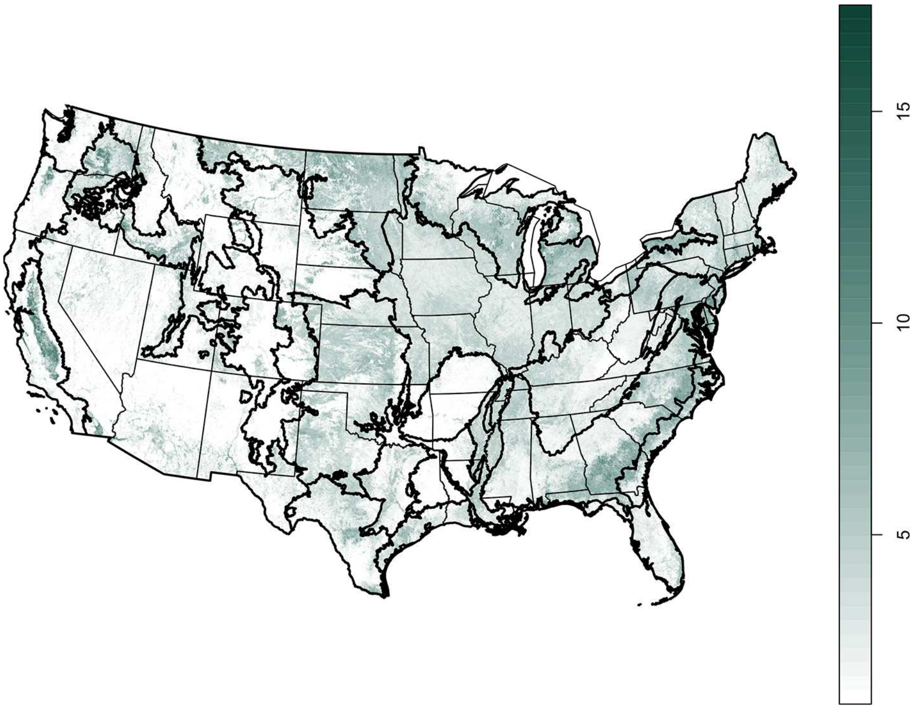


FIGURE 3 | Maximum crop effective richness, 2008–2018.

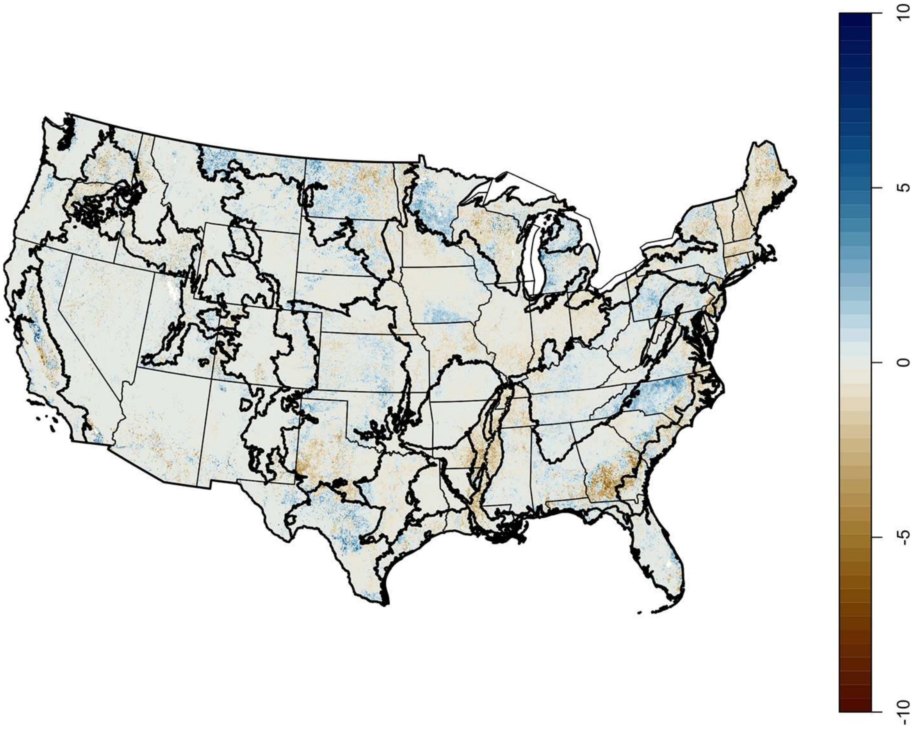


FIGURE 4 | Change in annual crop effective richness, 2008–2018.

Central Great Plains and Florida, and smaller portions of other areas (**Figure 2B**).

The agriculturally-dominated Central Great Plains nonetheless had low agricultural diversity; the northern regions, California, and the southeast had the highest diversity (**Figure 3**). California had the maximum value of CER per grid cell (17.5), while the Lake States had the highest average value (5.3). The Northern Great Plains and the Northern Atlantic Slope also had high average CER; these areas have both extensive and diverse agriculture. Change in CER over the 11 years of data currently available was highly patchy (**Figure 4**). The Mississippi Delta lost the most CER overall, an average of −0.8. The Northeast (−0.3) and the Atlantic (−0.1) LRRs were the only other regions with a negative mean change in CER. The Northern Great Plains and the Southwest Plateaus both increased by an average of 0.4. California had both the greatest losses and gains within individual grid cells, −9.8 and 9.2 species.

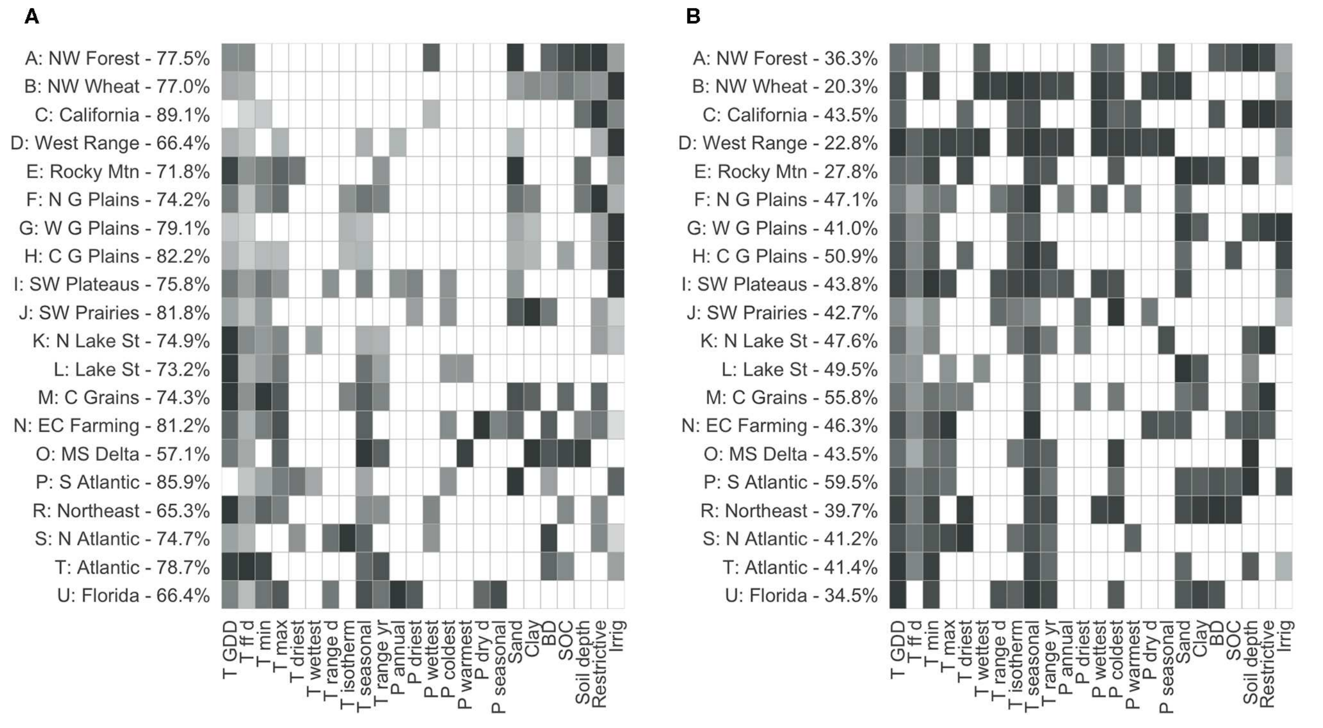


FIGURE 5 | Relative variable importance for the significant variables for each LRR model models of maximum crop effective richness **(A)** and change in crop effective richness **(B)**, 2008–2018. The darkest is the most important for each region; lighter colors are relative to that maximum. Region labels include the variance explained for that model. Color scale is as for **Figure 1B**.

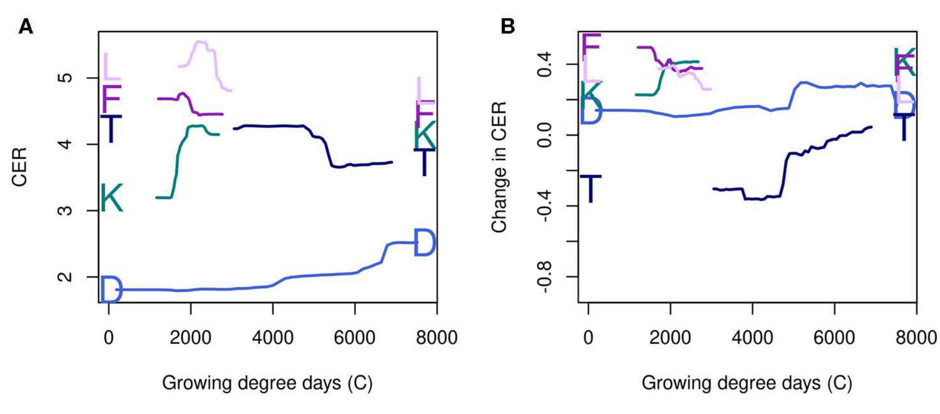


FIGURE 6 | Partial dependence plots for growing degree days for both maximum CER **(A)** and change in CER **(B)**, for several LRRs showing different responses, following the colors and letter code given in **Figure 1** and **Table 1**.

The RF models of CER explained 57.1–89.1% (Mississippi Delta and California, respectively; average 75.7%) of the variance in crop diversity (**Figure 5A**). There was a general tendency for the LRRs with a broad range of CER values to be the best-fit by the models; machine learning techniques like RF perform best across a range of potential predictors, rather than in regions where very little area is in agriculture (West Range), or where almost all of the area is in homogeneous agriculture (Mississippi Delta). Variability in crop diversity was substantially related to temperature and water availability. Growing degree days, length of the growing season, minimum and maximum temperature, and temperature seasonality were the major temperature variables; soil texture, irrigation, and presence of a soil restrictive layer were frequently-important controls on water availability, more so than precipitation directly.

It was more difficult to model change in CER 2008–2018: RF models did not explain as much variance, nor were certain variables as clearly important (**Figure 5B**). Only 20.3% of the variance was explained in the Northwestern Wheat region, while 59.5% was explained in the South Atlantic LRR (mean 41.8%). This was not unexpected: while temperature and water availability constrain potential agricultural uses, many other factors, such as market availability, population density, and agricultural policy, go into determining actual uses. There was no relationship across LRRs between change in CER and maximum CER ($r^2 = 0.022$, $p = 0.5283$), between maximum agricultural area and maximum CER ($r^2 = 0.087$, $p = 0.2056$), or between change in CER ($r^2 = 0.001$, $p = 0.9722$). Growing degree days, temperature seasonality, and soil texture were again frequently important variables.

The shape of the partial dependence plots for maximum CER and change in CER differed across regions, demonstrating the importance of stratifying very large datasets when variable interpretation is desired. Growing degree days illustrates this clearly; the Northern Lake States (K) and the Northern Great Plains (F) showed opposite relationships with CER, even over the same range in growing degree days (**Figure 6A**). Relationships between environmental variables and change in CER did not necessarily have the same shape as with maximum CER (**Figure 6B**). For instance, in the Atlantic Coast LRR (T), CER declined with increasing growing degree days, but change increased with growing degree days, while for the Northern Great Plains, both CER and change in CER declined with growing degree days.

Describing Potential Diversity

The final phase of the analysis was to develop quantile RF models of irrigated and rainfed maximum CER across the entire United States. These models characterize potential CER by identifying regions similar to those where diverse agriculture is currently found, and cannot extrapolate beyond existing systems. The RF models were quite good, explaining 79.6% of the variance in CER in irrigated areas, and 84.8% in primarily rainfed grid cells. These models were then used to predict the 90th percentile of potential CER across the contiguous US, based on soils and climate.

Agricultural diversity could be increased with irrigation in most parts of the US (**Figure 7**). The Mississippi Delta would not increase, because its agricultural systems are already designed for irrigation. Florida CER also did not increase with irrigation; this region has high annual precipitation, but uses irrigation to compensate for precipitation variability (Zhang et al., 2018).

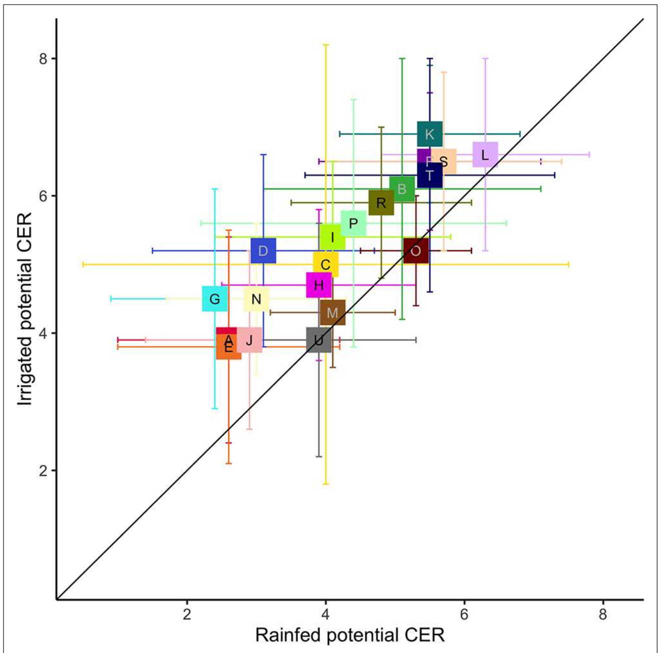


FIGURE 7 | The 90th percentile of maximum CER predicted by irrigated and rainfed RF models for the contiguous US. Values are mean and standard deviation for each LRR, following the colors and letter code given in **Figure 1** and **Table 1**.

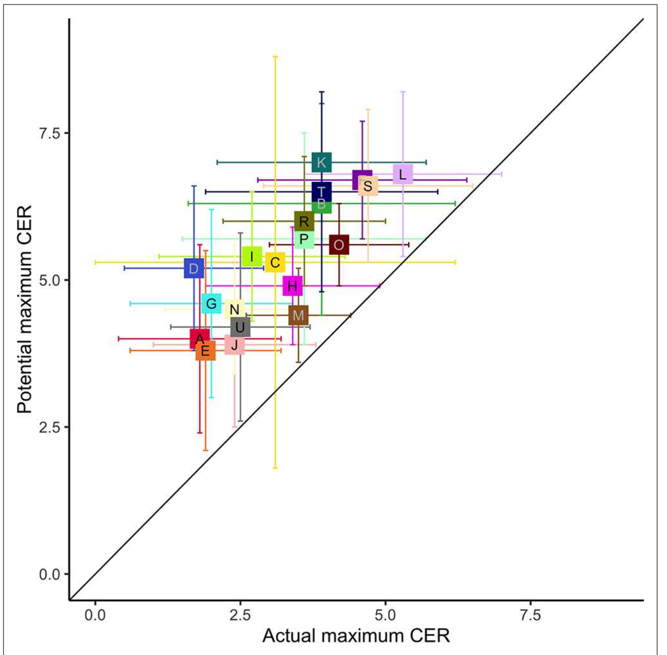


FIGURE 8 | The actual maximum CER for 2008–2018 and the maximum predicted value (rainfed or irrigated). Values are mean and standard deviation for each LRR, following the colors and letter code given in **Figure 1** and **Table 1**.

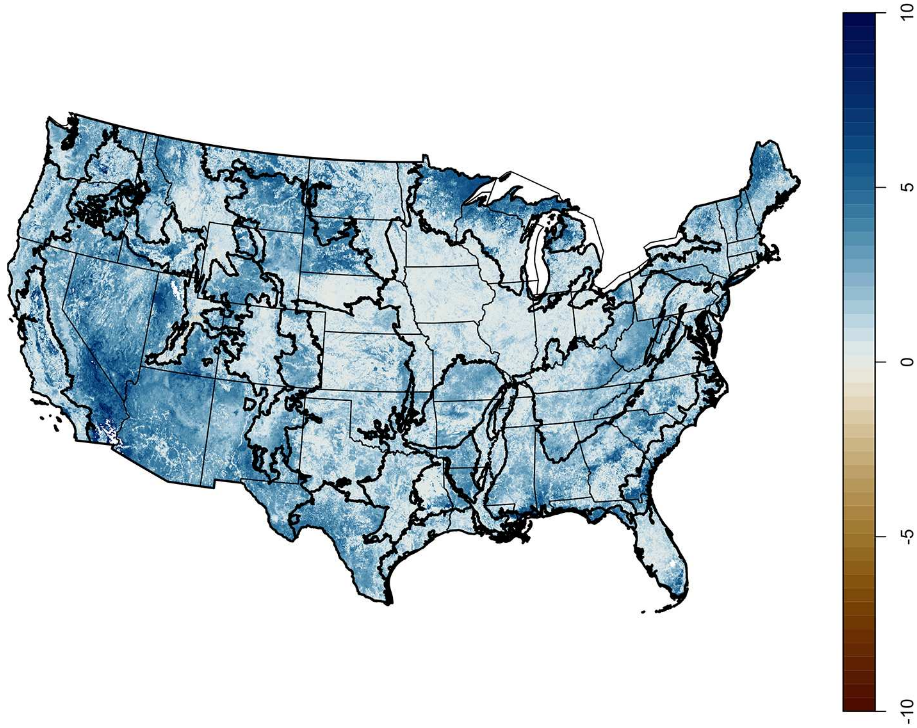


FIGURE 9 | The potential change in CER if the most diverse current system were applied across the contiguous US, given soils and climate.

Western Range, however, showed a large potential increase in agricultural diversity with irrigation. Interestingly, so did the Northern Lake States.

In all cases, the maximum potential agricultural diversity, the higher of the irrigated and rainfed predictions, was greater than the current agricultural diversity (**Figure 8**). Mapping that potential change (**Figure 9**), shows that some of the gain, particularly in the Northern Lake States, would come at the cost of clearing forest to expose areas of favorable climate and soils. Many areas could support more diverse agricultural systems if they were irrigated.

DISCUSSION

Analyses of crop diversity in the county-scale agricultural census data have found similar regional patterns (Aguilar et al., 2015; Hijmans et al., 2016), although these studies did not then go on to identify important driving variables. At the scale of LRRs, both temperature and water availability were important, although both the magnitude and shape of the relationship, and the relative importance of specific variables, varied considerably. When considering continental-scale ecological patterns, it is crucial to employ methods that can identify regional differences.

Increasing crop diversity may provide immediate benefits, such as the reduction of pathogen transmission and pest outbreaks, or more amorphous benefits by buffering climate variability (Lin, 2011). Potential crop effective richness is higher than the current level in nearly all of the coterminous US. However, irrigation is needed to achieve that potential, and both climate change and declines in aquifer levels make increasing irrigation, or even maintaining current levels, a difficult proposition. Instead, it may be necessary to evaluate the effect of transitioning to rainfed agriculture on crop diversity. For instance, the diverse agricultural systems in the California LRR are strongly dependent on irrigation (Matios and Burney, 2017). A transition to rainfed would result in change of crops and loss of overall diversity, unless new agricultural systems can be developed through alterations in policy, infrastructure, and breeding or selection of crops and varieties that are productive without irrigation.

These models do not include other constraints on agricultural usage: both topography (e.g., mountains) and current land use (development) render a site unsuitable for agriculture, as does certain ownership patterns, such as state and national forests and reserves. Clearing forests for agriculture would not be an overall benefit, regardless of potential agricultural diversity. The coarsest dataset included had a 4 km resolution. In some regions of the US, that is very large relative to local agricultural patterns, while in others it is rather small. No attempt to quantify landscape configuration within that grid cell was done for this study, although crop diversity was much less important for diversity of multiple taxa than compositional factors such as field size and overall agricultural area (Fahrig et al., 2015; Duflot et al., 2017).

The CDL does not capture sub-field heterogeneity, and does not identify either within-crop genetic diversity or within-field diversity in pasture and Harland, both of which may be important for ecosystem services (Jackson et al., 2007; Sanderson et al., 2007; Finger and Buchmann, 2015; Reiss and Drinkwater, 2018). This study does not explicitly include temporal diversity due to the use of cover crops or rotations, substituting instead the maximum

CER in any of the 11 years. Pooling all years to create a multiyear maximum diversity, instead of a spatial-only maximum diversity, might instead better capture temporal diversity created by complex rotations, which has been shown to be environmentally beneficial (Davis et al., 2012; McDaniel et al., 2014).

Decision support tools are being developed to assess tradeoffs in ecosystem services associated with different cropping systems (e.g., Tayyebi et al., 2016), but not enough information is available on the role of crop diversity within these systems to adequately quantify outcomes. Decisions on diversification are made at the farm- and field-scales, but have consequences for both ecosystem services and food security at much broader spatial extents. Tools such as crop models may provide linkages between agricultural management and fine-scale patterning and ecosystem services (Lin, 2011), within the broader biophysical context.

Agricultural field management practices such as tillage and pesticide application have very strong implications for ecosystem service provision. Future research will explore methods for including this information in regional and national analyses, as well as incorporating spatial and temporal components of agricultural diversity, and a more nuanced consideration of functional and structural crop diversity. Functional diversity may be more important for agroecosystems functioning than taxonomic diversity, as used here, although taxonomic diversity provides provisioning and insurance services (Jackson et al., 2007; Martin and Isaac, 2018).

Variables related to temperature and water availability were effective at modeling current maximum agricultural diversity, but not as good at modeling change in agricultural diversity. Economic and policy incentives often benefit monoculture systems more than diverse systems (Lin, 2011). Weigel et al. (2018) found lower agricultural diversity on higher-quality soils, where agriculture can be concentrated on the most valuable crops with the least risk. That appears to be the case in the most heavily-agricultural and most productive regions of the US as well. The importance of water availability suggests that other relevant management practices, such as the installation of tile drains, could be a major driver in some areas. No comprehensive dataset on this practice is currently available, but would be highly useful.

As for any statistical method, Random Forest models can only be used to predict within the bounds of the training data. In this context, RF models can identify areas that are like currently-diverse areas, but they cannot predict the role of entirely new systems. This limitation is clearly visible in the small potential increase in diversity predicted in the Central Grains region: the current agricultural system for that combination of climate and soils is so heavily dominated by monoculture grain and soybeans that the RF models cannot predict anything else. Within that limitation, these models can be used to predict potential changes in diversity due to addition or removal of irrigation, and due to changes in temperature and precipitation. Most studies of climate change effects on agriculture concentrate on crop yield, rather than area or diversity (e.g., Kang et al., 2009). The models developed in this study are based on standard climatic indices, and can be used with climate projections to model potential future outcomes if the same agricultural systems continue to be employed.

CONCLUSIONS

Agricultural ecosystems are maintained at a lower rate of biodiversity than the natural ecosystems found in comparable regions (Altieri, 1999); increasing agricultural diversity may improve ecosystem services such as nutrient cycling, pest and pathogen control, and even the provision of high-quality foodstuffs. Crop diversity as currently constituted in the United States is heavily reliant on irrigation. To achieve greater ecosystem service benefits from increased crop diversity, alternative agricultural systems must be developed. The spatial and temporal scale of the analysis here focused on structural and systemic aspects of crop diversity, rather than on individual crop selection, but clearly there will be a role for crop species and varieties that are productive under these alternative systems, as long as the use of one or a few varieties only does not reduce system diversity rather than enhance it.

The biophysical constraints on diversity imposed by temperature and water availability explain much of the broad pattern of diversity in the US, although the pattern of change in diversity is less clearly explained. Variable importance, and even the shape of the relationship between particular variables and crop effective richness, differs by region.

Management-based control of potential diversity through irrigation is the primary control on agricultural diversity. In the western US, climate change and declining water supply may require the transition of irrigated agriculture to rainfed by the end of this century; the eastern US may be able to support an increase in irrigated area with development of infrastructure (Elliott et al., 2014).

It may not be possible to optimize both agricultural diversity and food production everywhere (Holt et al., 2016). Nonetheless, a spatial understanding of the potential crop diversity offered by current agricultural systems aids in planning regional and national policies, and in evaluating the effects of novel practices that increase spatial and temporal agricultural diversity.

DATA AVAILABILITY STATEMENT

The datasets analyzed for this study are publicly available from the referenced sources.

AUTHOR CONTRIBUTIONS

SG designed the project, conducted the analyses, and drafted the manuscript.

FUNDING

This research was supported by the U.S. Department of Agriculture, Agricultural Research Service. Mention of trade names or commercial products in this publication is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the U.S. Department of Agriculture. USDA is an equal opportunity provider and employer.

ACKNOWLEDGMENTS

Doug Miller and Martina Calovi provided insightful comments at various stages of this project. This research was a contribution from the Long-Term Agroecosystem Research (LTAR) network. LTAR is supported by the United States Department of Agriculture.

REFERENCES

Aguilar, J., Gramig, G. G., Hendrickson, J. R., and Archer, D. W. (2015). Crop species diversity changes in the United States: 1978–2012. *PLoS ONE* 10:e0136580. doi: 10.1371/journal.pone.0136580

Altieri, M. A. (1999). The ecological role of biodiversity in agroecosystems. *Agric. Ecosyst. Environ* 74, 19–31. doi: 10.1016/S0167-8809(99)00028-6

Bivand, R. S., E., Pebesma, and, V., and Gomez-Rubio (2013). *Applied Spatial Data Analysis with R, Second edition*. New York, NY: Springer. doi: 10.1007/978-1-4614-7618-4

Bommarco, R., Kleijn, D., and Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends Ecol. Evol. (Amst)* 28, 230–238. doi: 10.1016/j.tree.2012.10.012

Boryan, C., Yang, Z., Mueller, R., and Craig, M. (2011). Monitoring US agriculture: the US Department of Agriculture, National Agricultural Statistics Service, Cropland Data Layer Program. *Geocarto. Int.* 26, 341–358. doi: 10.1080/10106049.2011.562309

Bowman, M. S., and Zilberman, D. (2013). Economic factors affecting diversified farming systems. *Ecol. Soc.* 18:33. doi: 10.5751/ES-05574-180133

Brown, J. F., and Pervez, M. S. (2014). Merging remote sensing data and national agricultural statistics to model change in irrigated agriculture. *Agric. Syst* 127, 28–40. doi: 10.1016/j.agsy.2014.01.004

Busby, J. R. (1991). BIOCLIM-a bioclimate analysis and prediction system. *Plant Protect. Q.* 6, 8–9.

Davis, A. S., Hill, J. D., Chase, C. A., Johanns, A. M., and Liebman, M. (2012). Increasing cropping system diversity balances productivity, profitability and environmental health. *PLoS ONE* 7:e47149. doi: 10.1371/journal.pone.0047149

Devictor, V., Mouillot, D., Meynard, C., Jiguet, F., Thuiller, W., and Mouquet, N. (2010). Spatial mismatch and congruence between taxonomic, phylogenetic and functional diversity: the need for integrative conservation strategies in a changing world. *Ecol. Lett.* 13, 1030–1040. doi: 10.1111/j.1461-0248.2010.01493.x

Di Falco, S., and Perrings, C. (2005). Crop biodiversity, risk management and the implications of agricultural assistance. *Ecol. Econom.* 55, 459–466. doi: 10.1016/j.ecolecon.2004.12.005

Duflot, R., Ernoult, A., Aviron, S., Fahrig, L., and Burel, F. (2017). Relative effects of landscape composition and configuration on multi-habitat gamma diversity in agricultural landscapes. *Agric. Ecosyst. Environ.* 241, 62–69. doi: 10.1016/j.agee.2017.02.035

Elliott, J., Deryng, D., Müller, C., Frieler, K., Konzmann, M., Gerten, D., et al. (2014). Constraints and potentials of future irrigation water availability on agricultural production under climate change. *Proc. Natl. Acad. Sci. U.S.A.* 111, 3239–3244. doi: 10.1073/pnas.1222474110

Fahrig, L., Girard, J., Duro, D., Pasher, J., Smith, A., Javorek, S., et al. (2015). Farmlands with smaller crop fields have higher within-field biodiversity. *Agric. Ecosyst. Environ.* 200, 219–234. doi: 10.1016/j.agee.2014.11.018

Finger, R., and Buchmann, N. (2015). An ecological economic assessment of risk-reducing effects of species diversity in managed grasslands. *Ecol. Econ.* 110, 89–97. doi: 10.1016/j.ecolecon.2014.12.019

Greenwell, B. M. (2017). pdp: An R package for constructing partial dependence plots. *R J.* 9, 421–436. doi: 10.32614/RJ-2017-016

Hengl, T., de Jesus, J. M., Heuvelink, G. B. M., Gonzalez, M. R., Kilibarda, M., Blagoti,ć, A., et al. (2017). SoilGrids250m: Global gridded soil information based on machine learning. *PLoS ONE* 12:e0169748. doi: 10.1371/journal.pone.0169748

Hijmans, R. J., Choe, H., and Perlman, J. (2016). Spatiotemporal patterns of field crop diversity in the United States, 1870–2012. *Agricul. Environ. Lett.* 1:160022. doi: 10.2134/ael2016.05.0022

Holt, A. R., Alix, A., Thompson, A., and Maltby, L. (2016). Food production, ecosystem services and biodiversity: we can't have it all everywhere. *Sci Total Environ.* 573, 1422–1429. doi: 10.1016/j.scitotenv.2016.07.139

Jackson, L. E., Pascual, U., and Hodgkin, T. (2007). Utilizing and conserving agrobiodiversity in agricultural landscapes. *Agric. Ecosyst. Environ.* 121, 196–210. doi: 10.1016/j.agee.2006.12.017

Jerrentrup, J. S., Dauber, J., Strohbach, M. W., Mecke, S., Mitschke, A., Ludwig, J., et al. (2017). Impact of recent changes in agricultural land use on farmland bird trends. *Agric. Ecosyst. Environ.* 239, 334–341. doi: 10.1016/j.agee.2017.01.041

Jost, L. (2006). Entropy and diversity. *Oikos* 113, 363–375. doi: 10.1111/j.2006.0030-1299.14714.x

Kang, Y., Khan, S., and Ma, X. (2009). Climate change impacts on crop yield, crop water productivity and food security – A review. *Progress Nat. Sci.* 19, 1665–1674. doi: 10.1016/j.pnsc.2009.08.001

Kennedy, C. M., Lonsdorf, E., Neel, M. C., Williams, N. M., Ricketts, T. H., Winfree, R., et al. (2013). A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecol. Lett.* 16, 584–599. doi: 10.1111/ele.12082

Khoury, C. K., Bjorkman, A. D., Dempewolf, H., Ramirez-Villegas, J., Guarino, L., Jarvis, A., et al. (2014). Increasing homogeneity in global food supplies and the implications for food security. *Proc. Natl. Acad. Sci. U.S.A.* 111, 4001–4006. doi: 10.1073/pnas.1313490111

Lark, T. J., Mueller, R. M., Johnson, D. M., and Gibbs, H. K. (2017). Measuring land-use and land-cover change using the U.S. Department of Agriculture's Cropland Data Layer: Cautions and recommendations. *Int. J. Appl. Earth Observ. Geoinform.* 62, 224–235. doi: 10.1016/j.jag.2017.06.007

Lin, B. B. (2011). Resilience in agriculture through crop diversification: Adaptive management for environmental change. *Bioscience* 61, 183–193. doi: 10.1525/bio.2011.61.3.4

Martin, A. R., Cadotte, M. W., Isaac, M. E., Milla, R., Vile, D., and Violle, C. (2019). Regional and global shifts in crop diversity through the Anthropocene. *PLoS ONE* 14:e0209788. doi: 10.1371/journal.pone.0209788

Martin, A. R., and Isaac, M. E. (2018). Functional traits in agroecology: advancing description and prediction in agroecosystems. *J. Appl. Ecol.* 55, 5–11. doi: 10.1111/1365-2664.13039

Massawe, F., Mayes, S., and Cheng, A. (2016). Crop diversity: an unexploited treasure trove for food security. *Trends Plant Sci.* 21, 365–368. doi: 10.1016/j.tplants.2016.02.006

Matios, E., and Burney, J. (2017). Ecosystem services mapping for sustainable agricultural water management in California's Central Valley. *Environ. Sci. Technol.* 51, 2593–2601. doi: 10.1021/acs.est.6b05426

McDaniel, M. D., Tiemann, L. K., and Grandy, A. S. (2014). Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. *Ecol. Appl.* 24, 560–570. doi: 10.1890/13-0616.1

Meinshausen, N. (2017). *quantregForest: Quantile Regression Forests R Package 1241 Version 1.3–7*. Available online at: https://CRAN.R-project.org/package=quantregForest (accessed May 14, 2020).

Pebesma, E. J., and Bivand, R. S. (2005). Classes and Methods for Spatial Data in R. *R News* 5, 9–13.

Pervez, M. S., and Brown, J. F. (2010). Mapping irrigated lands at 250-m scale by merging MODIS data and national agricultural statistics. *Remote Sens.* 2, 2388–2412. doi: 10.3390/rs2102388

Plourde, J. D., Pijanowski, B. C., and Pekin, B. K. (2013). Evidence for increased monoculture cropping in the Central United States. *Agric. Ecosyst. Environ.* 165, 50–59. doi: 10.1016/j.agee,.2012.11.011

PRISM Climate Group (2018). *PRISM 4km Gridded Daily Temperature and Precipitation Data*. Corvallis, OR: Oregon State University.

R Core Team (2019). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Ramcharan, A., Hengl, T., Nauman, T., Brungard, C., Waltman, S., WIlls, S., et al. (2018). Soil property and class maps of the conterminous United States at 100-meter spatial resolution. *Soil Sci. Soc. Am. J.* 82, 186–116. doi: 10.2136/sssaj2017.04.0122

Reiss, E. R., and Drinkwater, L. E. (2018). Cultivar mixtures: a meta-analysis of the effect of intraspecific diversity on crop yield. *Ecol. Appl.* 28, 62–77. doi: 10.1002/eap.1629

Reitsma, K. D., Clay, D. E., Clay, S. A., Dunn, B. H., and Reese, C. (2016). Does the U.S. Cropland Data Layer provide an accurate benchmark for land-use change estimates? *Agron. J.* 108, 266–272. doi: 10.2134/agronj2015.0288

Sanderson, M. A., Goslee, S. C., Soder, K. J., Skinner, R. H., Tracy, B. F., and Deak, A. (2007). Plant species diversity, ecosystem function, and pasture management: a perspective. *Can. J. Plant Sci.* 87, 479–487. doi: 10.4141/P06-135

Spawn, S. A., Lark, T. J., and Gibbs, H. K. (2019). Carbon emissions from cropland expansion in the United States. *Environ. Res. Lett.* 14:045009. doi: 10.1088/1748-9326/ab0399

Tayyebi, A., Meehan, T. D., Dischler, J., Radloff, G., Ferris, M., and Gratton, C. (2016). SmartScape^{TM}: a web-based decision support system for assessing the tradeoffs among multiple ecosystem services under crop-change scenarios. *Comput. Electron. Agricult.* 121, 108–121. doi: 10.1016/j.compag.2015.12.003

USDA (2006). *Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin*. Washington, DC: United States Department of Agriculture Handbook, 296.

USDA (2019). *2017 Census of Agriculture*. Washington, DC: United States Department of Agriculture. Available online at: https://www.nass.usda.gov/Publications/AgCensus/2017/index.php (accessed May 14, 2020).

Wang, A., Goslee, S. C., Miller, D. A., Sanderson, M. A., and Gonet, J. M. (2017). Topographic variables improve climatic models of forage species abundance in the northeastern United States. *Appl. Veget. Sci.* 20, 84–93. doi: 10.1111/avsc.12284

Weigel, R., Koellner, T., Poppenborg, P., and Bogner, C. (2018). Crop diversity and stability of revenue on farms in Central Europe: an analysis of big data from a comprehensive agricultural census in Bavaria. *PLoS ONE* 13:e0207454. doi: 10.1371/journal.pone.0207454

Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis*. New York, NY: Springer-Verlag. doi: 10.1007/978-3-319-24277-4_9

Wright, M. N., and Ziegler, A. (2017). ranger: a fast implementation of Random Forests for high dimensional data in C++ and R. *J. Stat. Softw.* 77, 1–17. doi: 10.18637/jss.v077.i01

Zhang, M., devLeon, C., and Migliaccio, K. (2018). Evaluation and comparison of interpolated gauge rainfall data and gridded rainfall data in Florida, USA. *Hydrol. Sci. J.* 63, 561–582. doi: 10.1080/02626667.2018.1444767

Conflict of Interest: The author declares that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 05 June 2020
doi: 10.3389/fenvs.2020.00066

Trends in Land Use, Irrigation, and Streamflow Alteration in the Mississippi River Alluvial Plain

Lindsey M. W. Yasarer[1]*, *Jason M. Taylor*[1], *James R. Rigby*[2] *and Martin A. Locke*[1]

[1] Water Quality and Ecology Research Unit, National Sedimentation Laboratory, Agricultural Research Service, United States Department of Agriculture, Oxford, MS, United States, [2] Watershed Physical Processes Research Unit, National Sedimentation Laboratory, Agricultural Research Service, United States Department of Agriculture, Oxford, MS, United States

The Mississippi River Alluvial Plain is a critical region for agricultural production in the United States, providing the majority of the nation's rice, catfish, and cotton. Although it is a humid region, high agricultural yields are maintained through irrigation from groundwater and surface water sources. Heavy groundwater extraction has led to cones of depression in the alluvial aquifer in both Arkansas and Mississippi. This study explores the link between increasing irrigation and streamflow alteration within the alluvial plain. Changing land use patterns were evaluated utilizing the USDA Census of Agriculture datasets to determine changes in land-use, irrigation, and crop yield from 1969 to 2017. Temporal land use patterns set the background for the analysis of sixteen long-term streamflow records from the USGS, which were assessed using the Indicators of Hydrologic Alteration (IHA) software to determine changes in low flow patterns in rivers overlying the Mississippi River Valley alluvial aquifer. Most streamflow records had significant hydrologic alteration with respect to low flow conditions, including higher frequency of low flow events, lower annual minima, or a declining base flow index. Changes in streamflow coincide with areas of massive increases in irrigated cropland area. This study provides further context for the tradeoffs between intensive agricultural production and agroecosystem sustainability.

Keywords: irrigation, streamflow, groundwater, land use, Mississippi, Arkansas, Louisiana

OPEN ACCESS

Edited by:
Mark A. Drummond,
United States Geological Survey (USGS), United States

Reviewed by:
Elzbieta Antczak,
University of Łódz, Poland
Jennifer Anna Maria Koch,
University of Oklahoma, United States

***Correspondence:**
Lindsey M. W. Yasarer
lindsey.yasarer@usda.gov

Specialty section:
This article was submitted to Land Use Dynamics, a section of the journal Frontiers in Environmental Science

Received: *06 December 2019*
Accepted: *07 May 2020*
Published: *05 June 2020*

Citation:
Yasarer LMW, Taylor JM, Rigby JR and Locke MA (2020) Trends in Land Use, Irrigation, and Streamflow Alteration in the Mississippi River Alluvial Plain. Front. Environ. Sci. 8:66. doi: 10.3389/fenvs.2020.00066

INTRODUCTION

The Mississippi River Alluvial Plain (MAP), including most of eastern Arkansas, western Mississippi, and northeastern Louisiana, relies on agricultural production to drive the regional economy (Alhassan et al., 2019). The region is intensively farmed in row-crops that produce significant yields of corn (*Zea* mays L.) and soybean [*Glycine max* (L.) Merr.], and accounts for ∼17–20% of total cotton (*Gossypium hirsutum* L.) and 69–78% of total rice (*Oryza sativa* L.) production nationally. Mississippi also returns the highest aquaculture yields in the nation (U.S. Department of Agriculture, National Agricultural Statistics Service, 2019). Although it is a humid region, the MAP receives most of its rainfall outside of the growing season, and thus producers rely on irrigation from either groundwater or surface water to reduce crop stress and to optimize crop yields (Massey et al., 2017). As of 2000, the Mississippi River Valley Alluvial Aquifer (MRVAA) was ranked third in the nation for total water withdrawals (35 billion liters per day) with more than 98% of this water used for irrigation (Maupin and Barber, 2005). Withdrawals from the MRVAA began as early as the 1900s but increased markedly between 1970 and 1980 (Evett et al., 2003; Clark et al., 2011). As a result of increased irrigation withdrawals, cones of depression have developed in both the Arkansas and Mississippi delta regions, which has drawn attention to the sustainability of groundwater resources for continued irrigation and agricultural economic development (Barlow and Clark, 2011; Kresse et al., 2014).

Generally, as groundwater levels decline, historically gaining streams receiving groundwater contributions may become losing, perched streams, where surface water seeps through an unsaturated zone into the aquifer (Brunner et al., 2011). Changes in surface water and groundwater exchange, or increased pumping of surface water for irrigation, can alter natural flow regimes in streams and rivers, contributing to decreased baseflow, as well as more frequent and extreme low flow conditions. The effect of groundwater extraction on nearby streamflow has been studied and modeled extensively (Hunt, 1999; Butler and Tsou, 2001; Fox and Durnford, 2003). However, there are spatial and temporal variations in surface-groundwater connectivity that are difficult to assess due to heterogeneity of streambed properties, variations in depth to groundwater both spatially and temporally, and disparities in infiltration rates due to depth and width of surface water bodies. However, it is generally accepted that a widening cone of depression in groundwater surface levels can increase the length of river disconnected from the aquifer and therefore alter baseflow (Brunner et al., 2011).

In addition to influencing water availability for agriculture, decreasing baseflow and increased number of extreme low flow events can have detrimental effects on aquatic ecosystems and associated biodiversity. Previous studies have demonstrated declines in fish species richness and abundance in conjunction with shifts from species with more specialized requirements to more tolerant generalist species as summer baseflows decline (Freeman and Marcinek, 2006; McCargo and Peterson, 2010; Buchannan et al., 2017). Similarly, loss of surface water flow and groundwater connectivity can result in significant declines in mussel richness and abundance due to habitat loss and thermal requirements (Golladay et al., 2004; Galbraith et al., 2010). In addition to dewatering critical habitat including shallow margins, coarse woody debris, riffle habitats (Bowen et al., 1998; Freeman et al., 2001; McCargo and Peterson, 2010), decreasing baseflows increase biological impairment by reducing dissolved oxygen, increasing water temperature, concentrating contaminants, and increasing diel swings in pH which can increase bioavailability and toxicity of contaminants to aquatic organisms (Brooks et al., 2006; Garvey et al., 2007; Carlisle et al., 2011; Valenti et al., 2011). Outside of river channels, loss of groundwater connectivity in riparian areas can produce mesic conditions that decrease riparian vegetation species richness, biomass and cover, or result in shifts from wetland to upland plant assemblages (Hanlon et al., 1998; Williams et al., 1999). River morphology can also be altered by these low flow conditions as the bed load will be deposited as flow velocity decreases, which may alter physical habitats and streambed substrate (Hauer et al., 2013).

The link between increasing crop irrigation, either with surface or groundwater, and streamflow alteration is often tenuous, due to information that may be lacking about the system—either irrigation usage, groundwater conditions, or the physical connection between groundwater and surface water bodies. However, several recent studies explore the concept of increased groundwater pumping and its impact on local streamflow depletion (Killian et al., 2019) and global environmental flow limits (de Graaf et al., 2019). This study evaluates changing land-use patterns in the Mississippi Alluvial Plain (MAP) by utilizing United States Department of Agriculture (USDA) Census of Agriculture datasets to determine changes in land-use, irrigation, and crop yield from 1969 to 2012. We used temporal trends in land use patterns to set the context for analysis of long-term patterns in low flow metrics in rivers overlying the MRVAA using the Indicators of Hydrologic Alteration (IHA) software. Based on conceptual linkages between the increase in irrigated cropland to both increased yield and increased streamflow alteration (**Figure 1**), we hypothesized that low flow components in MAP streams have been altered and these environmental deficits coincide with declining groundwater levels and increased irrigation demands within the region.

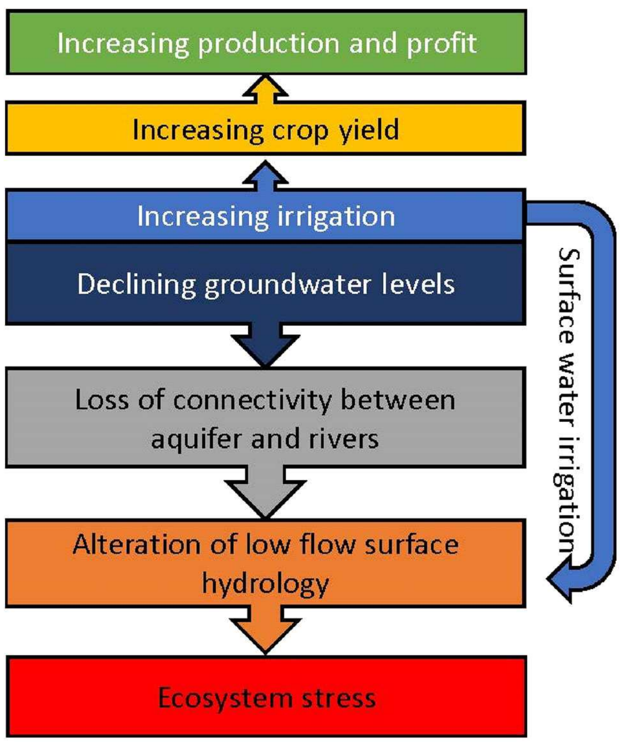


FIGURE 1 | Conceptual diagram outlining the hypothesized costs and benefits of increasing irrigation in the Mississippi Alluvial Plain. The ultimate benefit is increased production and profit, while the ultimate cost is ecosystem stress. This study focuses on the aspects from crop yield to alteration of low flow hydrology.

METHODS

Land Use Trend Analysis

We compiled records from the USDA Census of Agriculture including area in total cropland, harvested cropland, land in farms with irrigation, and irrigated cropland from 1969 to 2017 at the county or parish level to evaluate the change in area over time (Census of Agriculture, 1969–2017). In addition, we collected crop area harvested, harvested crop amount, and irrigated crop area for corn for grain, sorghum for grain, winter wheat, rice, upland cotton, and soybeans for each county or parish from 1969 to 2017. The census data was available every 5 years, except from 1974 to 1982 when it was collected every 4 years. The USDA Census of Agriculture is conducted by the USDA National Agricultural Statistics Service (NASS) and includes a complete count of U.S. farms and ranches, rural and urban, where $1,000 or more fruit, vegetables, or food animals were raised or sold. Census data was mapped to demonstrate differences between counties.

We analyzed counties/parishes if they intersected, or were located within, the watershed of a selected USGS gage, or if they were just downstream of a USGS gage. With respect to groundwater, the area of influence may include areas downstream of a surface water gage due to the influence of groundwater withdrawals. We calculated county-level yield for each crop type by dividing the total mass harvested in each county by the harvested area of the respective crop type. Crop mass harvested in the census does not differentiate between irrigated and non-irrigated cropland. We evaluated the difference in yield based on irrigation status utilizing available data from the USDA crop survey, however, only cotton yield was available for Louisiana, Arkansas, and Mississippi in both irrigated and non-irrigated cropland. We calculated the average and standard deviation of yield in the evaluated counties using available years of data to evaluate the effect of irrigation on yield. Availability of data differed by county with most counties providing yield from 1971 to 2018 (U.S. Department of Agriculture, National Agricultural Statistics Service, 2019). The Agricultural Resource Management Survey is conducted by the USDA NASS and includes a sample of farm operators that ensures adequate coverage by station and region.

Stream Gage Selection and Data Preparation

We identified USGS stream gages within the MAP region based on the availability of daily flow records of at least 20 years with minimal gaps (Kennard et al., 2010; **Table 1**). We utilized the Sunflower River at Sunflower despite a large data gap, as there was a sufficient period of data available before and after the gap, and the gage represented a critical geographic area. We did not fill gaps in the streamflow record. It was challenging to find enough gauges with the same time period of available data; therefore, the data range varied for the gauges in this study. Selected gages were also either unregulated or regulated with diversions for irrigation, as determined by information provided by the USGS StreamStats database (U.S. Geological Survey, 2016). Any gages with major upstream flood control dams were not included in the study, as releases from reservoirs can influence the flow record. To increase comparability between sites, flow was converted to stream yield (mm/day) by dividing by the watershed area provided by the USGS StreamStats website.

Indicators of Hydrologic Alteration (IHA)

We assessed potential differences and trends in selected baseflow metrics over the studied time period representing before and after increasing irrigation withdrawals using Indicators of Hydrologic Alteration (IHA) software version 7.1 (The Nature Conservancy, 2009). The program uses daily hydrologic data to assess 33 ecologically-relevant hydrologic parameters related to five fundamental characteristics of hydrologic regimes: (1) magnitude of mean daily water conditions; (2) magnitude and duration of annual extreme conditions at various time intervals; (3) timing of annual extreme conditions; (4) frequency and duration of high and low pulses, and; (5) rate and frequency of change in conditions (Richter et al., 1996). For all stations, we ran the IHA using daily stream yield (mm/day) for the period of available data using temporal trend analysis. Stream yield was utilized to control for watershed size and allow temporal trends to be comparable across gages. We analyzed time series trends from the IHA, and the number of indicators demonstrating an increase in low flow conditions or decreasing flow levels (i.e., negative monthly flow trends) with a significance level ≤ 0.05 were summed for each gage to evaluate the degree of alteration occurring across the region. Due to the large number of indicators, not all results are presented. We chose to highlight a subset of IHA metrics that, based on the literature, were hypothesized to respond most strongly to water withdrawals (Carlisle et al., 2011; Kennen et al., 2014). The IHA parameters highlighted in this study include the base flow index, 7 day minimum flows, and the number of days with zero flow per year. Base flow index is calculated as the 7 day minimum flow divided by the annual mean flow. The 7 day minimum flow is the 7 day mean of the annual minima.

For a selected subset of stations, Cache River at Egypt, AR; Languille River near Colt, AR; Big Sunflower River at Sunflower, MS; Boeuf River near Girard, LA; and Tensas River at Tendal, LA, we conducted a comparative analysis using streamflow (cms) to compare low flow metrics between two time periods representing historic, relatively unaltered, flow conditions and current stream flows under increasing irrigation demands. We utilized the results of this two-period analysis to produce flow duration curves for the pre-alteration period (start of record–1986) and the post alteration period (1987–2016), as well as to examine the change in 7-day minimum flow before and after 1987. We chose the year of 1987 as a breakpoint due to the analysis of census data indicating a rise in irrigated cropland area at this time (**Figure 3A**), as well as evidence that surface and groundwater connectivity began to change at this time in the MRVAA region (Clark et al., 2011; Pugh and Westerman, 2014).

Precipitation Trends and Drought Occurrence

We analyzed regional precipitation trends using the climate division database (nClimDiv) from the National Oceanic and Atmospheric Administration (NOAA) (Vose et al., 2014).

TABLE 1 | USGS gages utilized in study, including period of record, drainage area, and any known flow regulations from the USGS StreamStats program, listed in order from north to south (U.S. Geological Survey, 2016).

Map #	Gage name	USGS number	Period of record	Drainage area [km^2]	Flow regulations
1	Cache River at Egypt, AR	7077380	1965–2016	1,816	Unknown
2	Languille River near Colt, AR	7047942	1971–2016	1,386	Unknown
3	Cache River near Cotton Plant, AR	7077555	1987–2016	3,030	Unknown; small diversions observed for irrigation
4	Bayou Meto near Lonoke, AR	7264000	1955–2016	536	Some diversions for irrigation; low flow supplemented by irrigation runoff and pond drainage
5	Bayou Bartholomew at Garret Bridge, AR	7364133	1987–2015	984	Unknown; minor diversions possibly for irrigation
6	Bayou Bartholomew near McGehee, AR	7364150	1941–2016	1,492	Unknown; minor diversions possibly for irrigation
7	Bayou Macon at Eudora, AR	7369680	1988–2015	1,295	None; there may be minor diversions for irrigation
8	Big Sunflower River near Merigold, MS	7288280	1993–2015	1,432	Unknown; pumping for irrigation could be substantial.
9	Big Sunflower River at Sunflower, MS	7288500	1935–1980 2004–2015	1,987	Streamflow augmented by irrigation runoff; withdrawals for irrigation and 6cfs for industrial use
10	Bougue Phalia near Leland, MS	7288650	1996–2015	1,254	None
11	Abiaca Creek near Seven Pines, MS	7287150	1991–2011	247	None
12	Bayou Bartholomew near Jones, LA	7364200	1957–2016	3,074	Unknown
13	Bayou Macon near Delhi, LA	7370000	1940–2002	2,025	Small diversions for irrigation
14	Boeuf River near Girard, LA	7368000	1941–2016	3,175	Large diversions for irrigation; interchange of flow between Boeuf River and Bayou Lafourche by canal
15	Tensas River at Tendal, LA	7369500	1941–2016	800	Small diversions for irrigation

Divisions include: northeast, east central, and southeast Arkansas; the upper delta and lower delta in Mississippi; northeast Louisiana; and the Missouri bootheel. Because the period of record varies for the gages, we utilized 1960–2016 as the period to estimate mean precipitation for both the annual and growing season (April–October) time periods, as well as temporal trends over that time period. In addition, we also assessed the occurrence of droughts throughout the 1960–2016 record using the Palmer Drought Severity Index (PDSI). We considered droughts to have occurred in months with PDSI scores less than negative three, which is considered a severe drought.

RESULTS AND DISCUSSION

Trends in Land Use, Yield, and Irrigation

County-level census data demonstrated that total cropland and harvested cropland areas have both declined over the period of record from 1969 to 2017. Meanwhile, the amount of irrigated cropland has increased, on average, by 45,000 hectares in all counties analyzed and, in several counties, total irrigated land increased by over 80,000 hectares. In Chicot County, AR, the location of Bayou Macon gage at Eudora, the amount of irrigated land increased 83,000 hectares, a 1,264% increase from 1969 values. Also, in Bolivar County, MS, the location of the Big Sunflower River gage at Merigold, the amount of irrigated land increased 94,000 hectares, or around 541% over 1969 values (**Figure 2**). From a different perspective, the amount of harvested cropland that was irrigated in the region has increased on average from 11% in 1969 to 69% in 2017 (**Figure 2**). Increases in irrigated cropland were highest in the southeastern AR with several counties reporting over 90% of all cropland with irrigation in 2017 (**Figure 2**).

While the irrigated cropland trend has risen steadily since 1974, land in irrigated farms experienced a large jump in 1978 and then increased more slowly until leveling out around 1992 (**Figure 3A**). There are ∼5 million hectares of land within irrigated farms that are not irrigated, suggesting irrigated cropland areas could increase further if producers expand their irrigation practices to additional land on their farms. One factor contributing to increased irrigated land has been land forming, or precision leveling, which creates a consistent slope to facilitate furrow irrigation and improve surface drainage (Maletic and Hutchings, 1967; Massey et al., 2017). Furrow irrigation is the predominate irrigation application method in the MAP region with 75% of irrigated land in MS and 80% of irrigated land in AR and LA in furrow irrigation (Kebede et al., 2014). Furrow irrigation is known to cause deep percolation losses, as well as tail-water runoff, which result in inefficient use of aquifer water.

Over the period from 1969 to 2017 there were some minor changes in crop types within the study area (**Figure 3B**). Harvested soybean declined from 1969 to 1992 and then began to

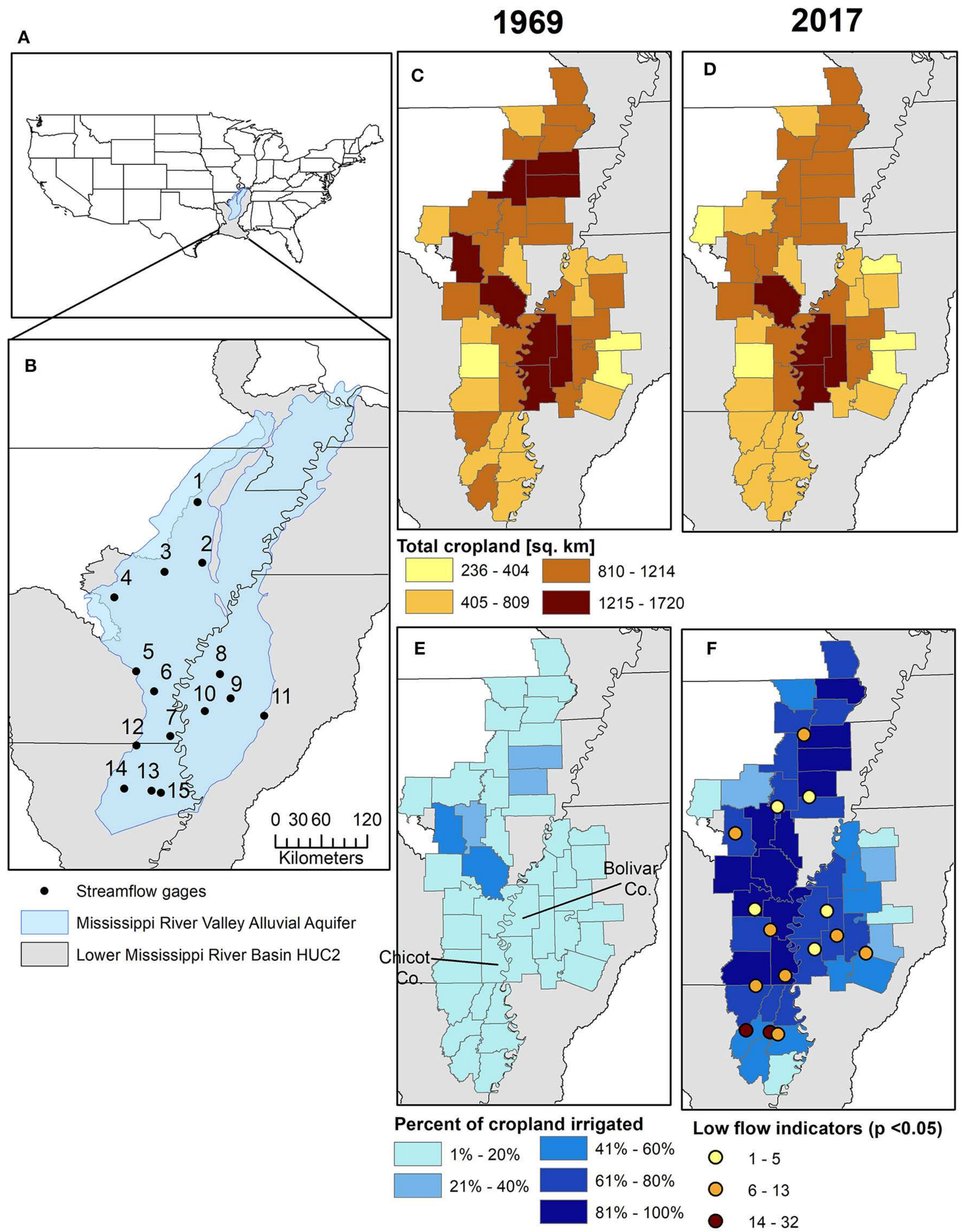


FIGURE 2 | (A) Location of Mississippi Alluvial Plain within the United States; **(B)** location of gauge locations within Mississippi Alluvial Plain and Lower Mississippi River Basin, numbers correspond to gauge information in **Table 1**; **(C)** harvested cropland by county and parish in 1969; **(D)** and 2017; **(E)** percent of cropland with irrigation in counties and parishes in 1969; **(F)** and 2017 with the color of USGS representing the number of low flow indicators with a statistically significant increasing trend.

increase from 2007 to 2017. Cotton declined from 1992 to 2012, while harvested corn increased during the same time frame. Our analysis demonstrates that corn area increased in most counties within the AR, LA, and MS portions of the MRVVA. **Figure 3B** demonstrates the change in area for the six major crops analyzed. Slight increases in grain sorghum and winter wheat were also

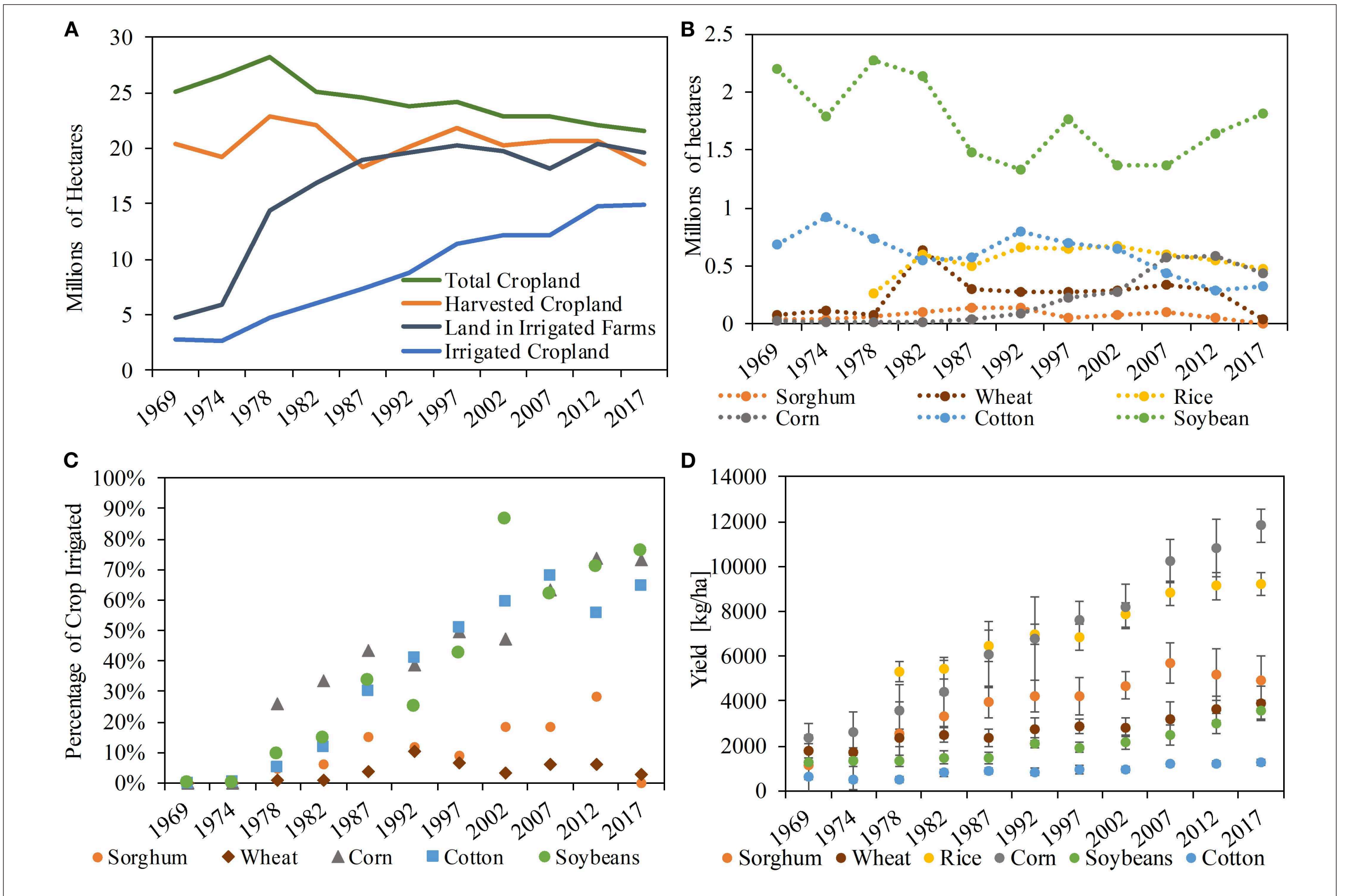


FIGURE 3 | (A) Temporal trends in total cropland, harvested cropland, land in irrigated farms, and irrigated cropland areas from 1969 to 2017 for all counties/parishes examined in study; **(B)** total crop area by major crop type in all counties/parishes analyzed from 1969 to 2017; **(C)** percentage of crop irrigated for all counties/parishes studied. Rice is not plotted as irrigation was 100% for all years that crop irrigation status is reported; **(D)** calculated crop yield using Census of Agriculture data for each major crop type from 1969 to 2017 .

found in most counties until 2012, but area harvested declined in 2017. There was not a consistent trend in growth or decline of rice cropland. Using census data alone, it is impossible to know if changes in crop area are related to access to irrigation. There are a multitude of factors at play in land use choices, among them are commodities prices, suitability of soil, previous land use history, and other socioeconomic and environmental factors (Miller and Plantinga, 1999; Caldas et al., 2016). However, access to irrigation may have allowed planting more water-demanding crops like corn (Smidt et al., 2016), especially when corn prices surged in the mid to late 2000s due to the increase in corn-based ethanol production in the United States (Welch et al., 2010; Gardebroek and Hernandez, 2013). Measured irrigation rates in Mississippi are highest for rice, followed by corn, soybean, and cotton (9,200, 3,100, 2,800, and 1,800 m^3 ha^{-1}, respectively; Massey et al., 2017).

While the total amount of cotton and soybean cropland may have decreased, irrigation of these crops has increased consistently throughout the 1980s and 1990s, so that in 2017 the majority of all cotton and soybean harvested within the region was irrigated (**Figure 3C**). This trend is consistent on a county basis as well; the proportion of soybean area irrigated has increased in all counties. In addition, the harvested mass of soybeans has also increased, even in counties with decreasing planted soybean area, suggesting that irrigation is increasing soybean yields while using less area. In general, yields have increased for all crops over the time period studied (**Figure 3D**). The greatest increases were seen in corn and grain sorghum, which had 400% and 330% increases in yield from 1969 to 2017, respectively. Winter wheat and cotton yield both increased by about 100%, and soybean yield increased approximately 175%. Rice had the lowest increase in yield, increasing 73% from 1978 to 2017. In addition to trends in increasing irrigated lands, observed increased yield is likely due to a combination of additional factors, including advanced agricultural technologies, genetically modified seed, and changes in planting density (Specht et al., 1999).

It is challenging to tease apart the effect of irrigation on crop yields using census data, as yield and irrigation both increased over time (**Figures 3C,D**, **4A**). Available USDA survey data, which report yield by irrigation status, help provide the missing link between irrigated status and increased yield. In AR, LA, and MS irrigated cotton yield was about 214, 170, and 225 kg/ha higher, respectively, than non-irrigated on average for statewide-reported values (**Figure 4**). According to these values, irrigation accounts for 23 to 29% of the yield increase. County-level survey data showed similar trends, with an approximate 196–220 kg/ha difference between irrigated and non-irrigated cropland (**Figure 4B**). Linear trends for irrigated and non-irrigated yields over time seem to have nearly the same slope. This suggests benefits of irrigation for cotton are fixed over time. Yields are not available for other crops of interest in all three states, therefore, we cannot estimate increases in yields of corn or soybeans resulting from increased irrigation in this region. However, the literature demonstrates that irrigation lowers risk of crop damage or failure due to lack of rainfall during the growing season or drought (Massey et al., 2017) and generally does lead to higher yields of corn, soybeans, and cotton (Grissom et al., 1955; Heatherly et al., 1990; Klocke et al., 2011). In this region the majority of rainfall falls outside of the growing season and irrigation is often necessary to meet crop water needs (Kebede et al., 2014).

USDA Census data provide snapshots of cropping, yield, and irrigation trends throughout time. Together these data provide an overview of long-term regional trends in agriculture; however, it does not provide detailed information on irrigation practices. Therefore, trends in irrigation methods utilized and irrigation rates need to be determined by other means. One clear limitation of utilizing the Census and Survey data are that both datasets rely on self-reported information and values are more likely approximates rather than measured values. Also, the temporal and spatial scale of the Census data can limit analysis of causal relationships amongst variables. Despite the limitations, it was clear that irrigated cropland increased linearly within the MAP region from 1974 to 2017, despite a slight decrease in total cropland. The percentage of major crops like corn, cotton, and soybeans that are irrigated have also clearly increased since 1974 from around 10–20 to 70% in 2017. While the Census data demonstrated that yields of important field crop commodities increased since 1974, most notably a 400% increase in corn yield, they do not provide any co-variables that may help explain the spatial and temporal variation in yield. As we have noted, there are many factors that influence yield and irrigation is only one of them.

Indicators of Hydrologic Alteration Analysis

All the gages studied had at least one statistically significant indicator that demonstrated low flow alteration. The Boeuf River, Bayou Macon, Cache River, and Big Sunflower River at Sunflower sites all demonstrated a large degree of alteration in low flow indicators. Twelve out of the 15 gages demonstrated statistically significant declines in 7 day minimum flows and the base flow index over the period of record (see **Table 2**). The most significant declines in the base flow index were found at Bayou Macon at Eudora, AR; Boeuf River near Girard, LA; and Big Sunflower River at Sunflower, MS. The USGS Streamstats database indicates that the Boeuf and Sunflower River gages have diversions from irrigation and these may play a role in flow regulation. Most AR and LA gages showed a decline in the base flow index over the period of record, and most of these gages also have diversions present for irrigation (see **Table 1**). The 7 day minimum flow also showed significant declines for all stations in LA and most stations in AR and MS.

A subset of the gages were selected to show long-term trends in 7 day minimum flows. In **Figure 5** it is apparent that the median flow and interannual variability decreased at all sites post-1987. At the Boeuf, Tensas, Cache, and Sunflower sites, minimum flow in recent decades is below the range of natural variation that was seen in the earlier portion of the record. The number of days with zero flow represent a more extreme low flow indicator and was only relevant for seven of the gages (**Table 2**). Of the seven, five had statistically significant trends including Bayou Macon near Delhi, LA, with a slope of 1.2, and Cache River near Egypt, AR, with a slope of 0.4 (**Table 2**). Increasing days with zero flow can be detrimental to the aquatic ecosystem and to animals that rely on the rivers as a water source.

Flow Duration Curve Analysis

Comparison of flow duration curves before and after 1987 indicated that the degree of flow alteration varied across sites and seasons (**Figure 6**). We observed significant declines in

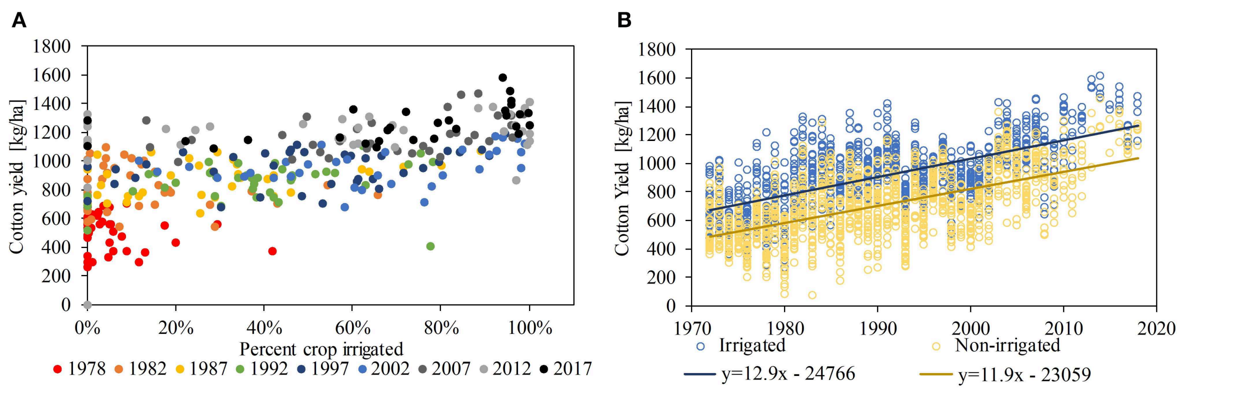


FIGURE 4 | County/Parish cotton yield trends: **(A)** cotton yield vs. percent crop irrigated from the Census of Agriculture, different years are represented by different color dots; **(B)** cotton yield vs. time for irrigated (blue) and non-irrigated (yellow) cropland from the survey dataset, the linear trend lines for each dataset are also shown.

TABLE 2 | Statistics related to the Indicators of Hydrologic Alteration analysis for each gage; gages are listed from most altered to least altered with respect to the slope of the base flow index.

Gage name	No. of altered indicators	No. of altered indicators related to low flow	Base flow index slope	7 day minimum slope	# of zero days
Bayou Macon at Eudora, AR	15	10	*−0.0037*	**−0.0027**	0
Boeuf River near Girard, LA	53	32	**−0.0024**	**−0.00082**	**0.20**
Big Sunflower River at Sunflower, MS	24	13	**−0.0023**	**−0.0026**	0
Bayou Bartholomew near Jones, LA	8	8	**−0.0021**	**−0.0017**	0.014
Cache River near Cotton Plant, AR	17	5	**−0.0016**	*−0.0015*	0
Bayou Macon near Delhi, LA	30	20	**−0.0015**	**−0.0017**	*1.2*
Cache River at Egypt, AR	28	10	**−0.0012**	**−0.0012**	**0.40**
Bougue Phalia near Leland, MS	8	5	**−0.0011**	**−0.0019**	0
Bayou Bartholomew near McGehee, AR	12	9	**−0.00069**	**−0.00080**	0
Tensas River at Tendal, LA	20	8	**−0.00041**	**−0.00030**	0
Languille River near Colt, AR	5	4	*−0.00038*	*−0.00055*	0.071
Bayou Meto near Lonoke, AR	11	7	*−0.00026*	−0.00018	*0.2552*
Bayou Bartholomew at Garret Bridge, AR	2	1	−0.00014	−0.00030	−0.55
Big Sunflower River near Merigold, MS	4	2	0.00022	0.00037	0
Abiaca Creek near Seven Pines, MS	16	11	0.0012	*−0.0039*	0

Bold indicates p ≤ 0.005 and italics indicates p ≤ 0.05.

flow across most of the flow duration curve for all seasons in the Boeuf River (**Figures 6A–C**). In contrast, strong seasonal patterns were present for the other four rivers. The Tensas, Cache, and Languille all had similar patterns for February flow before and after 1987, while flow for exceedance values between 40 and 100 declined in the Sunflower River post-1987 (**Figures 6D,G,J,M**). During the growing season (June), the Boeuf had lower cms across the entire exceedance probability distribution, whereas the Cache and Languille Rivers demonstrated a higher degree of alteration at lower flows only (exceedance probability > 40) (**Figures 6B,H,K**). During June pumping from either groundwater or surface water for irrigation would have begun and irrigation runoff would comprise some of the streamflow in these systems. All locations showed significant flow alteration in October, after harvest, when cumulative groundwater withdrawals for the year would be greatest from pumping throughout the growing season, and stream flows are naturally lower. The Tensas, Cache and Sunflower sites had higher flows at exceedance probabilities <40 and drastic declines at higher exceedance values since 1987 (**Figures 6F,I,O**). In the Cache River near Egypt, AR and the Boeuf River near Girard, LA both systems have zero flow values about 10% of the time in October after 1987.

Flow duration curves suggest low flow effects were widespread during the growing season and greatest in October when natural patterns of low flow conditions were combined with cumulative effects of groundwater withdrawals for irrigation during the preceding growing season (**Figure 6**). The seasonal patterns present in Tensas, Cache and Languille could indicate that low flow effects were due to surface water withdrawals or that groundwater contributions to streamflow were minimal during February both before and after 1987. Recent groundwater measurements showing the depth to water in the alluvial aquifer within the Cache River basin indicate that levels were likely too low for any groundwater connection in recent years (Arkansas Natural Resources Commission, 2015). The Cache River does not have any known regulations but does have small diversions for rice irrigation (**Table 1**). The Boeuf River station near Girard, LA, does have large diversions for irrigation, which is likely the driving cause of the alteration in the flow duration curves. The Tensas River at Tendal, LA, has small diversions for irrigation upstream from the station. This study was focused on low flow metrics; however, the flow duration curves indicate that high flow events were also altered at some sites due to levee development and stream downgrading, which separate streams from floodplains.

Regional Precipitation Trends and Drought Occurrence

It is unlikely that precipitation is a major factor driving observed patterns in hydrologic alteration within our study region. In fact, evaluation of precipitation trends suggest that mean annual precipitation is slightly increasing in the MAP between 7.8 and 17 mm/decade, and during the growing season it is increasing between 3.8 and 18 mm/decade (nClimDiv; Vose et al., 2014). Mean annual precipitation increased from north to south, with the lowest mean annual precipitation in the Missouri bootheel (1,230 mm) and the highest in northeast Louisiana (1,410 mm). With regard to growing season precipitation, east central Arkansas had the lowest (704 mm) and northeast Louisiana had the highest (732 mm). In general, regional precipitation data showed minimal differences over the time period analyzed and suggest changes in stream minimum flows are more likely associated with groundwater or surface water withdrawals, rather than changing climatic patterns. These findings are similar to

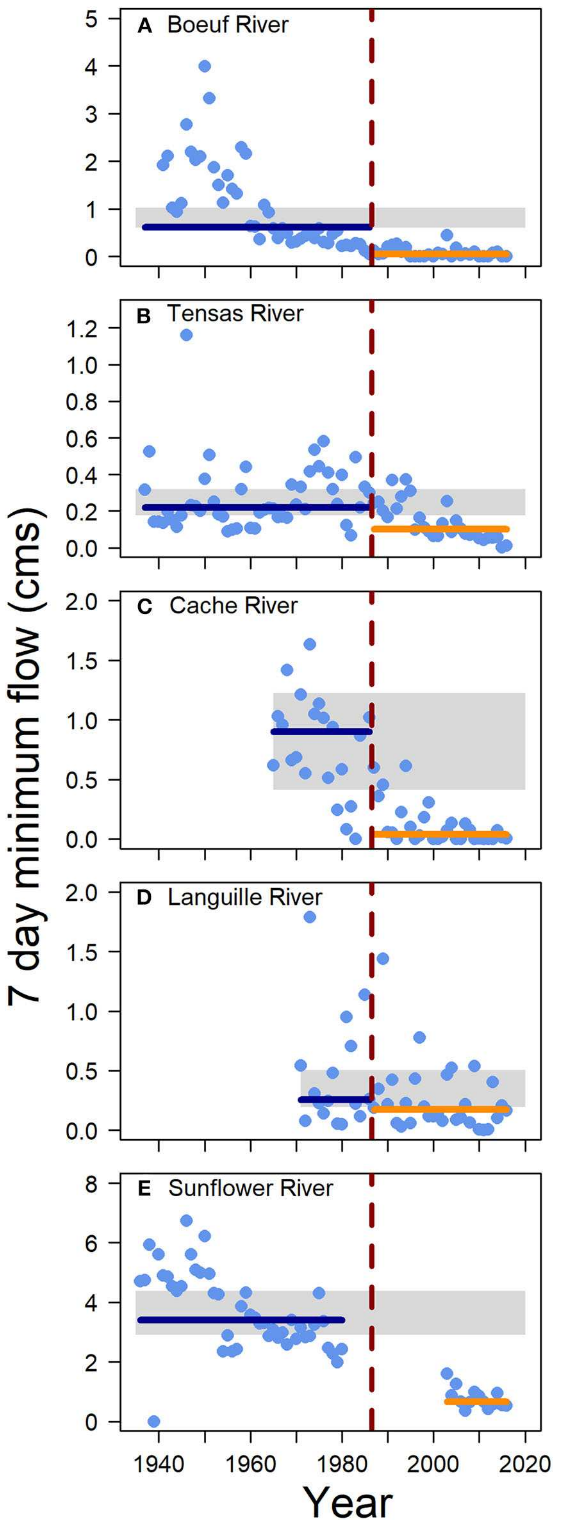


FIGURE 5 | Seven day minimum flow for all available years of data for **(A)** Boeuf River near Girard, **(B)** Tensas River near Tendal, **(C)** Cache River near Egypt, **(D)** Languille River at Colt, and **(E)** the Sunflower River at Sunflower gages; dark blue line is the median of values pre-1987 and the orange line is the median of values after and including 1987, gray bar represents the range of natural variability in the data pre-1987.

what Killian et al. (2019) also found in Mississippi. With respect to drought, the monthly PDSI scores indicate that droughts occurred in the MAP region in 1963, 1967, 1986, 2000, and 2010–2012. The longest drought occurred in 2010 immediately following some of the wettest months in the record, which occurred in late 2009. The occurrence of drought also does not seem to be driving the trends in streamflow, which at several locations show consistent decreases in flow regardless of drought conditions (e.g., Boeuf River, Tensas River, Cache River, and Sunflower River). However, drought may further stress systems if occurring during periods when flow is already below normal or cause extreme low flows (i.e., zero flow). Irrigation rates are also likely to increase during a drought, which may further drawdown groundwater and surface water bodies, but at the same time could contribute return flow to regional streams and rivers. Therefore, there are both positive and negative feedbacks between climate, irrigation, and streamflow.

Evidence of Groundwater Decline

USGS groundwater reports indicate there have been significant declines in groundwater levels across the AR counties examined in this study. In the Boeuf-Tensas area there was an average change of −1.42 m over the period from 2004 to 2014 (Arkansas Natural Resources Commission, 2015). Similarly, 43 out of 50 wells analyzed showed declines, and 40 of these wells had average declines >0.3 m per year. In the St. Francis study area, there was an average decline in groundwater levels of 0.68 m. The counties in the St. Francis study area are upstream of the Languille River near Colt, AR, which showed evidence of hydrologic alteration related to low flow. Crowley's Ridge divides the St. Francis area (eastern side) from the Cache study area (western area).

The Cache area has much steeper declines in groundwater levels, with an average change of −1.5 m between 2004 and 2014. The Cache area has been continuously designated as a critical ground water area since 2009. Surface interpolations of the alluvial aquifer show a cone of depression occurring in the Cache River basin, with the deepest portion 29 to 44 m below surface. Comparatively, in the areas outside of the cones of depression, depth to water is 0 to 13 m. Without knowledge of riverbed material and infiltration capability, it is not possible to know the extent that depth to groundwater affects surface water resources in the declining areas, although modeling studies and data collected in the region do suggest rivers are losing water to recharge the alluvial aquifer (Schrader, 2010; Pugh and Westerman, 2014).

In MS, a cone of depression has developed in the alluvial aquifer in the center of the delta region below the Sunflower River. This area is key to agricultural production in MS. Groundwater withdrawals have been associated with streamflow depletion in the Sunflower River at Sunflower and the Bogue Phalia River near Leland gages (Barlow and Clark, 2011; Barlow and Leake, 2012). Similarly, a recent study has found significant reductions in baseflow that correlate with areas of extensive groundwater declines in the MAP (Killian et al., 2019).

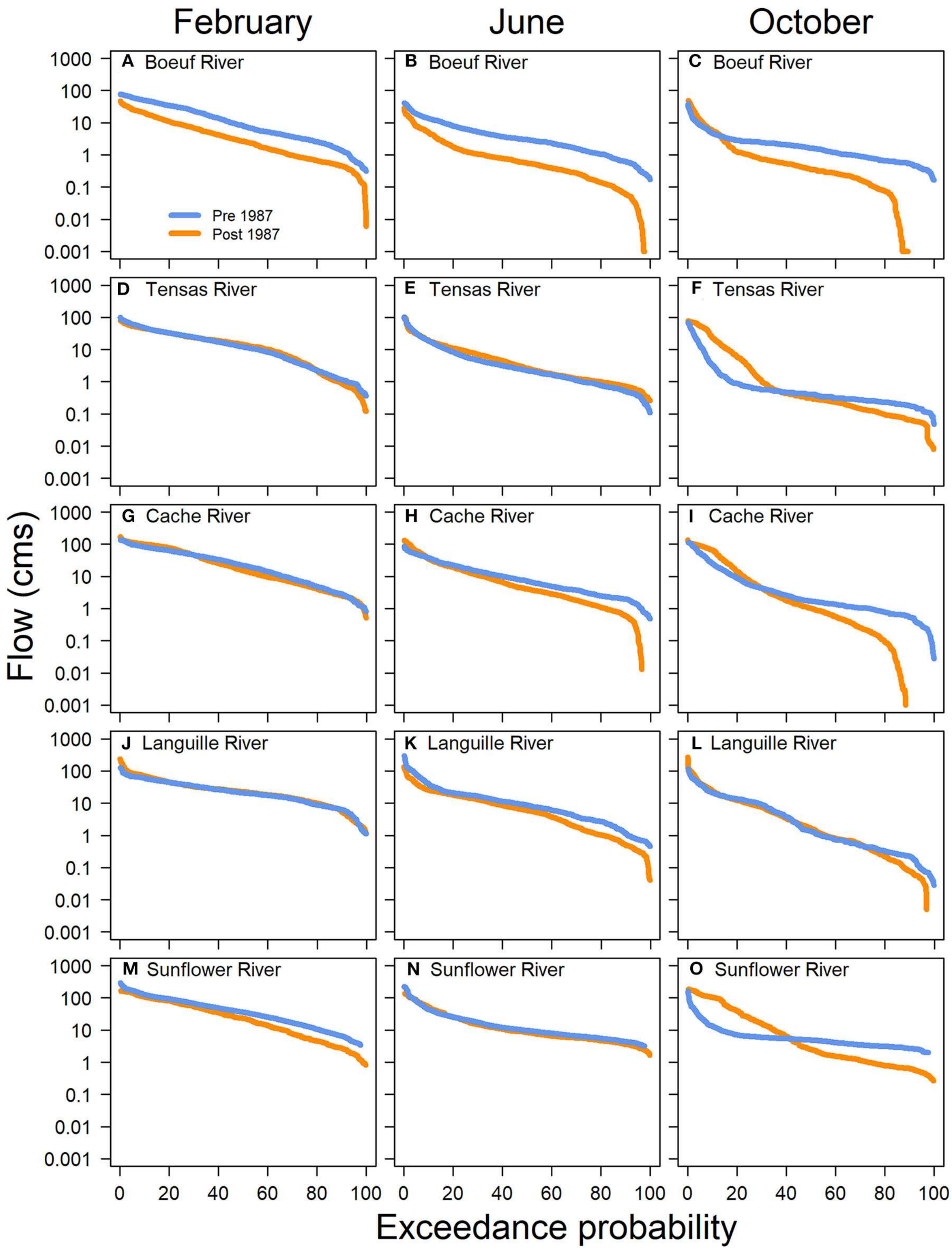


FIGURE 6 | Flow duration curves plotted as flow vs. exceedance probability for February, June, and October flow values for the **(A–C)** Boeuf River near Girard, **(D–F)** Tensas River near Tendal, **(G–I)** Cache River near Egypt, **(J–L)** Languille River at Colt, and **(M–O)** the Sunflower River at Sunflower gages. The blue lines represent flows before 1987 and the orange lines represent flow after 1987.

Groundwater withdrawals in northeastern LA have also increased over time in most of the studied parishes for use in rice and other crop irrigation (White, 2019a,b,c,d,e,f,g). Modeling simulations show decreasing groundwater levels in these parishes over the period from 2004 to 2016 (Karakullukcu, 2018).

As groundwater supplies reach critical levels from increased irrigation demand, solutions such as improved irrigation efficiency, increased use of on-farm reservoirs and/or irrigation tailwater recovery systems, and managed aquifer recharge, have all been suggested to help slow or reverse the decline of groundwater levels in the MRVAA (Barlow and Clark, 2011; Reba et al., 2017). On-farm reservoirs and tailwater recovery systems have been used in eastern AR for several decades (Yaeger et al., 2018) and increasingly in Mississippi since 2014 (Prince Czarnecki et al., 2016; Brock et al., 2019). While these practices may reduce groundwater withdrawals, it is unclear if they will be sufficient to improve groundwater levels (Barlow and Clark, 2011).

Uncertainties

This study suggests, as others have also indicated, that surface water flow signatures are changing in the MAP (Killian et al., 2019). These changes coincide with increases in irrigated cropland area throughout the region and declines in groundwater levels. Several of the gages studied have known diversions for irrigation (**Table 1**), and for others it is unknown. Landowners in AR have the rights to riparian reasonable use if their land touches a water body. Consequently, permission or a permit are not required from the government for a riparian owner to use surface water (Evett et al., 2003). Therefore, it is challenging to know how much surface water is utilized for irrigation and to tease apart the proportion of flow alteration due to surface withdrawals vs. groundwater decline and leaking streambeds. Published irrigation water withdrawals from 2005 indicate that most recorded irrigation withdrawals are from groundwater. Yet, AR counties Lonoke, Desha, Chicot, and Arkansas record 22–42% of total irrigation water withdrawals from surface water sources (Holland, 2007).

Louisiana parishes studied also indicate surface water as part of total irrigated water use. Surface water bodies utilized include Bayou Macon, the Boeuf River, and the Tensas River. However, the proportion of surface water used is ∼10 to 20% of total water usage for irrigation (White, 2019a-g). In MS, declining baseflow at the Big Sunflower River at Sunflower has been linked to the groundwater cone of depression in the central MS Delta (Killian et al., 2019). However, as Killian et al. (2019) mentions, the region's natural hydrology is heavily modified due to agriculture and streamflow-control structures; therefore, results need to be interpreted cautiously. Similarly, indicators of hydrologic alteration suggest that streamflow has been altered in many locations, but there is not enough evidence to delineate linkages between low flow alteration and groundwater/surface water withdrawals for agriculture. To reduce uncertainties, more data collection about streambed properties, groundwater movement, and surface water withdrawals would improve understanding of the driving forces of streamflow alteration.

CONCLUSIONS

The MAP is a rich agricultural landscape that supports a strong regional economy. Data compiled from the Census of Agriculture suggest total cropland area has decreased, yet productivity has increased due to increasing yields and intensification of agricultural production. Irrigation is one of the many factors that lead to high yields and intense production in this region. Data compiled from the Census of Agriculture indicate irrigated cropland has increased drastically from 1969 to 2017. Water use reports indicate both groundwater and surface water are utilized for irrigation in this region; however, groundwater is the predominant water source. Reliance on groundwater from the alluvial aquifer has led to cones of depression in AR and MS, as well as declining levels in LA. Stream gage records from stations overlying the alluvial aquifer show evidence of hydrologic alteration over time, including declining base flow index, increased number of low flow events and decreasing low flow values, as well as altered flow duration curves during the growing and harvest seasons. Streamflow alteration seems to be a function of both ground and surface water irrigation.

The coupled trends in groundwater decline and streamflow alteration present significant future challenges for sustaining agricultural production in the region, while also protecting natural resources and associated biodiversity. As more producers utilize ground and surface water resources for irrigation to increase production, water resources will likely continue to decline. Efforts are already underway in the MAP region to study the alluvial aquifer, to quantify irrigation withdrawals, and to examine options for reducing withdrawals through irrigation efficiency or increased recharge to the aquifer (Barlow and Clark, 2011). Further studies in Mississippi and Arkansas are exploring infiltration basins and tailwater recovery reservoirs to either increase groundwater recharge or to utilize captured runoff for irrigation (Reba et al., 2017; Yaeger et al., 2018; Brock et al., 2019). Future studies in the region will examine utilizing excess surface water during the winter and spring to enhance aquifer recharge or to optimize storage for irrigation. Slowing, or even reversing, the decline in groundwater levels will require a combination of creative engineering and agronomic water management solutions.

Our results suggest that there is an inherent trade off in increasing production via irrigation as declining water resources will result in less profit for future generations and stress to aquatic ecosystems that may be irreversible. Providing further evidence for environmental impacts of increased irrigation in the region may assist policy makers and decision makers when evaluating strategies to improve water management.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

LY conceived the study, organized databases, performed data analysis, and wrote the first draft of the manuscript. LY, JT, JR, and ML contributed to design of the study. JT prepared figures and wrote sections of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

ACKNOWLEDGMENTS

We acknowledge Jay Rao and Lucas Leeper who assisted in data collection for this project. This work was improved through discussion with Mary Yaeger, and revisions from Matt Moore, Henrique Momm, and two reviewers. This research was a contribution from the Long-Term Agroecosystem Research (LTAR) network. LTAR was supported by the United States Department of Agriculture. Mention of trade names or commercial products in this publication is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the USDA. USDA is an equal opportunity provider and employer.

REFERENCES

Alhassan, M., Lawrence, C. B., Richardson, S., and Pindilli, E. J. (2019). *The Mississippi Alluvial Plain Aquifers: an Engine for Economic Activity: U.S. Geological Survey Fact Sheet, 2019–3003*. U.S. Geological Survey.

Arkansas Natural Resources Commission (2015). *Arkansas Groundwater Protection and Management Report for 2014: A Supplement to the Arkansas Water Plan*.

Barlow, J. R. B., and Clark, B. R. (2011). *Simulation of Water-Use Conservation Scenarios for the Mississippi Delta Using an Existing Regional Groundwater Flow Model. U.S. Geological Survey Scientific Investigation Report 2011-5019*. U.S. Geological Survey

Barlow, P. M., and Leake, S. A. (2012). *Streamflow Depletion by Wells: Understanding and Managing the Effects of Groundwater Pumping on Streamflow*. U.S. Geological Survey Circular. Reston, VA: U.S. Geological Survey.

Bowen, Z. H., Freeman, M. C., and Bovee, K. D. (1998). Evaluation of generalized habitat criteria for assessing impacts of altered flow regimes on warmwater fishes. *Trans. Am. Fish. Soc.* 127, 455–468.

Brock, M. L., Tagert, M. L. M., Paz, J. O., and Yasarer, L. (2019). "Characterizing surface water use for irrigation in the Mississippi Delta," in *Proceedings of the 2019 ASABE Annual International Meeting* (Boston, MA).

Brooks, B. W., Riley, T. M., and Taylor, R. D. (2006). Water quality of effluent-dominated ecosystems: ecotoxicological, hydrological, and management considerations. *Hydrobiologia* 556, 365–379. doi: 10.1007/s10750-004-0189-7

Brunner, P., Cook, P. G., and Simmons, C. T. (2011). Disconnected surface water and groundwater: from theory to practice. *Ground Water* 49, 460–467. doi: 10.1111/j.1745-6584.2010.00752.x

Buchannan, B. P., Auerbach, D. A., McManamay, R. A., Taylor, J. M., Flecker, A. S., Archibald, J. A., et al. (2017). Environmental flows in the context of unconventional natural gas development in the Marcellus Shale. *Ecol. Appl.* 27, 37–55. doi: 10.1002/eap.1425

Butler, J. J. Jr., Zlotnik, V. A., and Tsou, M. S. (2001). Drawdown and stream depletion produced by pumping in the vicinity of a partially penetrating stream. *Ground Water* 39, 651–659. doi: 10.1111/j.1745-6584.2001.tb02354.x

Caldas, M. M., Bergtold, J. S., Peterson, J. M., and Earnhart, D. H. (2016). Land-use choices: the case of conservation reserve program (CRP) re-enrollment in Kansas, USA. *J. Land Use Sci.* 11, 579–594. doi: 10.1080/1747423X.2016.1215563

Carlisle, D. M., Wolock, D. M., and Meador, M. R. (2011). Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. *Front. Ecol. Environ.* 9:53. doi: 10.1890/100053

Census of Agriculture. (1969–2017). *National Agricultural Statistics Service*. Washington, DC: United States Department of Agriculture. Available online at: https://www.nass.usda.gov/AgCensus/index.php (accessed August 10, 2019).

Clark, B. R., Hart, R. M., and Gurdak, J. J. (2011). Groundwater availability of the Mississippi embayment. *US Geological Survey Professional Paper* (Reston, VA).

de Graaf, I. E. M., Gleeson, T., van Beek, L. P. H., Sutanudjaja, E. H., and Bierkens, M. F. P. (2019). Environmental flow limits to global groundwater pumping. *Nature* 574, 90–94. doi: 10.1038/s41586-019-1594-4

Evett, S., Carman, D., and Bucks, D. (2003). "Expansion of irrigation in the Mid-South United States: water allocation and research issues," in *Proceedings of the 2nd International Conference on Irrigation and Drainage, Water for a Sustainable World – Limited Supplies and Expanding Demand* (Phoenix, AZ), 247–260.

Fox, G. A., and Durnford, D. S. (2003). Unsaturated hyporheic zone flow in stream/aquifer conjunctive systems. *Adv. Water Resour.* 26, 989–1000. doi: 10.1016/S0309-1708(03)00087-3

Freeman, M. C., Bowen, Z. H., Bovee, K. D., and Irwin, E. R. (2001). Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes. *Ecol. Appl.* 11, 179–190. doi: 10.1890/1051-0761(2001)011[0179:FAHEOJ]2.0.CO;2

Freeman, M. C., and Marcinek, P. A. (2006). Fish assemblage responses to water withdrawals and water supply reservoirs in Piedmont streams. *Environ. Manage.* 38, 435–450. doi: 10.1007/s00267-005-0169-3

Galbraith, H. S., Spooner, D. E., and Vaughn, C. C. (2010). Synergistic effects of regional climate patterns and local water management on freshwater mussel communities. *Biol. Conserv.* 143, 1175–1183. doi: 10.1016/j.biocon.2010.02.025

Gardebroek, C., and Hernandez, M. A. (2013). Do energy prices stimulate food price volatility? Examining volatility transmission between US oil, ethanol and corn markets. *Energy Econ.* 40, 119–129. doi: 10.1016/j.eneco.2013.06.013

Garvey, J. E., Whiles, M. R., and Streicher, D. (2007). A hierarchical model for oxygen dynamics in streams. *Can. J. Fish. Aquatic Sci.* 64, 1816–1827. doi: 10.1139/f07-144

Golladay, S. W., Gagnon, P., Kearns, M., Battle, J. M., and Hicks, D. W. (2004). Response of freshwater mussel assemblages (Bivalvia: Unionidae) to a record drought in the Gulf Coastal Plain of southwestern Georgia. *J. North Am. Benthol. Soc.* 23, 494–506. doi: 10.1899/0887-3593(2004)023<0494:ROFMAB>2.0.CO;2

Grissom, P., Raney, W. A., and Hogg, P. (1955). *Crop Response to Irrigation in the Yazoo-Mississippi Delta*. Mississippi State College Agricultural Experiment Station.

Hanlon, T. C., Williams, C. E., and Moriarity, W. J. (1998). Species composition of soil seed banks of Allegheny Plateau riparian forests. *J. Torrey Bot. Soc.* 125, 199–215. doi: 10.2307/2997218

Hauer, C., Unfer, G., Holzmann, H., Schmutz, S., and Habersack, H. (2013). The impact of discharge change on physical instream habitats and its response to river morphology. *Clim. Change* 116, 827–850. doi: 10.1007/s10584-012-0507-4

Heatherly, L. G., Wesley, R. A., and Elmore, C. D. (1990). Corn, sorghum, and soybean response to irrigation in the Mississippi River Alluvial Plain. *Crop Sci.* 30, 665–672. doi: 10.2135/cropsci1990.0011183X003000030038x

Holland, T. W. (2007). *Water Use in Arkansas, 2005*. Scientific Investigations Report 2007–5241, U.S. Geological Survey.

Hunt, B. (1999). Unsteady stream depletion from ground water pumping. *Ground Water* 37, 98–102. doi: 10.1111/j.1745-6584.1999.tb00962.x

Karakullukcu, R. (2018). *Groundwater modeling for Mississippi river alluvial aquifer of Northeastern Louisiana* (Master's thesis). Louisiana State University, Baton Rouge, LA, United States.

Kebede, H., Fisher, D. K., Sui, R., and Reddy, K. N. (2014). Irrigation methods and scheduling in the Delta region of Mississippi: current status and strategies to improve irrigation efficiency. *Am. J. Plant Sci.* 5, 2917–2928. doi: 10.4236/ajps.2014.520307

Kennard, M. J., Mackay, S. J., Pusey, B. J., Olden, J. D., and Marsh, N. (2010). Quantifying uncertainty in estimation of hydrologic metrics for ecohydrological studies. *River Res. Appl.* 26, 137–156. doi: 10.1002/rra.1249

Kennen, J. G., Riskin, M. L., and Charles, E. G. (2014). Effects of streamflow reductions on aquatic macroinvertebrates: linking groundwater withdrawals and assemblage response in southern New Jersey streams, USA. *Hydrol. Sci. J.* 59, 545–561. doi: 10.1080/02626667.2013.877139

Killian, C. D., Asquith, W. H., Barlow, J. R. B., Bent, G. C., Kress, W. H., Barlow, P. M., et al. (2019). Characterizing groundwater and surface-water interaction using hydrograph-separation techniques and groundwater-level data throughout the Mississippi Delta, USA. *Hydrogeol. J.* 27, 2167–2179. doi: 10.1007/s10040-019-01981-6

Klocke, N. L., Currie, R. S., Tomsicek, D. J., and Koehn, J. (2011). Corn yield response to deficit irrigation. *Trans. ASABE* 54, 931–940. doi: 10.13031/2013.37118

Kresse, T. M., Hays, P. D., Merriman, K. R., Gillip, J. A., Fugitt, D. T., Spellman, J. L., et al. (2014). *Aquifers of Arkansas – Protection, Management, and Hydrologic and Geochemical Characteristics of Groundwater Resources in Arkansas.* Scientific Investigations Report 2014–5149, U.S. Geological Survey.

Maletic, J. T., and Hutchings, T. B. (1967). "Selection and classification of irrigable land," in: *Irrigation of Agricultural Lands*, eds R. M. Hagan, H. R. Haise, T. W. Edminster (Madison, WI: ASA Agronomy Monograph), 125–173.

Massey, J. H., Stiles, C. M., Epting, J. W., Powers, R. S., Kelly, D. B., Bowling, T. H., et al. (2017). Long-term measurements of agronomic crop irrigation made in the Mississippi delta portion of the lower Mississippi river valley. *Irrigation Sci.* 35, 297–313. doi: 10.1007/s00271-017-0543-y

Maupin, M. A., and Barber, N. L. (2005). *Estimated Withdrawals From Principal Aquifers in the United States*. U.S. Geological Survey Circular. Reston, VA: U.S. Geological Survey.

McCargo, J. W., and Peterson, J. T. (2010). An evaluation of the influence of seasonal base flow and geomorphic stream characteristics on Coastal Plain stream fish assemblages. *Trans. Am. Fish. Soc.* 139, 29–48. doi: 10.1577/T09-036.1

Miller, D. J., and Plantinga, A. J. (1999). Modeling land use decisions with aggregate data. *Am. J. Agric. Econ.* 81, 180–194. doi: 10.2307/1244459

Prince Czarnecki, J. M., Omer, A. R., and Dyer, J. L. (2016). Quantifying capture and use of tailwater recovery systems. *J. Irrig. Drainage Eng.* 143:05016010. doi: 10.1061/(ASCE)IR.1943-4774.0001124

Pugh, A. L., and Westerman, D. A. (2014). Mean annual, seasonal, and monthly precipitation and runoff in Arkansas, 1951-2011. *U. S. Geol. Survey Sci. Invest. Rep.* 49:20145006. doi: 10.3133/sir20145006

Reba, M. L., Massey, J. H., Adviento-Borbe, M. A., Leslie, D., Yaeger, M. A., Anders, M., et al. (2017). Aquifer depletion in the Lower Mississippi River Basin: challenges and solutions. *J. Contemp. Water Res. Educ.* 162, 128–139. doi: 10.1111/j.1936-704X.2017.03264.x

Richter, B. D., Baumgartner, J. V., Powell, J., and Braun, D. P. (1996). A method for assessing hydrologic alteration within ecosystems. *Conserv. Biol.* 10, 1163–1174. doi: 10.1046/j.1523-1739.1996.10041163.x

Schrader, T. P. (2010). Water levels and selected water quality conditions in the Mississippi River Valley alluvial aquifer in eastern Arkansas, 2008. *U. S. Geol. Survey Sci. Invest. Rep.* 71:20105140. doi: 10.3133/sir20105140

Smidt, S. J., Haacker, E. M. K., Kendall, A. D., Deines, J. M., Pei, L., Cotterman, K. A., et al. (2016). Complex water management in modern agriculture: trends in the water-energy-food nexus over the High Plains Aquifer. *Sci. Total Environ.* 566–577, 988–1001. doi: 10.1016/j.scitotenv.2016.05.127

Specht, J. E., Hume, D. J., and Kumudini, S. V. (1999). Soybean yield potential – a genetic and physiological perspective. *Crop Sci.* 39, 1560–1570. doi: 10.2135/cropsci1999.3961560x

The Nature Conservancy (2009). *Indicators of Hydrologic Alteration Version 7.1 User's Manual*.

U.S. Department of Agriculture, National Agricultural Statistics Service. (2019). *Quick Stats. c1971*. Washington, DC. Available online at: https://quickstats.nass.usda.gov/ (accessed August 10, 2019).

U.S. Geological Survey (2016). *StreamStats. c2016*. Available online at: http://streamstats.usgs.gov (accessed February 01, 2018).

Valenti, T. W., Taylor, J. M., Back, J. A., King, R. S., and Brooks, B. W. (2011). Influence of drought and total phosphorus on diel pH in wadeable streams: implications for ecological risk assessment of ionizable contaminants. *Integr. Environ. Assess. Manage.* 7, 636–647. doi: 10.1002/ieam.202

Vose, R. S., Applequist, S., Durre, I., Menne, M. J., Williams, C. N., Fenimore, C., et al. (2014). Improved historical temperature and precipitation time series for U.S. climate divisions. *J. Appl. Meteorol. Climatol.* 53, 1232–1251. doi: 10.1175/JAMC-D-13-0248.1

Welch, H. L., Green, C. T., Rebich, R. A., Barlow, J. R. B., and Hicks, M. B. (2010). *Unintended Consequences of Biofuel Production – The Effects of Large-Scale Crop Conversion on Water Quality and Quantity*. Open-File Report 2010–1229, U.S. Geological Survey.

White, V. E. (2019a). *Water Resources of East Carroll Parish, Louisiana: U. S. Geological Survey Fact Sheet 2019–3014*. U.S. Geological Survey.

White, V. E. (2019b). *Water Resources of Franklin Parish*, *Louisiana: U. S. Geological Survey Fact Sheet 2019–3021*. U.S. Geological Survey.

White, V. E. (2019c). *Water Resources of Madison Parish*, Louisiana: U. S. Geological Survey Fact Sheet 2019–3018. U.S. Geological Survey.

White, V. E. (2019d). *Water Resources of Morehouse Parish*, *Louisiana: U. S. Geological Survey Fact Sheet 2018–3068*. U.S. Geological Survey.

White, V. E. (2019e). *Water Resources of Richland Parish*, *Louisiana: U.S. Geological Survey Fact Sheet 2019–3005*. U.S. Geological Survey.

White, V. E. (2019f). *Water Resources of Tensas Parish, Louisiana: U.S. Geological Survey Fact Sheet 2019–3004*. U.S. Geological Survey.

White, V. E. (2019g). *Water Resources of West Carroll Parish*, *Louisiana: U.S. Geological Survey Fact Sheet 2018–3069*. U.S. Geological Survey.

Williams, C. E., Moriarity, W. J., Walters, G. L., and Hill,. L. (1999). Influence of inundation potential and forest overstory on the ground-layer vegetation of Allegheny Plateau riparian forests. *Am. Midland Nat.* 141, 323–338.

Yaeger, M. A., Massey, J. H., Reba, M. L., and Adviento-Borbe, M., Arlene, A. (2018). Trends in the construction of on-farm irrigation reservoirs in response to aquifer decline in eastern Arkansas: implications for conjunctive water resource management. *Agric. Water Manage.* 208, 373–383. doi: 10.1016/j.agwat.2018.06.040

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 16 July 2020
doi: 10.3389/fsufs.2020.00096

Potential Farm-Level Economic Impact of Incorporating Environmental Costs Into Nitrogen Decision Making: A Case Study in Canadian Corn Production

Kamaljit Banger, Joshua Nasielski, Ken Janovicek, John Sulik and Bill Deen*

Plant Agriculture, University of Guelph, Guelph, ON, Canada

OPEN ACCESS

Edited by:
Fardausi Akhter,
Agriculture and Agri-Food Canada (AAFC), Canada

Reviewed by:
William May,
Agriculture and Agri-Food Canada (AAFC), Canada
Emma Louise Burns,
Australian National University, Australia

***Correspondence:**
Kamaljit Banger
kamal.banger@gmail.com

Specialty section:
This article was submitted to Agroecology and Ecosystem Services, a section of the journal Frontiers in Sustainable Food Systems

Received: 24 October 2019
Accepted: 29 May 2020
Published: 16 July 2020

Citation:
Banger K, Nasielski J, Janovicek K, Sulik J and Deen B (2020) Potential Farm-Level Economic Impact of Incorporating Environmental Costs Into Nitrogen Decision Making: A Case Study in Canadian Corn Production. Front. Sustain. Food Syst. 4:96. doi: 10.3389/fsufs.2020.00096

Corn yield response to nitrogen (N) rates typically follows a flat plateau polynomial function with a relatively "flat" region on either side of the Economically Optimum N Rate (EONR). This flat region indicates that a wide range of N rates can approximate the maximum returns achieved at the EONR. To avoid yield penalties due to N stress, farmers tend to over- apply N which results in complex tension between farmers and other stakeholder groups. Using 10-years field data (2009–2018) from Elora, Ontario, we estimated the magnitude of cost to farmers if optimal N rate is based on both economic and environmental costs, and assessed whether incorporating environmental costs into optimum N rate increases profit variability. A cropping system model (DeNitrification and Decomposition model, DNDC) was calibrated and validated for corn yield and environmental N losses against five N rates (30, 58, 87, 145, and 218 kg N ha^{-1}) during 2009–2018. Our results suggest that N rates could vary by 46–91 kg N ha^{-1} around the EONR without reducing profits substantially (<\$25 ha^{-1} of maximum profits) during 2009–2018. When environmental costs were accounted for, environmentally optimal N rate was reduced by 11–54 kg N ha^{-1} (7–31% of EONR) with maximum reductions in N rates occurred in an extremely dry (2012) year. With conservative estimates of the environmental costs of N loss, our study suggests that the environmental benefits accrued at environmentally optimal N rates are 2–4-folds' greater than the reduction in net farmer income. This indicates that the environmental returns to policies which compensate farmers for applying environmentally optimal N are large. Results of this study further suggest that farmers need to adjust N rates depending on the weather in a growing season.

Keywords: corn, EONR: economically optimum N rate, environmental cost, nutrient recommendation, climate extremes

INTRODUCTION

Improving nitrogen (N) management in the North American corn belt is essential for increasing food production and reducing environmental degradation (Frink et al., 1999; Ladha et al., 2005; Ewing and Runck, 2015). In corn production, <50% of applied N is used by the growing crop thus leaving a remainder vulnerable to loss through leaching or gaseous pathways

(Cassman et al., 2002). For instance, excessive N leaching from fields makes groundwater unfit for human consumption and causes eutrophication in streams, lakes, and coastal oceans (Howarth and Marino, 2006; Anderson et al., 2008; Dodds et al., 2009). Fertilizers are responsible for nitrous oxide (N_2O) emissions, which is a potent greenhouse gas and also plays an important role in stratospheric ozone depletion (IPCC, 2014). Therefore, N management must address the twin challenges that under- N application results in yield penalties while over-application above crop needs causes environmental degradation and unnecessary economic losses.

Currently, farm-level N fertilizer application decisions are primarily driven by crop yields and farm profits, not environmental concerns (Sawyer et al., 2006; N-Calculator, 2020). The Economically Optimal N Rate (EONR), commonly used to estimate N application rates is defined as the N rate where a unit of fertilizer N provides a yield increase equal in value to the cost of the N fertilizer (Sawyer et al., 2006; Morris et al., 2018). While the yield response to N function used to derive EONR implicitly considers crop N demand, soil N supply, and N losses to the environment, EONR estimation does not explicitly considers environmental costs associated with N loss. Conversion of yield response and N fertilizer rate into an economic profit response results in a relationship typically represented by a polynomial function with a relatively "flat" region on either side of the EONR (Rajsic and Weersink, 2008; Cabas et al., 2010). The existence of a flat profit response suggests that actual N rate can deviate above or below the EONR without significantly impacting profitability (Pannell, 2006). The existence of a flat payoff response has two implications for N fertilizer management decisions. Firstly, even though EONR is highly variable (Sogbedji et al., 2001; Derby et al., 2004; Tremblay et al., 2012; Dhital and Raun, 2016), farmers have limited economic incentives to invest in new technologies aimed at more accurately estimating EONR (Liu et al., 2006). Secondly, given the similar economic risks of under-application vs. over-application of N fertilizer relative to EONR, farmers opt for over-application (Rajsic and Weersink, 2008), and, as a result, increase environmental risks whilst also missing an opportunity to realize economic gains. Farmer tendency to over apply relative to EONR results in a complex tension between farmers and other stakeholder groups (Ewing and Runck, 2015). For instance, in Central Iowa and Southern Minnesota, US, corn yields are very high due to better agronomic management and particularly due to high N rates (NAAS, 2015). At the same time, urban residents of this region may pay more than US$4.00 per 1,000 gallons of water so that local water supplies meet U.S. Environmental Protection Agency standards of 10 mg of nitrate-N ($NO_{3}-$) per liter (Powlson et al., 2008). Thus, farmers often feel a social responsibility to adjust N rates and reduce environmental degradation.

Canadian corn production is concentrated in the humid areas of eastern Canada, where monthly precipitation is relatively evenly distributed across the growing season. As such, N losses are possible during both the growing and non-growing season. For example, gaseous N losses such as ammonia volatilization and N_2O emissions accounted for 65 kg N ha^{-1} in the broadcast and 27 kg N ha^{-1} in the injected fertilizer management scenarios when total N rate was 148 kg N ha^{-1} (Drury et al., 2017). At provincial scale in Ontario, Drury et al. (2007) estimated that residual soil N remaining in soil at the end of growing season during 1986–2001 was 30–36 kg N ha^{-1}, most of which is potentially leachable during the fall, winter and early spring prior to planting (Drury et al., 2007). As such, there is normally little residual fertilizer N available for subsequent crops, and fall application of N fertilizer is rare in part due to the high potential of N losses. Commonly in Ontario, N is applied to corn fields as urea and incorporated via some form of tillage prior to planting in the spring. Often, some N (30 kg N ha^{-1} or less) is also applied close to the seed trench during planting, a practice colloquially called "starter" fertilization because this N is positionally available to corn roots as the crops starts growing. Rather than applying the majority of N pre-plant, a growing proportion of Ontario farmers apply N in-season, typically as liquid urea ammonium nitrate that is either injected in the soil profile or streamed onto the soil surface.

However, existing Decision Support Systems (DSS) in the Northern Corn Belt do not currently account for environmental costs associated with N losses due to fertilizer management (Banger et al., 2017). Recently, Morris et al. (2018) highlighted this inability to account for environmental costs as one of the important limitation of existing N management DSS in the US Corn Belt. Some private sector tools such as Adapt-N, Climate FieldView, and Encirca estimate different N loss pathways in response to fertilizer management (Sela et al., 2016; FieldView, 2019) but they only consider N losses to enable estimation of N fertilizer replacement costs. In order to achieve economic profits and ecosystem sustainability, it is critical to account for the ecosystem services hampered by different N loss pathways from a field (Banger et al., 2017). Several researchers have estimated the damages to different ecosystem services per unit of N loss in terms of economic cost (Birch et al., 2011; Compton et al., 2011; Sobota et al., 2015). Accounting for both economic and environmental costs in the DSS could encourage collaborations between farmers and other stakeholders such as municipalities, provincial, and federal regulatory agencies to form practical strategies for overall ecosystem sustainability. Moreover, as long as an "environmentally" optimal N rate is within the range of the flat payoff response around (i.e., above and below) the EONR, incorporating environmental costs may result in only trivial reductions in net farm-level returns.

To develop new N management DSS and facilitate collaboration avenues between farmers and other stakeholders, we estimated EONR and assessed environmental losses in a 10-yr continuous corn experiment in Ontario, Canada. We assessed the magnitude and year to year variations in the environmental cost in response to N rates in Canadian corn production. Therefore, the objectives of this study were to (1) estimate the magnitude of cost to farmers if economic and environmental costs are included into a "environmentally optimal" N rate; and (2) to assess whether incorporating environmental costs into optimum N rate increases profit variability.

TABLE 1 | Experimental design for assessing the effects of nitrogen rate on corn yield and environmental losses during 2009–2018.

Yr	1	2	3	4	5	6	7	8	9	10
2008	57	57	57	57	57	57	57	57	57	57
2009	0	28	57	115	188			——————115——————————		
2010			——————115——————————			115	57	188	28	0
2011	188	115	28	0	57			——————115——————————		
2012			——————115——————————			0	28	57	115	188
2013	115	188	0	28	57			——————115——————————		
2014			——————115——————————			188	115	28	0	57
2015	28	188	0	57	115			——————115——————————		
2016			——————115——————————			57	0	115	188	28
2017	115	57	188	28	0			——————115——————————		
2018			——————115——————————			28	188	0	57	115

In order to neutralize the long-term legacy effects of treatments, a consistent 115 kg N ha^{-1} was applied at sidedress stage following each nitrogen treatment. All plots received 30 kg N ha^{-1} with starter fertilizer applied at planting in addition to the N treatments shown below.

METHODS

The field experiment was located in Elora, Ontario (43°38'31.1" N 80°24'14.8" W) in a tile drained continuous corn system. The soil was silt loam (Albic fluvisol, WRB 2006) with pH of 7.7, in which sand, silt, clay, and soil organic carbon were 32, 48, 20, and 4.5%, respectively. During the study period (2009–2018), the field was managed with fall moldboard plow and spring secondary tillage prior to planting. Approximately, 79,000 plants ha^{-1} maize hybrids (DKC 39-97) seeds were planted on 0.76 m row spacing. Both potassium and phosphorus were applied in the fall prior to plowing, with rates based on provincial soil test recommendations. At planting, 30 kg N ha^{-1} fertilizer (15-15-15-2Zn) was applied in bands 5 cm beside and 5 cm below the seed. Main plots were split into individual treatment plots (82 m^2) and received one of five N rates as urea ammonium nitrate (UAN) side-dressed at the V6 growth stage (Abendroth et al., 2011) at 7 cm soil depth in the center of the rows at 0, 28, 57, 115, and 188 kg N ha^{-1}. In this way, total N rate was of 30 kg N ha^{-1} (N1), 57 kg N ha^{-1} (N2), 87 kg N ha^{-1} (N3), 145 kg N ha^{-1} (N4), 218 kg N ha^{-1} (N5). In general, a consistent N rate application over a long-term can result in strong legacy effects, which may pose difficulties in estimating N rate at EONR. To avoid legacy effects, the five N rate treatments were constantly randomized from year-to year, and every 2nd year was a "reset year"' when treatment plots receive a uniform N application of 145 kg N ha^{-1} (**Table 1**). For analysis of N rate effects on yield and EONR, we did not use 145 kg N ha^{-1} (legacy treatment) after every N rate treatment. In N rate treatments, corn yield, aboveground N uptake, and soil N concentration was recorded each year from all treatments. Further details on the data collection protocols can be obtained from previous publications (Nasielski et al., 2020).

In this study, we used a well-calibrated and validated cropping system model (DeNitrification Decomposition, DNDC v. CAN, version 9.5). During 10-yr time period, DNDC model has been calibrated and validated for local conditions for corn yield, N_2O emissions, and NO_3– leaching losses by previous researchers (Abalos et al., 2016b; Congreves et al., 2016a; Jarecki et al., 2018; Nasielski et al., 2020) A detailed description of the model subcomponents and design can be found in previous publications (Kroebel et al., 2011; Smith et al., 2013; Abalos et al., 2016a; Congreves et al., 2016b; Dutta et al., 2016; Banger et al., 2020). In brief, the model input datasets were developed for Elora experimental research station. Daily weather data (maximum and minimum temperature, solar radiation, precipitation, wind speed, humidity) were obtained from the weather station maintained by Environment Canada on the research station. To initialize the model, a 10-year spin up corn-soy-barley rotation was run prior to the analyzed simulations using actual weather data (1999–2008) collected on-station, to stabilize C:N dynamics of the cropping system. The simulation process was continuous, with the 10 years simulated sequentially without annual reset. The actual dates of planting, harvest, tillage, and fertilization from the field experiment were used in the model for every year, and the "reset year" was simulated but not included in the analysis. Overall, 50 simulations were performed (five N rate treatments for 10 years). In this study, validated version of DNDC was used to assess the impact of five N rate treatments [N1, N2, N3, N4, and N5] yield and N loss pathways in 10 simulated years (2009–2018). Over a 10-yr period, the DNDC estimated daily environmental N losses (N_2O and NO_x gas emissions, NO_3– leaching and NH_3 volatilization) were aggregated to the crop year period (time period from May 1 to 30 April).

For three N loss pathways, we used literature values for environmental costs (**Table S1**). Sobota et al. (2015) reviewed potential damage costs of N ($/kg N) to air, land, and water resources in the US in the early 2000s. All the specific environmental costs were divided into three categories (low, median, and high). For NO_3– leaching included damages to eutrophication and colon cancer. For ammonia volatilization, we used respiratory diseases, changes in carbon sequestration, and loss of biodiversity (**Table S1**). We used only three environmental damages due to N_2O emissions including greenhouse effect, UV exposure to crops, and UV exposure to humans. Our environmental costs for individual N loss pathways were obtained from Europe and the US, particularly

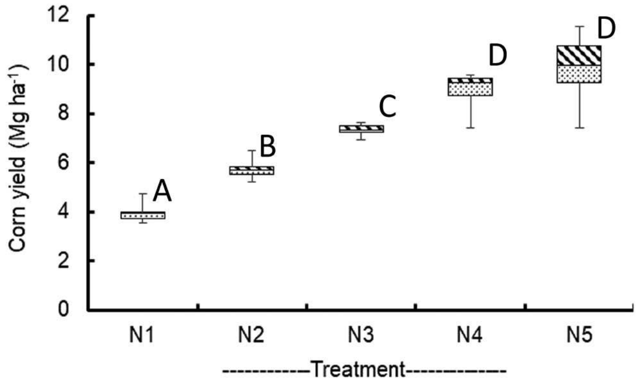


FIGURE 1 | Crop yield in the five nitrogen rate treatments during 2009–2018. In all the treatments, 30 kg N ha^{-1} was applied at planting and rest was sidedressed in the growing season. Total N rates in five treatments were as follows: N1: 30 kg N ha^{-1}; N2: 57 kg N ha^{-1}; N3: 87 kg N ha^{-1}; N4: 145 kg N ha^{-1}; N5: 218 kg N ha^{-1}. Treatments indicated by letters are significantly different from each other with a *p*-value of 0.05 based on a Tukey HSD (Honestly Significant Difference).

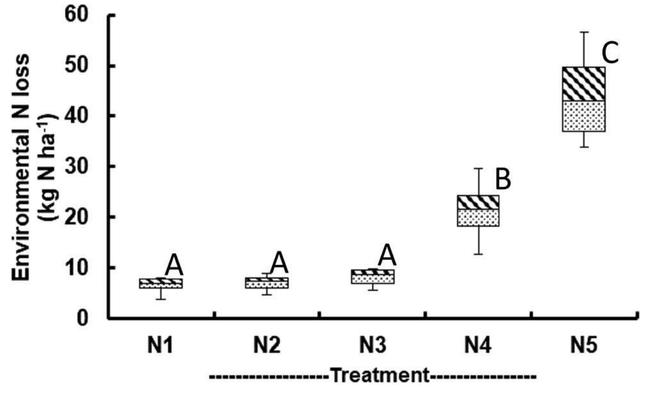


FIGURE 2 | DNDC estimated environmental nitrogen loss (kg N ha^{-1}) from corn production. In the treatments, 30 kg N ha^{-1} was applied at planting and rest was sidedressed in the growing season. Total N rates in five treatments were as follows: N1: 30 kg N ha^{-1}; N2: 57 kg N ha^{-1}; N3: 87 kg N ha^{-1}; N4: 145 kg N ha^{-1}; N5: 218 kg N ha^{-1}. Treatments indicated by letters are significantly different from each other with a *p*-value of 0.05 based on a Tukey HSD (Honestly Significant Difference).

in the Chesapeake Bay watershed (Dodds et al., 2009; Birch et al., 2011; Van Grinsven et al., 2013; Sobota et al., 2015). Given that environmental, recreational, and health costs of N contamination may differ in Canadian ecosystems, we provided conservative estimations for environmental costs associated with N loss pathways. For example, we excluded several environmental costs which were less meaningful for Ontario corn production such as the damages to the coastal ecosystems (Compton et al., 2011). Additionally, we used a low potential environmental cost category for all the N loss pathways (Sobota et al., 2015). While environmental costs of N losses are thus only rough approximations for Ontario corn production, the goals of our study were to highlight twin challenges and identify some policy solutions for reducing environmental degradation while maintaining or improving crop yields.

In this study, EONR was estimated based on the corn yield in five N rates used in the DNDC simulations. We used 10-yr average prices for corn (OMAFRA, 2019) and fertilizer (McEwan, 2019) to estimate EONR. Quadratic plateau yield response to N curves were fitted using the nlin procedure of SAS version 9.4. Constraints were imposed such that the fitted linear coefficient is $\geq$0 and the fitted quadratic coefficient is $\leq$0. These constraints force the fitting of a non-response (plateau) starting at the lowest N rate for cases with an overall tendency for decreasing yields with increasing N rates or a positive linear response for cases that have accelerating rates of yield response with increasing N rates.

To estimate $EONR_{env}$, firstly we calculated the environmental cost associated with three N loss pathways for every unit of N applied. After N cost returns were calculated on a 1 kg N ha^{-1} interval as the difference of the monetary value of corn yield estimated from the quadratic-plateau response equations and the monetary cost of fertilizer N applied. After N cost returns including environmental costs were calculated by also subtracting the monetary environmental N costs estimated on the same 1 kg N ha^{-1} intervals. Yearly estimates for ENOR and $ENOR_{env}$ occurred at the N rates that maximize these 2 after N cost responses each year. In this way, EONR is the maximum economic N rate that takes into account corn price and fertilizer N cost but not environmental costs while $EONR_{env}$ takes into account corn price, N cost as well as environmental costs.

In this study, we used a two-way analysis of variance without replications (ANOVA) to test if two factors including N rate (N1, N2, N3, N4, and N5) and year (10 years from 2009 to 2018) were significantly different. Corn yield, N loss, and environmental cost variables were evaluated at five N rates during 2009–2018. ANOVA test if results are significant overall, but it does not identify where those differences in group means exist. To identify which specific group's means are different, we used Tukey's Honest Significant Difference (HSD) test (Tukey, 1991). This *post-hoc* test identifies pairwise differences among all possible sample means. In this study, we have identified differences between five N rates and 10 yrs at $p < 0.05$ level of significance.

RESULTS

Corn Yield and Environmental Nitrogen Losses

Across the 10-yr period, corn yield ranged from 3.6 to 11.5 Mg ha^{-1} in five N rate treatments (**Figure 1**). In N1, corn yield remained below 4.74 Mg ha^{-1} which increased significantly ($p < 0.05$) in N2 (5.2–6.5 Mg ha^{-1}), N3 (6.9–7.3 Mg ha^{-1}), and N4 and N5 (7.4–11.5 Mg ha^{-1}). Corn yields in N4 and N5 were not statistically different during 2009–2018. When the three DNDC estimated N loss pathways (NO_3– leaching, ammonia volatilization, and N_2O emissions) were aggregated, total N loss

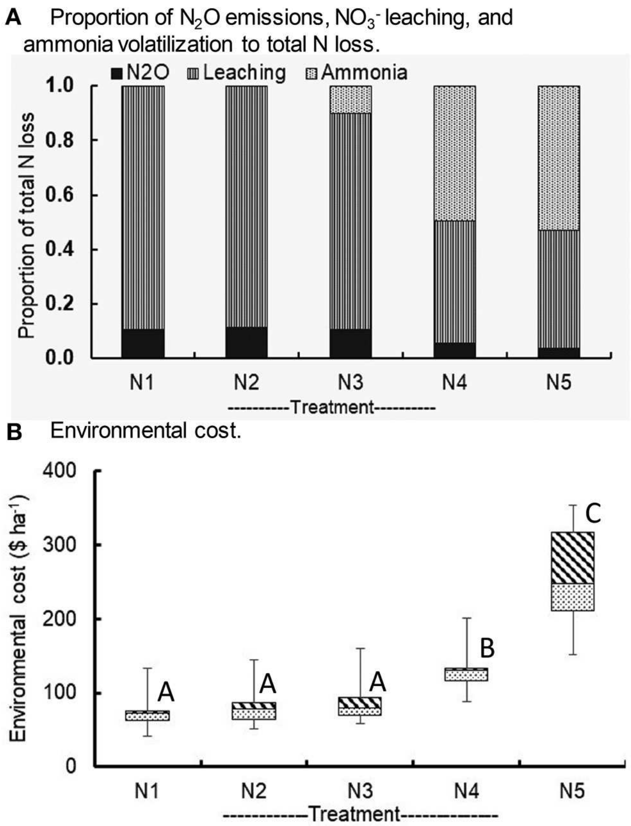


FIGURE 3 | Ten-yr average proportion of three nitrogen loss pathways (N_2O emissions, NO_{3^-} leaching, and ammonia volatilization) and their environmental costs. In the treatments, 30 kg N ha^{-1} was applied at planting and rest was sidedressed in the growing season. Total N rates in five treatments were as follows: N1: 30 kg N ha^{-1}; N2: 57 kg N ha^{-1}; N3: 87 kg N ha^{-1}; N4: 145 kg N ha^{-1}; N5: 218 kg N ha^{-1} Treatments indicated by letters are significantly different from each other with a *p*-value of 0.05 based on a Tukey HSD (Honestly Significant Difference). **(A)** Proportion of N20 emissions, NO_{3^-} leaching, and ammonia volatilization to total *N* loss. **(B)** Environmental cost.

varied from 3.9 kg to 56.5 kg N ha^{-1} across the treatments during 2009–2018 (**Figure 2**). Unlike corn yield, environmental N loss was statistically similar in N1, N2, and N3 (3.9–16.4 kg N ha^{-1}) which increased substantially once fertilizer N rate exceeded 87 kg N ha^{-1} across the years. For instance, N loss was significantly ($p < 0.05$) greater in the N5 (33.8–56.5 kg N ha^{-1}) followed by N4 (12.8–29.6 kg N ha^{-1}) than N1, N2, and N3 treatments (3.9–16.4 kg N ha^{-1}). The model predicted that the years with extreme dry (2012) and extreme wet (2013) growing season had relatively higher environmental N loss than other years (**Figure S1**).

DNDC simulated NO_{3^-} leaching accounted for 44–89% of total N loss in the five N treatments over the 10-yr study period (**Figure 3A**). Although the magnitude of NO_{3^-} leaching, ammonia volatilization, and N_2O emissions increased with N rate (**Figure S2**), their relative proportions to total N loss changed substantially. For instance, the contribution of NO_{3^-} leaching to total N loss was greater in N1, N2, N3 (79–89% of total N loss) and decreased substantially in N4 and N5 (44–45% of total N loss). This occurred because of disproportionally greater increase in ammonia volatilization than NO_{3^-} leaching due to increase N rates. For instance, ammonia volatilization losses were 0 in N1 and increased to 50–53% of total N loss in N4 and N5. Ten-yr average N_2O emissions were smaller (4–11% of total N loss) than NO_{3^-} leaching and ammonia volatilization losses during 2009–2018. Environmental cost associated with the NO_{3^-} leaching, ammonia volatilization, and N_2O emissions was significantly ($p < 0.05$) greater in N5 ($152–353 ha^{-1}) followed by N4 ($88–201 ha^{-1}) than other treatments ($41–160 ha^{-1}) during 2009–2018 (**Figure 3B**).

Effects of Accounting for Environmental Costs on Corn Yield

Over 10-yrs, EONR varied from 151 to 218 kg N ha^{-1}; averaging 189 kg N ha^{-1} (**Figure 4**). In 4 years (2010, 2013, 2014, and 2018), EONR was >189 kg N ha^{-1}, while EONR ranged from 151 to 188 kg N ha^{-1} in other years. Farm-level profits at EONR ranged from $1,281 to $2,391 ha^{-1} during the study period. EONR and economic returns were not significantly correlated since the years with greater EONR did not always result in high economic profits (**Figure 4**). Our results suggest that farmers had a wide range of flexibility in adjusting N rates without significantly reducing economic profits during 2009–2018. For instance, N rate between 122 and 213 kg N ha^{-1} achieved economic profits within $25 ha^{-1} of maximum profit range in 2012. In other years, there was a flexibility of adjusting N rates by 46–77 kg N ha^{-1} from EONR within the economic threshold of $25 ha^{-1} (**Figure 4**). The second highest environmental N loss occurred in 2013 (12.5–52.8 kg N ha^{-1}) which had 9–58% higher May-August cumulative rainfall than other years during 2009–2018.

When environmental cost was incorporated into the estimate of optimal N rate, "environmentally optimal N rate" ($EONR_{env}$) ranged from 115 to 192 kg N ha^{-1}, representing a reduction of 11–54 kg N ha^{-1} relative to EONR (**Table 2**). The yearly N rate reductions in $EONR_{env}$ were 7–31% of EONR. The highest reduction in the N-rate occurred in 2012, which received 77% lower May-August cumulative rainfall (498 mm) compared to the 30-yr long-term average (**Figure S1**). In contrast, the second most reduction in the $EONR_{env}$ occurred in 2013, which had 9–58% higher May-August cumulative rainfall than other years during 2009–2018. Reduced N rate at $EONR_{env}$ corresponded with reductions in corn yields. Relative to EONR, corn yields at $EONR_{env}$ were reduced by 1–7% (**Table 2**). Reductions in N rate to account for environmental costs, reduced farm level economic profits by 0.4–2.8% (a net reduction of $7–66 ha^{-1}) relative to profits at N rates associated with EONR during 2009–2018. Although farm level economic profits were reduced at $EONR_{env}$, the reduction in environmental costs at $EONR_{env}$ were 2–4-folds greater than the farm level economic losses during 2009–2018 (**Table 2**).

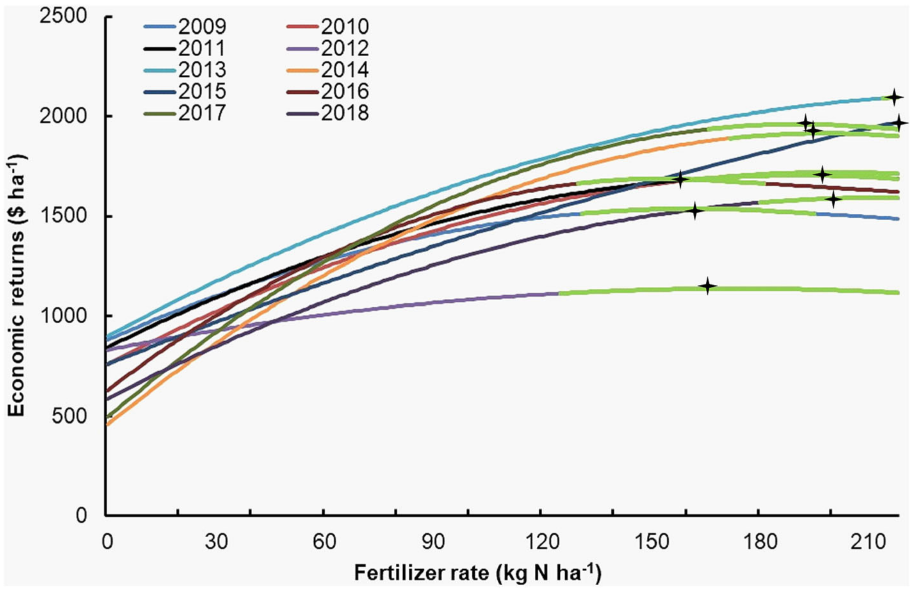


FIGURE 4 | Relationship between net returns to nitrogen costs and nitrogen rates during 2009–2018. The underlying yield responses were fitted using a quadratic-plateau model. After *N* cost return response was calculated as the value of com estimated from quadratic plateau yield response equations subtract the cost of N required to attain the estimated yield. Green colored line indicates the threshold within 25$ ha^{-1} of economic profits. Star indicates the maximum economic rate of nitrogen fertilizer.

DISCUSSION

Our results from a 10 year data set where soil type and management were held constant strengthens previous observations that corn N response and resulting EONR is highly variable due to the complex interactions in crop growth, weather variability, and environmental losses (Vetsch and Randall, 2004; Xie et al., 2013; Morris et al., 2018). Although EONR is highly variable our analysis also suggests that failure to apply exactly at EONR has relatively low farm level economic impact in most years. Conversion of N yield responses to profit responses demonstrates that in any given year, N rates can vary substantially from EONR (46–91 kg N ha^{-1}) without significant reductions in farm level economic profits (<$25 ha^{-1} of maximum economic profits) (**Figure 4**). These results correspond with other studies demonstrating a flat profit response and limited impact on profit when actual N rate deviates from EONR (Pannell, 2006; Rajsic and Weersink, 2008; Cabas et al., 2010). The uncertainty in EONR across years and existence of flat profit functions within each year has two implications. First, it reduces the incentive for farmers to adopt strategies to identify EONR and apply at that rate. **Figure 4** demonstrates that accurate prediction of EONR was not actually required in 9 of 10 years in which a consistent application of 180 kg N ha^{-1} fell within $25 ha^{-1} of maximum farm-level profit threshold plateau (**Figure 4**). Second, as has been shown by Rajsic and Weersink (2008), it may result in farmers having a tendency to over- apply N fertilizers to ensure there is sufficient N for years with unexpectedly high EONR. While farmers respond to the existence of flat profit functions by increasing N rate, other members of society would like to see N rate reduced below EONR to reduce environmental costs associated with N application. To address this twin challenge of simultaneously improving food production and reducing environmental degradation, using a 10-yr corn experiment we estimated an environmentally optimal N rate, $EONR_{env}$ that accounts for environmental costs of N fertilizer application. To the best of our knowledge, very limited research has been conducted to assess the environmental cost associated with N management in the U.S. (Compton et al., 2011; Sobota et al., 2015), and no such study is available for Canadian corn production. Therefore, it is very difficult to compare the results of this study with previous findings.

Our results suggested when N rate was optimized to also consider environmental costs, N rate reductions of 7–31% (11–54 kg N ha^{-1}) compared to EONR were required (**Table 2**). As a consequence of these N rate adjustments, environmental costs due to N fertilizer application were reduced by $13–177 ha^{-1} during 2009–2018. In Iowa and New York, Sela et al. (2016) have shown that Adapt-N estimated N rates were 34% lower (53 and 31 kg ha^{-1}) than farmer applied N rates, which also reduced environmental loss by 38% (28 kg ha^{-1}). In 2004–2008, a sensor based N applications were able to achieve a net reduction of 16 kg N ha^{-1} compared to grower selected rates in 55 on-farm trials (Scharf et al., 2011). Our research is fundamentally different from these studies. Unlike previous studies which compare improved estimates of N fertilizer application rates against N rates selected by the farmers, we have estimated changes in N rates when environmental costs are accounted as farm inputs. Our study assessed magnitude of cost to farmers if environmentally optimal N rate is based on both economic and environmental costs.

Although reduction in N rates associated with $EONR_{env}$ had significant environmental benefits, it caused farm-level economic

TABLE 2 | Reduction in nitrogen rates, yield, farm level economic loss, and environmental costs at $EONR_{env}$ relative to EONR.

Yr	*N*-rate	Yield	Farm-level economic loss	Environmental cost
	(kg N ha^{-1})	(Mg ha^{-1})	———($ ha^{-1})———	
2009	16 (10.3)	0.15 (1.6)	9 (0.5)	32 (1.8)
2010	30 (14.9)	0.32 (2.9)	23 (1.2)	47 (2.4)
2011	20 (10.7)	0.19 (1.8)	11 (0.6)	23 (1.2)
2012	53 (31.4)	0.52 (7.0)	32 (2.5)	141 (11.0)
2013	54 (22.0)	0.70 (5.2)	66 (2.8)	177 (7.4)
2014	13 (6.7)	0.13 (1.1)	8 (0.3)	15 (0.7)
2015	42 (22.5)	0.51 (5.9)	29 (2.0)	117 (7.9)
2016	11 (7.6)	0.11 (1.1)	7 (0.4)	13 (0.7)
2017	28 (31)	0.37 (3.0)	35 (1.6)	73 (3.3)
2018	19 (9.0)	0.18 (1.7)	9 (0.5)	18 (1.0)

Numbers in brackets represent the relative change (%) at $EONR_{env}$ relative to EONR.

losses ranging from \$7 to \$66 ha^{-1} with an average value of \$26 ha^{-1} during 2009–2018 (**Table 2**). Based on our analysis, incorporation of environmental costs in 4 out of 10 years caused reductions of >\$25 ha^{-1} (**Table S2**). More importantly, our results suggest that the greatest reductions in farm-level economic profits occurred in an extremely dry and an extremely wet year. For instance, when rainfall in May-August was lower than normal in 2012, $EONR_{env}$ was 31% lower than EONR (**Table 2**). On the other hand, $EONR_{env}$ shifted much below EONR due to greater rainfall in May-August which resulted substantial environmental N losses. It should be noted that our analysis compares farm level costs at EONR to $EONR_{env}$. But as was already previously discussed, farmers may have a tendency to over-apply N given the uncertainty in EONR over the years. As a consequence, our analysis may actually overestimate farm level costs of $EONR_{env}$. But if farmers do tend to apply above EONR this also means a reduction in environmental costs, as well as yield reductions, associated with $EONR_{env}$ are underestimated in our analysis.

Using data from a field experiment at Elora, our results suggest that tensions between farmers and other stakeholders are inevitable if $EONR_{env}$ is. In 4 out of 10 years the farm level cost would not be trivial. These tensions stem from the fundamental differences in the philosophies on how various stakeholders view farm profits and environmental conditions, and their decision making around N use reflects these preferences. From a farmer's standpoint, it is challenging to adopt environmentally optimal N rates as farmers desire to maximize profits (Ewing and Runck, 2015), although it may not be sustainable in the long-term. For policy makers, it is difficult to regulate fertilizer use. Therefore, it is increasingly of policy concern because better management practices (BMPs) are voluntarily adopted by corn growers of North America (Tomer et al., 2013). We believe that positive outcome oriented agricultural policies should engage different stakeholders such as farmers, municipalities, consultants, and policy makers. We have several encouraging examples across the globe where farmers have collaborated with other stakeholders to reduce nutrient pollution in a watershed. For instance, in New Zealand, a community-based audited self-management approach has been successfully implemented. In this context, farmers and regulatory body are working together to improve the quality and quantity of shared local water sources in a watershed (Holley, 2015). Farmers and stakeholders collaborate to assess N carrying capacity and explore ways to achieve a specified environmental goal, while independent third parties verify the goals. Our results have shown that if farmers are compensated for economic losses to adjust N rates, environmental benefits to the society would be 6-fold that of the cost of compensation to farmers (**Table 2**). The US Department of Agriculture's (USDA) has introduced a practice standard for nutrient management that incentivizes farmers to use an adaptive management approach. In Canada, federal, provincial, and local governments can be involved in the cost sharing of environmental initiatives such as Environmental Farm Plan, Carbon Clean Water Act, and Carbon Credits. Our study highlights that the concept of environmental cost will help set goals in light of political, economic, and social support. It would also involve standardizing methods to estimate environmental costs in Canadian agroecosystems.

The outcome of this research has an imperative implication for developing new N management DSS. In the North American Corn Belt, the majority of the farmers apply fertilizers either before or at planting using a constant N rate (Randall and Schmitt, 1998; IFA, 2013). Our results demonstrate that a constant application of 180 kg N ha^{-1} was able to achieve economic profits within \$25 ha^{-1} of maximum farm-level profit threshold (**Figure 4**). When both farm-level economic and environmental costs are considered, we advocate split instead of single fertilizer application so that farmers have flexibility in adjusting N rates based on weather during a growing season. We emphasize that farmers should consider flat profit response curve above which farm-level profits do not accrue rather environmental costs increase substantially (Rajsic and Weersink, 2008). In 6 out of 10 years, $EONR_{env}$ shifted away from the \$25 ha^{-1} of maximum farm-level profit threshold plateau. It suggests that farm-level economic losses for reducing environmental costs were substantial in 6 of 10 years. Without economic incentives, farmers are not likely to adopt environmentally optimal N rates. Therefore, different stakeholders should work with farmers and explore practical ways to compensate for farm-level economic losses incurred in order to adjust N rates. To help farmers and facilitate engagements between different stakeholders in adjusting N rates, new generation N management DSS must incorporate environmental costs (Banger et al., 2017; Morris et al., 2018). Future studies should focus on developing a DSS which farmers can use to assess environmental tradeoffs associated with N rates in a growing season.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

KB and BD developed the conceptual framework of the manuscript. JN calibrated and validated the model. KJ and JS conducted the analysis. KB, BD, KJ, JN, and JS wrote the manuscript draft. All authors contributed to the article and approved the submitted version.

FUNDING

This research was undertaken thanks in part to funding from the International Plant Nutrition Institute, the Canada First Research Excellence Fund - Grant #499005, and Grain Farmers of Ontario - Grant #53198.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.00096/full#supplementary-material

REFERENCES

Abalos, D., Brown, S. E., Vanderzaag, A. C., Gordon, R. J., Dunfield, K. E., and Wagner-Riddle, C. (2016a). Micrometeorological measurements over 3years reveal differences in N2O emissions between annual and perennial crops. *Glob. Change Biol*. 22, 1244–1255. doi: 10.1111/gcb.13137

Abalos, D., Smith, W. N., Grant, B. B., Drury, C. F., MacKell, S., and Wagner-Riddle, C. (2016b). Scenario analysis of fertilizer management practices for N2O mitigation from corn systems in Canada. *Sci. Total Environ*. 573, 356–365. doi: 10.1016/j.scitotenv.2016.08.153

Abendroth, L., Elmore, R. W., Boyer, M. J., and Marlay, S. (2011). *Corn Growth and Development* Ames, IA: Iowa State University.

Anderson, D. M., Burkholder, J. M., Cochlan, W. P., Glibert, P. M., Gobler, C. J., Heil, C. A., et al. (2008). Harmful algal blooms and eutrophication: examining linkages from selected coastal regions of the United States. *Harmful Algae*. 8, 39–53. doi: 10.1016/j.hal.2008.08.017

Banger, K., Wagner-Riddlge, C., Grant, B., Smith, W., Drury, C., and Yang, J. (2020). Modifying fertilizer rate and application method reduces environmental nitrogen losses and increases corn yield in Ontario. *Sci. Total Environ*. 722:137851. doi: 10.1016/j.scitotenv.2020.137851

Banger, K., Yuan, M., Wang, J., Nafziger, E. D., and Pittelkow, C. M. (2017). A vision for incorporating environmental effects into nitrogen management decision support tools for U.S. maize production. *Front Plant Sci*. 28:1270. doi: 10.3389/fpls.2017.01270

Birch, M. B. L., Gramig, B. M., Moomaw, W. R., Doering, O. C., and Reeling, C. J. (2011). Why metrics matter: evaluating policy choices for reactive nitrogen in the chesapeake bay watershed. *Environ. Sci. Technol*. 45, 168–174. doi: 10.1021/es101472z

Cabas, J., Weersink, A., and Olale, E. (2010). Crop yield response to economic, site and climatic variables. *Clim. Change* 101, 599–616. doi: 10.1007/s10584-009-9754-4

Cassman, K. G., Dobermann, A., and Walters, D. T. (2002). Agroecosystems, nitrogen-use efficiency, and nitrogen management. *Ambio* 31, 132–140. doi: 10.1579/0044-7447-31.2.132

Compton, J. E., Harrison, J. A., Dennis, R. L., Greaver, T. L., Hill, B. H., Jordan, S. J., et al. (2011). Ecosystem services altered by human changes in the nitrogen cycle: a new perspective for US decision making. *Ecol. Lett*. 14, 804–815. doi: 10.1111/j.1461-0248.2011.01631.x

Congreves, K. A., Dutta, B., Grant, B. B., Smith, W. N., Desjardins, R. L., and Wagner-Riddle, C. (2016a). How does climate variability influence nitrogen loss in temperate agroecosystems under contrasting management systems? *Agric. Ecosyst. Environ*. 227, 33–41. doi: 10.1016/j.agee.2016.04.025

Congreves, K. A., Grant, B. B., Dutta, B., Smith, W. N., Chantigny, M. H., Rochette, P., et al. (2016b). Predicting ammonia volatilization after field application of swine slurry: DNDC model development. *Agric. Ecosyst. Environ*. 219, 179–189. doi: 10.1016/j.agee.2015.10.028

Derby, N. E., Casey, F. X. M., Knighton, R. E., and Steele, D. D. (2004). Midseason nitrogen fertility management for corn based on weather and yield prediction. *Agron. J*. 96, 494–501. doi: 10.2134/agronj2004.4940

Dhital, S., and Raun, W. R. (2016). Variability in optimum nitrogen rates for maize. *Agron. J*. 108, 2165–2173. doi: 10.2134/agronj2016.03.0139

Dodds, W. K., Bouska, W. W., Eitzmann, J. L., Pilger, T. J., Pitts, K. L., Riley, A. J., et al. (2009). Eutrophication of US freshwaters: analysis of potential economic damages. *Environ. Sci. Technol*. 43, 12–19. doi: 10.1021/es801217q

Drury, C. F., Yang, J. Y., De Jong, R., Yang, X. M., Huffman, E. C., Kirkwood, V., et al. (2007). Residual soil nitrogen indicator for agricultural land in Canada. *Can. J. Soil Sci*. 87, 167–177. doi: 10.4141/S06-064

Drury, C. F., Yang, X. M., Reynolds, W. D., Calder, W., Oloya, T. O., and Woodley, A. L. (2017). Combining urease and nitrification inhibitors with incorporation reduces ammonia and nitrous oxide emissions and increases corn yields. *J. Environ. Qual*. 46, 939–949. doi: 10.2134/jeq2017.03.0106

Dutta, B., Smith, W. N., Grant, B. B., Pattey, E., Desjardins, R. L., and Li, C. (2016). Model development in DNDC for the prediction of evapotranspiration and water use in temperate field cropping systems. *Environ. Model. Softw*. 80, 9–25. doi: 10.1016/j.envsoft.2016.02.014

Ewing, P. M., and Runck, B. C. (2015). Optimizing nitrogen rates in the midwestern United States for maximum ecosystem value. *Ecol. Soc*. 20:18. doi: 10.5751/ES-06767-200118

FieldView (2019). *Climate FieldView*. Available online at: https://climate.com/features/fertility-management (accessed October 20, 2019).

Frink, C. R., Waggoner, P. E., and Ausubel, J. H. (1999). Nitrogen fertilizer: retrospect and prospect. *Proc. Natl. Acad. Sci. U.S.A*. 96, 1175–1180. doi: 10.1073/pnas.96.4.1175

Holley, C. (2015). Crafting Collaborative Governance: Water Resources, California's Delta Plan, and Audited Self-Management in New Zealand. *UNSW Law Research Paper no. 2015-15 ed Social Science Research Network. Environmental Law Reporter* Omafra.

Howarth, R. W., and Marino, R. (2006). Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades. *Limnol. Oceanogr*. 51, 364–376. doi: 10.4319/lo.2006.51.1_part_2.0364

IFA (2013). *Fertilizer Consumption Statistics*. Paris: International Fertilizer Industry Association.

IPCC (2014). *Climate change. Mitigation of Climate Change. Contribution of working group III to the fifth assessment report of the Intergovernmental Panel on Climate Change*. Cambridge, NY: IPCC.

Jarecki, M., Grant, B., Smith, W., Deen, B., Drury, C., VanderZaag, A., et al. (2018). Long-term trends in corn yields and soil carbon under diversified crop rotations. *J. Environ. Qual*. 47, 635–643. doi: 10.2134/jeq2017.08.0317

Kroebel, R., Smith, W. N., Grant, B. B., Desjardins, R. L., Campbell, C. A., Tremblay, N., et al. (2011). Development and evaluation of a new Canadian spring wheat sub-model for DNDC. *Can. J. Soil Sci*. 91, 503–520. doi: 10.4141/cjss2010-059

Ladha, J. K., Pathak, H., Krupnik, T. J., Six, J., and van Kessel, C. (2005). Efficiency of fertilizer nitrogen in cereal production: retrospects and prospects. *Adv. Agron*. 87, 85–156. doi: 10.1016/S0065-2113(05)87003-8

Liu, Y. Y., Swinton, S. M., and Miller, N. R. (2006). Is site-specific yield response consistent over time? Does it pay?. *Am. J. Agricu. Econ*. 88, 471–483. doi: 10.1111/j.1467-8276.2006.00872.x

McEwan, K. (2019). *Reports on Farm Input Prices*. Available online at: https://www.ridgetownc.com/research/research_reports_topic.cfm?ref=FARM%5FINPUT%5FPRICES (accessed October 20, 2019).

Morris, T. F., Murrell, T. S., Beegle, D. B., Camberato, J. J., Ferguson, R. B., Grove, J., et al. (2018). Strengths and limitations of nitrogen rate recommendations for corn and opportunities for improvement. *Agron. J*. 110, 1–37. doi: 10.2134/agronj2017.02.0112

NAAS (2015). *National Agricultural Statistics Service*. Available online at: https://www.nass.usda.gov (accessed October 20, 2019).

Nasielski, J., Grant, B., Smith, W., Niemeyer, C., Janovicek, K., and Deen, B. (2020). Effect of nitrogen source, placement and timing on the environmental performance of economically optimum nitrogen rates in maize. *Field Crops Res*. 246:107686. doi: 10.1016/j.fcr.2019.107686

N-Calculator (2020). *The Ontario Corn Nitrogen Calculator* Omafra.

OMAFRA (2019). *Statistics-Field Crops*. Available online at: http://www.omafra.gov.on.ca/english/stats/crops/estimate_hist_metric.xlsx (accessed October 20, 2019).

Pannell, D. J. (2006). Flat earth economics: the far-reaching consequences of flat payoff functions in economic decision making. *Rev. Agric. Econ*. 28, 553–566. doi: 10.1111/j.1467-9353.2006.00322.x

Powlson, D. S., Addisott, T. M., Benjamin, N., Cassman, K. G., de Kok, T. M., van Grinsven, H., et al. (2008). When does nitrate become a risk for humans?. *J. Environ. Qual*. 37, 291–295. doi: 10.2134/jeq2007.0177

Rajsic, P., and Weersink, A. (2008). Do farmers waste fertilizer? A comparison of ex post optimal nitrogen rates and ex ante recommendations by model, site and year. *Agricu. Syst*. 97, 56–67. doi: 10.1016/j.agsy.2007.12.001

Randall, G. W., and Schmitt, M. A. (1998). "Advisability of fall-applying nitrogen," in *Proceedings of the 1998 Wisconsin Fertilizer, Aglime, and Pest Management Conference* (Madison: University of Wisconsin), 90–96.

Sawyer, J., Nafziger, E., Randall, G., Bundy, L., Rehm, G., and Joern, B. (2006). *Concepts and Rationale for Regional N Rate Guidelines for Corn*. Ames, IA: Iowa State University Extension Regional Publication PM2015.

Scharf, P. C., Shannon, D. K., Palm, H. L., Sudduth, K. A., Drummond, S. T., Kitchen, N. R., et al. (2011). Sensor-based nitrogen applications out-performed producer-chosen rates for corn in on-farm demonstrations. *Agron. J*. 103, 1683–1691. doi: 10.2134/agronj2011.0164

Sela, S., van Es, H. M., Moebius-Clune, B. N., Marjerison, R., Melkonian, J., Moebius-Clune, D., et al. (2016). Adapt-N outperforms grower-selected nitrogen rates in northeast and midwestern united states strip trials. *Agron. J*. 108, 1726–1734. doi: 10.2134/agronj2015.0606

Smith, W. N., Grant, B. B., Desjardins, R. L., Kroebel, R., Li, C., Qian, B., et al. (2013). Assessing the effects of climate change on crop production and GHG emissions in Canada. *Agricu. Ecosyst. Environ*. 179, 139–150. doi: 10.1016/j.agee.2013.08.015

Sobota, D. J., Compton, J. E., McCrackin, M. L., and Singh, S. (2015). Cost of reactive nitrogen release from human activities to the environment in the United States. *Environ. Res. Lett*. 10:025006. doi: 10.1088/1748-9326/10/2/025006

Sogbedji, J. M., van Es, H. M., Klausner, S. D., Bouldin, D. R., and Cox, W. J. (2001). Spatial and temporal processes affecting nitrogen availability at the landscape scale. *Soil Tillage Res*. 58, 233–244. doi: 10.1016/S0167-1987(00)00171-9

Tomer, M. D., Porter, S. A., James, D. E., Boomer, K. M. B., Kostel, J. A., and McLellan, E. (2013). Combining precision conservation technologies into a flexible framework to facilitate agricultural watershed planning. *J. Soil Water Conserv*. 68, 113A—120A. doi: 10.2489/jswc.68.5.113A

Tremblay, N., Bouroubi, Y. M., Belec, C., Mullen, R. W., Kitchen, N. R., Thomason, W. E., et al. (2012). Corn response to nitrogen is influenced by soil texture and weather. *Agron. J*. 104, 1658–1671. doi: 10.2134/agronj2012.0184

Tukey, J. W. (1991). The philosophy of multiple comparisons. *Stat. Sci*. 6, 100–116. doi: 10.1214/ss/1177011945

Van Grinsven, H. J. M., Holland, M., Jacobsen, B. H., Klimont, Z., Sutton, M. A., and Willems, W. J. (2013). Costs and benefits of nitrogen for europe and implications for mitigation. *Environ. Sci. Technol*. 47, 3571–3579. doi: 10.1021/es303804g

Vetsch, J. A., and Randall, G. W. (2004). Corn production as affected by nitrogen application timing and tillage. *Agron. J*. 96, 502–509. doi: 10.2134/agronj2004.5020

Xie, M., Tremblay, N., Tremblay, G., Bourgeois, G., Bouroubi, M. Y., and Wei, Z. (2013). Weather effects on corn response to in-season nitrogen rates. *Can. J. Plant Sci*. 93, 407–417. doi: 10.4141/cjps2012-145

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 21 July 2020
doi: 10.3389/fsufs.2020.00098

Past and Current Dynamics of U.S. Agricultural Land Use and Policy

Kaitlyn Spangler[1]*, Emily K. Burchfield[2] and Britta Schumacher[1]

[1] Department of Environment and Society, Utah State University, Logan, UT, United States, [2] Department of Environmental Sciences, Emory University, Atlanta, GA, United States

Over the past century, agricultural land use in the United States has seen drastic shifts to support increasing demand for food and commodities; in many regions, this has resulted in highly simplified agricultural landscapes. Surmounting evidence exhibits the negative impacts of this simplification on the long-term provisioning of necessary ecosystem services to and from agriculture. However, transitions toward alternative systems often occur at a small scale, rather than at a systemic level. Within the National Research Council's (NRC) sustainable agricultural systems framework, we utilize national open-source datasets spanning several decades to broadly assess past and current agricultural landscapes across the U.S. We integrate and analyze agricultural land use and land cover data with policy data to address two main objectives: (1) Document and visualize changes over recent decades in cropland conversion, agricultural productivity, and crop composition across the U.S.; and (2) identify broad policy changes of the U.S. Farm Bills from 1933 to 2018 associated with these land use trends. We show that U.S. agriculture has gradually trended toward an intensely regulated and specialized system. Crop production is heavily concentrated in certain areas, larger farms are getting larger, while the number of smaller operations is decreasing, and crop diversity is declining. Meanwhile, federal agricultural policy is increasing in scope and influence. Through these data-driven insights, we argue that incremental and transformative pathways of change are needed to support alternative production practices, incentivize diversified landscapes, and promote innovation toward more sustainable agricultural systems across multiple scales.

Keywords: land use, policy, sustainable agricultural systems, U.S., crop production

OPEN ACCESS

Edited by:
Mark A. Drummond,
United States Geological Survey (USGS), United States

Reviewed by:
Roger F. Auch,
United States Geological Survey (USGS), United States
J. Arbuckle,
Iowa State University, United States

****Correspondence:***
Kaitlyn Spangler
kspangler@aggiemail.usu.edu

Specialty section:
This article was submitted to Agroecology and Ecosystem Services, a section of the journal Frontiers in Sustainable Food Systems

Received: *04 November 2019*
Accepted: *09 June 2020*
Published: *21 July 2020*

Citation:
Spangler K, Burchfield EK and Schumacher B (2020) Past and Current Dynamics of U.S. Agricultural Land Use and Policy. Front. Sustain. Food Syst. 4:98. doi: 10.3389/fsufs.2020.00098

INTRODUCTION

Agriculture has drastically transformed Earth's surface over the last century. Concerns arise in the ability of the global agri-food system to meet current and future food demands while maintaining biological diversity and conservation needs. Globally, since the 1960s, the large-scale demand and movement of commercial crops grown in intensive management systems has increased, contributing to a narrowing of crop species and genetic diversity worldwide (Harlan, 1975; Heal et al., 2004; Khoury et al., 2014). Surmounting evidence illustrates the negative ecological impacts of this shift, largely due to intensive annual crop production and landscape simplification (Pimentel et al., 1995; Tilman, 1999; Horrigan et al., 2002; Robinson and Sutherland, 2002; Benton et al., 2003; Bianchi et al., 2006). Simplified agricultural landscapes are associated with the degradation of key ecosystem services (ES)—or the benefits humans receive freely from the environment—that are essential to agricultural production, such as soil fertility, nutrient cycling, and genetic biodiversity, as well as regulating services including soil retention, pollination, natural pest control, and water purification (Tscharntke et al., 2005, 2012; Hendrickx et al., 2007; Meehan et al., 2011; Bommarco et al., 2013; McDaniel et al., 2014; Landis, 2017). ES generated by agricultural systems are primarily acquired through provisioning services, i.e., food, fiber, and fuel production, but also through cultural services, such as enhancing landscape aesthetics, building social networks, and market participation, and other services, such as wildlife habitat preservation; these mechanisms feed back into supporting and regulating services. Ecological functions that *disrupt* agricultural production (referred to as *disservices*), such as competition for water or crop damage from natural predators and pests, may further contribute to disservices generated *from* agriculture, including nutrient runoff or habitat loss (Rabalais et al., 2002; Zhang et al., 2007; Hillier et al., 2009; Cardinale et al., 2012; Hooper et al., 2012). Managing agriculture to optimize ecosystem health and the provisioning of key ES for agriculture while minimizing disservices can increase the stability and quantity of production over time, decrease need for external inputs, and increase ES delivery to the broader ecosystem (Cassman, 1999; Tscharntke et al., 2005; Bommarco et al., 2013, 2018; Pywell et al., 2015; Burchfield et al., 2019).

Recent calls for transformations in our agricultural landscapes emphasize the importance of agricultural systems that boost ES for agriculture through practices that are environmentally, economically, and socially beneficial while also maintaining or increasing productivity (Reganold et al., 2011). The National Research Council's (NRC) Committee on Twenty First Century Systems Agriculture (NRC, 2010) defined several objectives for sustainable agricultural systems. First and foremost, agricultural sustainability is defined within four main themes: (1) Satisfy human food, feed, and fiber needs and contribute to biofuel needs; (2) enhance environmental quality and the resource base; (3) sustain the economic viability of agriculture; and (4) enhance the quality of life for farmers, farm workers, and society as a whole. These main objectives of sustainability align with NRC's "systems agriculture" approach to understanding the interactions among actors and components of the system as a whole, rather than the function of each component separately. The NRC further identified three main qualities of system's robustness to use as considerations for systems moving toward sustainability. Robustness encompasses resistance (ability to withstand shocks), resilience (capacity to absorb shocks and stressors over time), and adaptability (ability to make necessary systemic changes in response to long-term environmental changes).

Identifying pathways toward sustainable change cannot be viewed through a dichotomous conventional-sustainable lens but rather contextualized within social, political, economic, and ecological drivers. As the NRC states, "The committee's definition of sustainable farming does not accept a sharp dichotomy between conventional and sustainable farming systems, not only because farming enterprises reflect many combinations of farming practices, organization forms, and management strategies, but also because all types of systems can potentially contribute to achieving various sustainability goals and objectives" (2010, p. 37). Although poorly defined across disciplines, agroecology has long presented viable alternatives to industrial agricultural practices (Francis et al., 2003). Rather than focusing on certain agroecological on-farm practices, we ground this paper in the broad definition from Brym and Reeve (2016, p. 214): agroecology is a "field of study motivated to understand ecological, evolutionary, and socioeconomic principles and use them in an improvement process that sustains food production, conserves resources, and maintains social equality." This definition aligns with calls from the NRC to move toward greater sustainability through several pathways of change, either incremental or transformative. Incremental change can gradually increase and support the adoption of current conservation practices to make them more widespread within conventional systems, as well as also support research for the economic viability of such practices. Transformative change would support broader, systemic shifts from conventional and agroecological approaches through establishing new markets and supporting ecologically based management (e.g., organic, mixed systems) (NRC, 2010).

We build upon prior research that has attempted to assess and interpret changes in U.S. agricultural systems over time. Several studies have focused on land use change within specific regions of the U.S., such as agricultural land cover loss due to competing development demands in the Eastern U.S. (Drummond and Loveland, 2010; Sayler et al., 2016) or cropland concentration due to high soil quality in the South (Hart, 1978). A large number of studies have shown how the Corn Belt has intensified agricultural land toward specialized commodity production over time due to favorable climatic conditions, high quality land, and political incentives (Hart, 1986, 1991, 2001, 2004; Hudson, 1994; Drummond et al., 2012; Auch and Karstensen, 2015; Laingen, 2017). Other studies discuss trends of fluctuating conversion from grassland and marginal cropland to intensive commodity and biofuel production in the Great Plains, driven by enrollment in federal conservation programs, technological advances, improved management practices, and increased precipitation (Drummond et al., 2012; Wright and Wimberly, 2013; Reitsma et al., 2015; Taylor et al., 2015; Auch et al., 2018). However, these studies are limited in geographic scope and do not contextualize such trends in the national aggregate. Research with a broad U.S. focus are either outdated (Hudson, 1994; Hart, 2001; Cozen, 2010) or fail to discuss political drivers and environmental implications within an agroecological framework (Sleeter et al., 2013; Sohl et al., 2016; Auch et al., 2018; Hudson and Laingen, 2018). Other recent research has attempted to project recent land cover datasets farther back in time to assess historical land use trends (e.g., Arora and Wolter, 2018) but do not extend past the 1980s and emphasize the need to understand current land use trends through historical processes. Given the trajectory of U.S. federal agricultural policy, land use changes prior to the 1970s and 1980s are important in understanding how current trends were established and are reinforced. Data-driven research can help identify trends within and across agricultural systems to better inform the prioritization of sustainability objectives.

This paper serves as a high-level overview of how agricultural land use and policy drivers have changed at a national level over the past half century. Rather than attempting to evaluate the current state of sustainability of the U.S. agricultural system, this data-driven narrative serves two main objectives: (1) to clarify the magnitude and extent of large-scale agricultural landscape transformations, as well as the changes in policy structure, and (2) provide a framework to interpret and assess sustainable pathways for future agricultural change at the national scale. After discussing the methods, we present data trends and figures and contextualize these findings in the discussion section. We conclude with recommendations of national-level factors to consider within transitions toward more sustainable agriculture systems.

METHODS

We utilized open-source datasets and open-source programming software to visualize policy, land use, and agricultural production changes. The majority of these data are focused on the county scale, as it is the finest resolution at which farm-level data is aggregated in the U.S. Using county-level data enabled us to understand, visualize, and interpret the spatial and temporal complexities of national agricultural trends. Through such visualizations, we illustrated trends in cropland transitions, crop composition, and the policy structure of the Farm Bills.

Datasets

Various multiscale datasets were synthesized and merged into a panel dataset (**Table 1**). Crop acreage, farm size, and chemical inputs were obtained through the National Agricultural Statistics Service (NASS) (USDA NASS, 2019c), whereby the county-year scale is the highest resolution available. The NASS database presents data both from the U.S. Census of Agriculture and a variety of national agricultural surveys administered by the USDA. USDA surveys are administered at the county and state scale annually with foci such as crop/stocks to measure crop acreage and yield, farm labor, crop prices and markets, and more specific topics, such as milk or broiler production. For some surveys, data are available from the mid-1800s to present day. The NASS QuickStats interface provides all of this survey information but does not indicate which survey the data are from or clearly define the cutoff of *who* counts in the surveys; additionally, the sampling strategy is determined by each state. Openly available from 1997 onward, the Census is conducted every 5 years and is administered to all farms and ranches (in rural or urban settings) producing and potentially selling at least $1,000 of their products. The Census is the only source of detailed county-level agricultural data that is collected, tabulated, and published using a uniform set of definitions and methodology. Thus, the Census is considered the most complete count and measurement of U.S. farms, operators, and ranches in the U.S. Though the combination of these data is limited in its generalizability given its inconsistency of data collection measures, it provides the most comprehensive, open-source record of historical U.S. agricultural data.

There are few land cover datasets that cover the entire U.S. and also extend decades back in time. Given its moderate spatial and temporal resolution, we utilized the National Wall-to-Wall Anthropogenic Land Use Trends (NWALT) dataset created by the U.S. Geological Survey (USGS) (Falcone, 2015). It uses the 2011 National Land Cover Database (NLCD) (Homer et al., 2015) as a base grid and other USGS and USDA historical imagery and datasets to map land use farther back in time with similar accuracy. NWALT classifications agreed with NLCD land use classifications from 2001-2011 with at least 94% accuracy and agreed with over 99.5% of county-level cropland changes from the USDA Census of Agriculture (Falcone, 2015). This dataset contains five 60-meter (m) resolution raster datasets from the years 1974, 1982, 1992, 2002, and 2012 of land use across the coterminous U.S, extending farther back in time than most other land cover datasets. However, some of the underlying data may span several years rather than an exact snapshot in time (Falcone, 2015); therefore, NWALT can be used for assessing broad temporal trends. We computed agricultural land as a percentage of overall county land to match the spatial resolution of NASS data. Agricultural land pixels are differentiated in this dataset by cultivated crop production and pasture/hay production based on 2011 NLCD classifications. Agricultural infrastructure, such as farm roads, are not included in these classifications.

The USDA's Economic Research Service (ERS) Major Land Uses (MLU) series has been collected every 5 years beginning in 1945, coinciding with the Census of Agriculture. As such, the ERS MLU is the longest running, most comprehensive accounting of all major land uses in the U.S. The dataset provides acreage across six land use categories (cropland, grassland pasture and range, forest-use land, special-use areas, urban areas, and miscellaneous other land) at both regional (Pacific, Mountain, Southern Plains, etc.) and state scales, compiled by reconciling several data sources. Thus, despite the ERS's use of standardized procedures to measure land use (Barnard and Hexem, 1988), there is a degree of uncertainty introduced by making comparisons through time. For this dataset, cropland includes cropland used for crops

TABLE 1 | Datasets used to visualize crop composition, acreage, productivity, and policy changes.

Variable	Spatial resolution	Temporal resolution	Duration	Source
Crop acreage	County/National	Annual	1920–2019	USDA NASS Survey
Major land use	State	Every 5 years	1945–2012	ERS MLU
Average farm size	County	Every 5 years	1997–2017	USDA Census of Agriculture
Agricultural inputs	County	Every 5 years	1997–2017	USDA Census of Agriculture
Agricultural land cover	County	Every 10 years	1974–2012	NWALT
Farm Bill	National	Every 5 years	1933–2018	National Ag. Law Center

(harvested, crop failure, and cultivated summer fallow), cropland used for pasture (considered to be in long term rotation), and cropland idled. Grassland, pasture and range includes grassland and other non-forested pasture and range in farms, as well as estimates for open and non-forested grazing lands not in farms. Special use areas include rural transportation, rural parks and wildlife, defense and industrial areas, and miscellaneous farmland (farmsteads, farm roads and lanes, and misc. farmland). Urban areas include densely populated urbanized areas of 2,500 to 50,000 people or more, and forested areas including forest cover of grazed (commercial use) and non-grazed forest. We utilized this dataset to track trends in cropland conversion in comparison to other ERS MLUs between 1945 and 2012 (Bigelow and Borchers, 2017).

Finally, the U.S. Farm Bill (FB) policy documents from 1933 to 2018 are openly available through the National Agricultural Law Center (2019). While not the *only* important agricultural policy in recent U.S. history, the FB has played a key role in how, where, why, and what type of food is produced at a national scale. Over time, it has grown in size to encompass nearly all aspects of food production. These policy documents have changed in structure, starting with a 25-page document in 1933 encompassing two main topics: (1) agricultural adjustment and (2) agricultural credit, and becoming a 529-page document in 2018, encompassing 12 specific "Titles" ranging from Commodities to Nutrition to Rural Development. Within these Titles are statutes and funding programs that largely define the broader policy structure within which agricultural land use decisions are made.

Data Exploration

Using exploratory mapping and data mining techniques in R (version 3.6.3) (R Core Team, 2020), we selected variables of interest and assessed their spatiotemporal consistency and availability. This included plotting variables over time at county, state, and national scales to determine data reliability and representativeness, noting when and how representation changed across scales We focused on county-level data whenever possible as the most interpretable scale of agricultural landscape change. Particularly for NASS data, availability is variable by county, state, and year based on changing federal data collection, reporting procedures, and data privacy concerns; there are noted inconsistencies *across* USDA datasets as well (Hart, 2001; Arora and Wolter, 2018). Nonetheless, land use science and spatial modeling communities have acknowledged and accepted the need to use data at multiple scales given a lack of other alternatives (Rindfuss et al., 2004; Auch et al., 2018). Ultimately, we focused on six main variables of interest: (1) acres planted (by crop, per county and nationally), (2) percent planted (by crop, per county), (3) average acres per farm operation (per county), (4) percent crop and pasture land (per county), (5) cropland acreage (as a proportion of national acreage), and (6) agricultural input use (per county).

Given the changing structure and purpose of federal FB policies, we conducted a broad content analysis of the FB documents as a systematic way of capturing the frequency and content of textual data of the FBs from 1933 to 2018 (Krippendorf, 2004). With the qualitative coding software ATLAS.ti, we utilized a predetermined coding scheme to identify two major themes in each FB: (1) the number of distinct crops and (2) the stated purpose. These codes aimed to operationalize the scope and purpose of the FB as it relates to commodity production. Coding was limited to Titles, programs, and definitions that directly defined commodity crops, stipulated support and subsidies for their production, and promoted commodity markets; these included commodity programs, trade, agricultural marketing, credit, and crop insurance but excluded nutrition, conservation, forestry, research, etc. While excluded Titles do play a role in commodity production and land use, we explicitly focused on those that drive and regulate the composition of crops produced. Further, commodity definitions in the FB are defined within the commodity programs, and other Titles, such as conservation, are based upon these prior definitions. We contextualized these results within academic and gray literature.

RESULTS

The results of this data synthesis are organized by three main themes. The first theme is land use which includes cropland, farm size, and productivity by visualizing trends in location of agricultural land, regional farm size variation, and how these changes relate to increased productivity of U.S. agriculture. The second theme is crop composition, including the composition of crops and how their relative acreage varies across space and has changed through time. The third theme is policy, presenting data to contextualize the overarching FB policy structure, how it has changed, and how it affects the first and second themes.

Changes are referenced within the regional specifications of the USDA ERSs Farm Resource Regions (FRRs) (**Figure 1**). These regions portray the geographic distribution of, and specialization within, the production of U.S. farm commodities (ERS, 2000). FRRs aggregate areas with similar types of farms, commodities, soil, physiographic, and climate characteristics nationally to contrast with the state and county boundaries (that are often political rather than biophysical borders) used to visualize data trends. We utilized these regions to further understand and contextualize trends across themes.

Cropland, Farm Size, and Productivity

U.S. cropland has changed in both amount and type over recent decades. From 1945 to 2012, cropland as a proportion of total land use decreased; meanwhile urban and special use areas increased (**Figure 2**). As seen in **Figure 2**, there was a slight decrease from 23.7% of the national share of land use in 1945 to 20.7% in 2012 (3% decrease). Comparatively, urban areas increased from 0.8% of the national share in 1945 to 3.7% in 2012. Special-use areas increased from 4.5% in 1945 to 8.9% in 2012. Grassland, pasture and range decreased by 0.03%. Forest-use decreased from 31.6 to 28.5%. Miscellaneous land uses decreased from 4.9 to 3.6%. However, both the ERS MLU and NWALT data confirm that cropland as a percentage of national land has decreased by 3% just since the 1970s. Therefore, this decline primarily occurred within the past four decades.

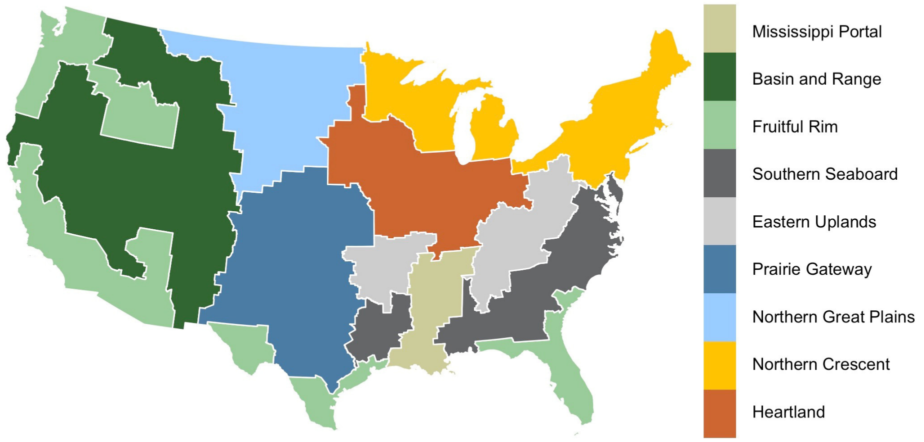


FIGURE 1 | Farm Resource Regions (FRRs) across the U.S., determined by crop production type, amount, and value.

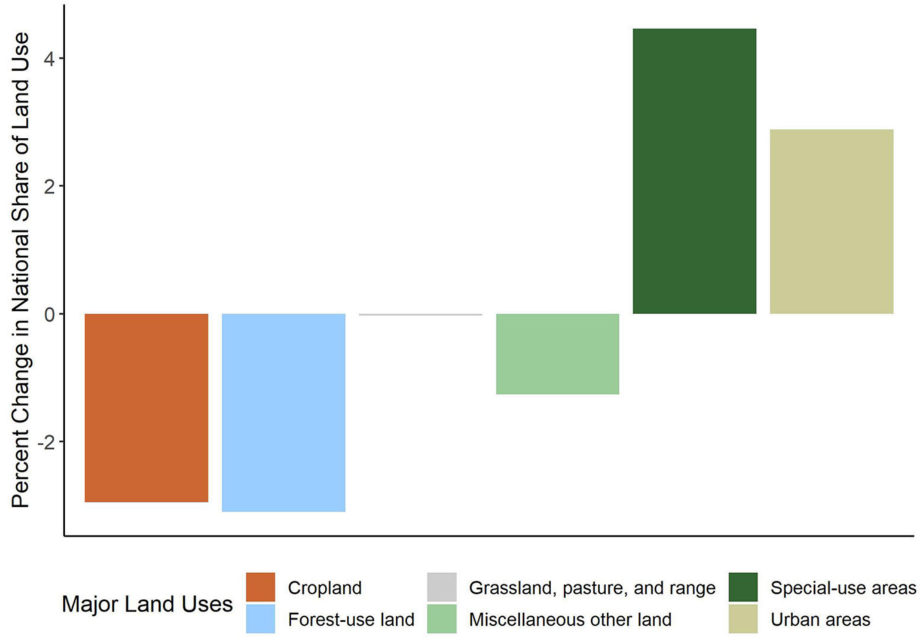


FIGURE 2 | Percent change in the national share of land use across the ERS Major Land Use categories between 1945 and 2012. Data: ERS MLU.

Further, crops are grown in fairly concentrated regions, and there are no obvious changes in *location* of cropland. According to the NWALT data, counties where cropland is dominant have remained consistent over the past few decades without dramatic conversion of other land uses to cropland (see **Supplementary Figure 1**); by "dominant," we mean that cropland accounts for most of the land use in a county. Though dominance does not tell the full story of a commodity (i.e., it does not demonstrate which counties are the most *productive*), it is an important metric in understanding the composition of U.S. agricultural landscapes. As **Figure 3** illustrates, some counties, e.g., in the Heartland region, are almost entirely covered by cropland (nearly 100%), while others, e.g., in the Basin and Range region, produce few, if any, crops. **Figure 3** also illustrates where cropland is most prevalent by county. The Southern Seaboard and the Fruitful Rim of California and the Pacific Northwest demonstrate clear intra-regional agricultural clustering, whereby crop production is concentrated in a select few counties. The midwestern Heartland and Mississippi Portal regions are dominated by cropland compared to the rest of the country; these areas of cropland dominance largely align with spatial

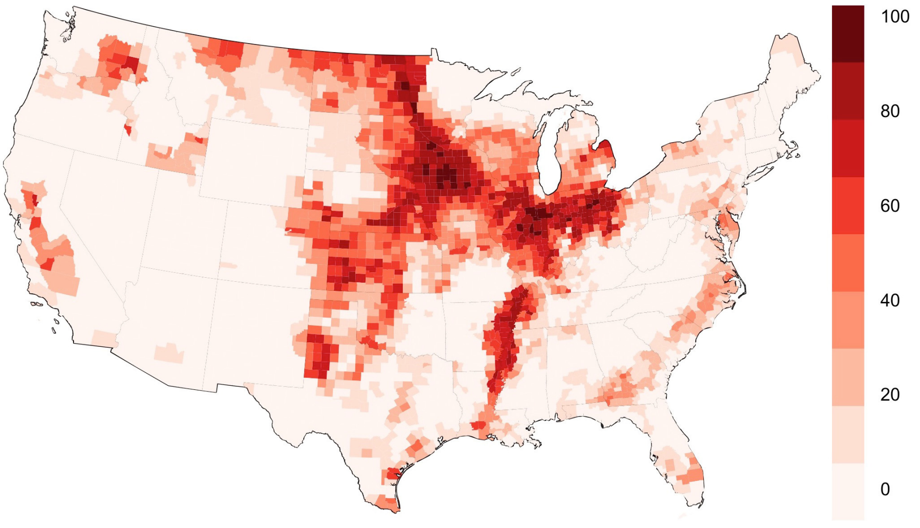


FIGURE 3 | Percent cropland by county in 2012. Data: NWALT.

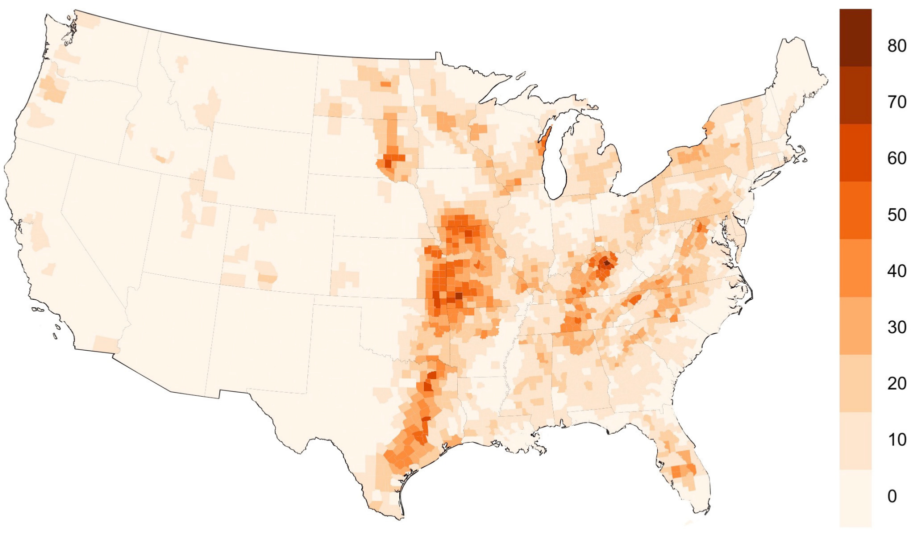


FIGURE 4 | Percent pasture and hay land by county in 2012. Data: NWALT.

trends in harvested acres for corn, soy, and wheat (see **Supplementary Figures 2–4**).

Pasture and land in hay production also demonstrate patterns of clustering. The proportion of land devoted to hay and pasture in the U.S. has decreased by 13.8% from the 1970s to 2012 (according to NWALT data), which is a larger change than the decrease in cropland (−2%). Furthermore, according to the ERS MLU data, grassland pasture and range have only lost 0.08% of its share of total land use between 1945 and 2012. Areas within the Heartland, Eastern Uplands, and Prairie Gateway regions exhibit high proportions of pasture and hay (**Figure 4**), whereby some counties are 50 to 70% covered by such production. However, these areas of landscape dominance do not necessarily produce the highest yields or relative yields (yield/harvested acre) in the U.S. For instance, clusters of counties in the West Coast portion of the Fruitful Rim harvest more hay per acre than any county in the Heartland (see **Supplementary Figure 5**). Pasture-dominant areas do not appear to overlap with crop-dominant areas, indicating divergent specialization in intensive crop and pastureland.

Farm size has been changing alongside the concentration of national agricultural land. The total number of farms has declined over time. In 2018, the USDA estimated 2 million farms nationally, which is 12,800 farms less than the estimate for 2017 (USDA NASS, 2019b). In 2011, the estimate was nearly 2.13 million; over 8 years, there was a 4.7% decrease in the number

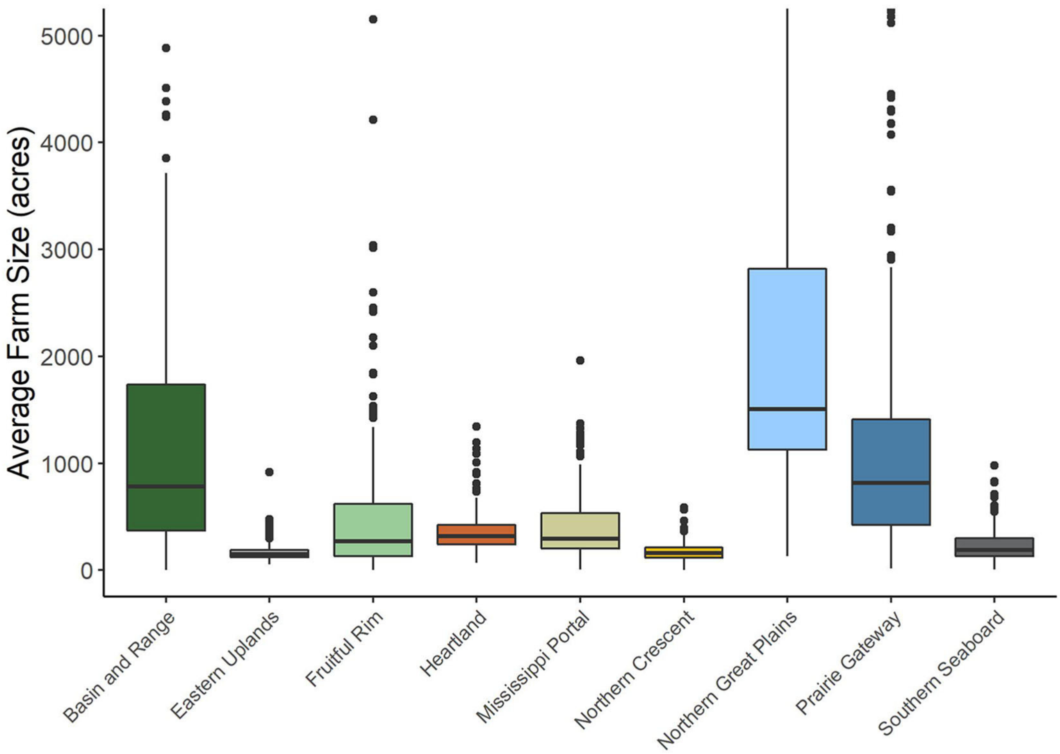


FIGURE 5 | Average farm size (acres per operation) by FRR in 2012. Counties with an average farm size > 5,000 acres [*n* = 46, range = 5,119 to 37,952 acres] were removed from visualization for readability. Data: USDA NASS Survey.

of farms nationally (USDA NASS, 2019b). The peak number of farms in the U.S. was in 1935 at 6.8 million farms, but this number has steadily decreased since then (Hoppe, 2014). Meanwhile, highly productive industrial farms have expanded in size while midsize farms continue to decrease in number. For example, of all agricultural land in the U.S. in 2018, 40.8% is operated by large-scale farms that earn sales of $500,000 or more, but these large operations comprise merely 7.5% of all total *number* of farms; farms that earn less than $100,000 comprise 30.1% of all farmland but comprise 81.5% of all farms (USDA NASS, 2019b). Thus, significantly fewer large-scale family and commercial farms operate a greater proportion of cropland.

Given this shift, total average farm size has not changed much in recent decades. According to the Census of Agriculture, the national average farm size changed from 440 acres in 1982, to 491 acres in 1992, to 433 acres in 2012, and 443 in 2019 (USDA, 1982, 1992; USDA NASS, 2019b). Therefore, average farm size has remained relatively stable due to a disproportionately greater number of smaller farms and larger farms increasing in size (Hoppe, 2014; MacDonald and Hoppe, 2017).

Regional differences of farm size further affect these averages. As seen in **Figure 5**, the largest farms are found in the Northern Great Plains [median = 1,505 acres, mean = 2,135 acres, standard deviation (SD) = 1,528 acres] and Basin and Range Regions (median = 783 acres, mean = 1,369 acres, SD = 1,516 acres), while the smallest farms are found in the Eastern Uplands (median = 148 acres, mean = 165 acres, SD = 77 acres) and Northern Crescent Regions (median = 161 acres, mean = 168 acres, SD = 80 acres). However, most regions have several outlier counties that exhibit average county farm sizes significantly beyond the regional mean. In particular, counties in the Basin and Range (median = 783 acres, mean = 1,368 acres, SD = 1,515 acres), Fruitful Rim (median = 271 acres, mean = 1,145 acres, SD = 3,756 acres), and Prairie Gateway (median = 817 acres, mean = 1,143 acres, SD = 1,186 acres) exhibit a wide range of average farm sizes; some counties in these regions average well over 5,000 acres per operation. Since most pasture and hay production occurs within the Prairie Gateway (**Figure 4**), these data show that such production in certain counties comprises much larger farms than the rest of the region. Contrastingly, regions such as the Eastern Uplands, Heartland, Northern Crescent, and Southern Seaboard exhibit outliers noticeably closer to the regional median. Given that the majority of cropland falls within the Heartland region (**Figure 3**), these data demonstrate that most of these farms are similar in size and are not the largest on average at a national scale (median = 319 acres, mean = 343 acres, SD = 155 acres).

Further, **Figure 6** illustrates the variability in average farm size by county. The largest farms (in acres/operation per county) are found primarily in the western U.S. with a clear distinction between eastern and western counties. This also indicates where the largest farms in the Basin and Range, Prairie Gateway, and

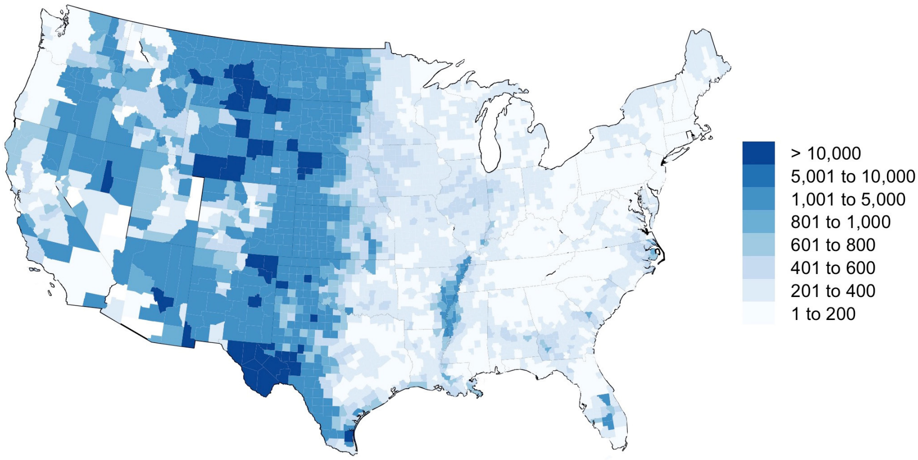


FIGURE 6 | Average farm size (acres per operation) by county in 2012. Data: USDA NASS Census of Agriculture.

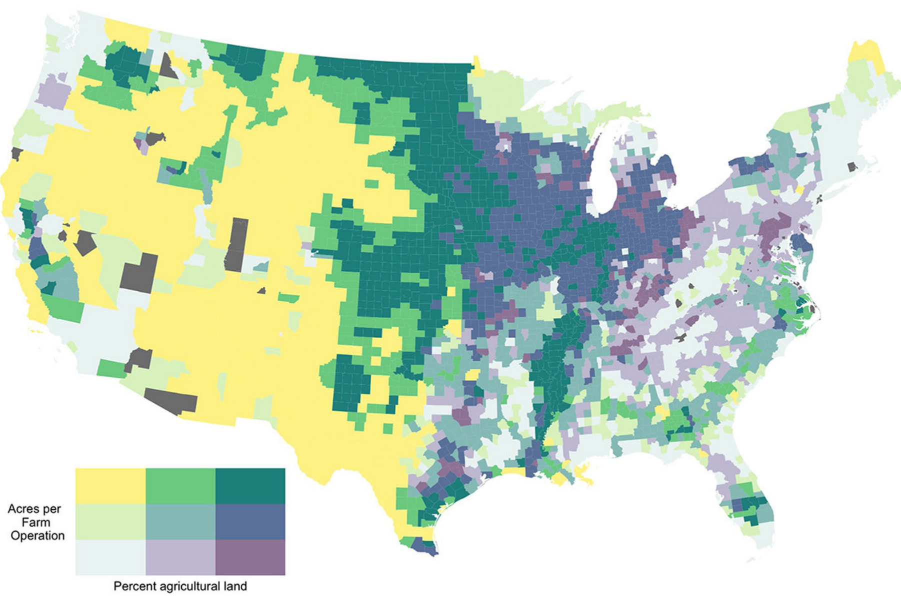


FIGURE 7 | Bivariate choropleth constructed by binning county-level average farm size (by acre per operation per county) and percent agricultural land by county (both pasture and crop production) into thirds and pairing each tercile into distinct categories. Yellow indicates counties with large average farm sizes (in acres/operation) and a low percentage of agricultural land. Teal indicates counties with large average farm sizes and a high percentage of agricultural land. Purple indicates counties with a small average farm size but a large percentage of the county as agricultural land. Light blue is both low percentage agricultural land and a small average farm size per county. Dark gray counties indicate missing data. Data: NWALT and USDA NASS Census of Agriculture.

Texas portion of the Fruitful Rim regions are located. Farms that average over 10,000 acres are exclusively found in these regions and are clustered together. Most of the average farm sizes in these regions exceed 1,000 acres, if not 5,000 acres. In the Heartland, however, most farms do not exceed an average of 400 acres per operation.

When directly comparing farm size and dominance of agricultural land (including both cropland and hay/pasture production) by county, certain areas exhibit large farm sizes but are not dominated by agricultural production at the county scale. By binning both average farm size by county and percentage agricultural land by county into thirds and pairing each tercile into distinct categories, we visualize the spatial relationship between farm size and agricultural dominance (**Figure 7**). Counties largely in the Heartland, Mississippi Portal, and Northern Great Plains exhibit, on average, medium and large farms with the highest percentage of agricultural land (in teal). Much of the counties in the Basin and Range and Prairie Gateway exhibit large average farm sizes (in acres/operation) and a low percentage of agricultural land (yellow). Counties with relatively small average farm size but a large percentage of the county as agricultural land (dark purple) are scattered throughout the rest of the Heartland, while both low percentage agricultural land and a relatively small average farm size per county (light blue-green) are almost exclusively found in the Southern Seaboard, Northern Crescent, and northwestern Fruitful Rim. These trends reflect the different landscape composition patterns across the country. Greater availability of land in the western U.S. may allow for much larger farms on average for grazing and pasture, but the concentration of these farms is relatively low compared to densely concentrated crop-producing farms across the midwestern U.S.

In conjunction with a decrease in national cropland and regional variations of farm size and type, U.S. agriculture has become more productive writ large since the 1970s. Total Factor Productivity (TFP) accounts for all of the land, labor, capital, and material resources employed in farm production and then compares them with the total amount of crop and livestock output. If, for instance, total output grows faster than total inputs, the total productivity of the factors of production (i.e., TFP) is increasing. TFP data is only publicly available at the state level from 1960 through 2004. Based on this data, since 1960, every state reflects an *increase* in TFP; no state or region has become less productive (ERS, 2019a). Farms in the Heartland and the Mississippi Portal have become over 100 to 150 percent more productive (see **Supplementary Figure 6**). Meanwhile, the Pacific Northwest portion of the Fruitful Rim and Basin and Range reflect TFP gains between 150 and 200 percent. Other areas in the Basin and Range, particularly throughout Colorado, Kansas, Montana, and Texas, have seen lesser gains but are still ∼50 to 75 percent more productive than 1960. Productivity gains in the Southern Seaboard and the Northern Crescent reflect around a 100 to 125 percent increase on average. These increases are regionally concentrated to reflect the intensification of agricultural production in certain areas, particularly through increases in external inputs (**Figure 8**).

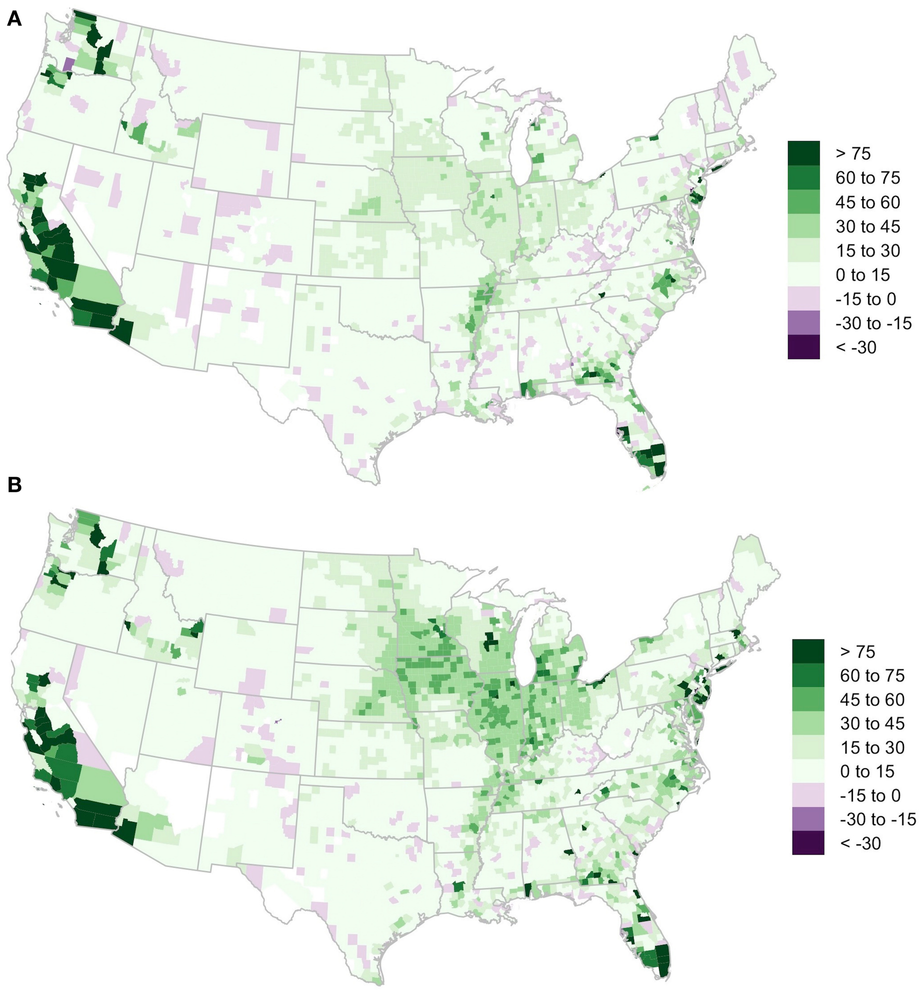


FIGURE 8 | Change (in USD) in inputs per operated acre by county between 1997 and 2017 by county. **(A)** Change (in USD) in chemical expense per operated acre. **(B)** Change (in USD) in fertilizer expense per operated acre. 1997 USD values are adjusted for inflation using average consumer price indices (CPI) from January-December 1997 (avg. CPI ∼ 160.52) and January-December 2017 (avg. CPI ∼ 245.12). Data: USDA NASS Census of Agriculture.

Those same U.S. regions that have realized huge gains in TFP have, at the same time, become more reliant on off-farm inputs like synthetic fertilizers and chemicals. Certain counties in the West Coast portion of the Fruitful Rim and along the Southern Seaboard have increased expenditures on chemicals by, on average, \$30 to over \$75 per acre (**Figure 8A**) and on fertilizers by similar amounts (**Figure 8B**). Areas within the Heartland and Mississippi Portal have largely increased their chemical expenses by \$0 to \$30 per acre (**Figure 8A**) but have increased fertilizer expenses between an average of \$15 to \$45 per acre (**Figure 8B**). These large expenditure changes over the past two decades stand in contrast to places along the Southern Seaboard, within the Basin and Range, and the Prairie Gateway that have maintained spending, only shifting (increased or decreased) by \$15 per acre. Again, these regional differences highlight the resource-intensive crop production practices of select U.S. agricultural regions. Overall, the majority (∼80%) of counties show increasing use of, and expenditure on, synthetic inputs since 1997; few places (only within certain counties in the west and in the Eastern Uplands) have decreased spending per acre. However, since TFP has increased alongside external input use, this suggests that crop yield is rising faster than input use.

Crop Composition

Crop composition has seen drastic changes at a national level as agricultural production has become more productive and input intensive. Since 1963, harvested soybean and corn acreage (although complementary for crop rotation) has increased by 76 percent (74 million acres), while acreage for other feed crops such as oats, barley, sorghum, and hay have declined by a combined 50 million acres (Bigelow and Borchers, 2017). Wheat, once the dominant crop in the U.S., comprises the third largest acreage planted of U.S. crops at 46 million (Ash et al., 2018).

Since the 1970s (and preceding that), the composition of crop acreage (total acres planted per crop) across the U.S. has become increasingly specialized. Demonstrated in **Figure 9**, by 2019, total crop acreage of major crops is nearly dominated by corn, soy, and wheat (winter, spring, and durum). In 1925, corn and wheat comprise a majority of the acreage planted with cotton and oats following closely behind; however, the difference in acreage planted for these crops is comparatively small. From the mid-1920s to the 1970s, acreage for cotton, oats, barley, and peanuts gradually decreases; meanwhile, acreage for soybeans rapidly increases, and wheat and corn acreage remain consistently dominant. From the 1970s through 2019, acres planted for corn, soy, and wheat (particularly soy) increase at the same time other major commodities decrease. Steady declines of the planted acreage of sorghum, cotton, barley, and oats become evident as corn and wheat remain consistent, and soy continues to expand. Meanwhile, acreage of peanuts, canola, and rice remain negligible in a national context (see **Supplemental Figure 7** for separated crop trends). Therefore, the 1970s era onward was characterized by observable specialization toward certain crops. As of 2019, these crops (corn, soy, and wheat) comprise a

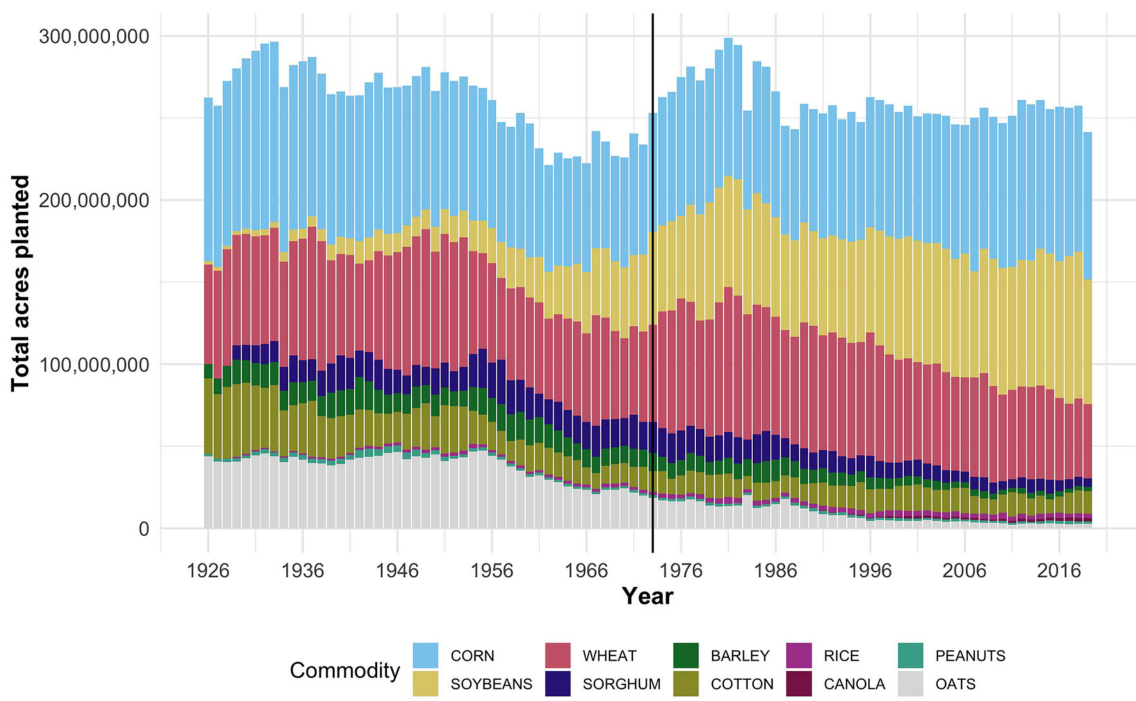


FIGURE 9 | Total acres planted of 10 major U.S. crops between 1920 and 2019. Top 10 crops determined by acres planted in 2019. A vertical line at 1973 indicates the passing of the 1973 Farm Bill and marked transition toward crop specialization. Data: USDA NASS Survey.

total of 210,958,000 planted acres; corn and soy alone cover nearly 166 million. According to the 2017 Census estimates of total cropland in the U.S., corn, soy, and wheat cover 64.7% of harvested cropland acres; corn and soy alone cover 56.6% (USDA NASS, 2019a).

Although the national trend in planted crop acres is dominated by corn, soybeans, and wheat, regional variability of agricultural land use diversity exists. The Shannon's Diversity Index (SDI) is a measure of evenness and abundance of different land use types as a way of measuring ecological diversity in a given area (Gustafson, 1998; Aguilar et al., 2015). **Figure 10** illustrates the SDI per 20 km based on agricultural land use categories as defined by the USDA Cropland Data Layer (CDL) database (only available from 2008 to 2018 thus limiting its historical depth to interpret land use trends over time; Arora and Wolter, 2018) and computed by Burchfield et al. (2019). This index provides a measure of crop diversity for 20-kilometer (km) pixels within a given year. Areas of low diversity (light green) are concentrated in the Heartland and Basin and Range regions. Counties of high diversity (dark blue) are concentrated along the Southern Seaboard, Fruitful Rim of California and the Pacific Northwest, and the Northern Great Plans. Thus, certain agriculturally dominant regions, such as the Heartland, are highly specialized and non-diverse, while others, such as the Fruitful Rim of California, are highly diverse. Such variation in agricultural land use diversity emphasizes the different production systems and agroecological contexts in which crops are grown nationally.

These trends in crop diversity contextualize where the majority of crops that dominate U.S. crop production (as demonstrated from **Figure 9**) are concentrated. **Figure 11** illustrates percent of a county cultivated for the two major crops: corn and soybeans. By visualizing the percent of each county cultivated by these crops in the U.S., regional dominance of this commodity production is evident.

Dominant counties of 40% or higher of cultivated land for each crop largely fall within the midwestern Heartland region. Further, this region has a comparatively lower SDI value (**Figure 10**) than most other productive regions. Yet, areas along the Mississippi Portal and the Prairie Gateway demonstrate dominance of soybean cultivation *and* a comparatively high SDI value. The location of these dominant landscapes further illustrates how and where crop specialization has occurred and continues to occur.

Policy Structure

Agricultural land use changes in the U.S. take place within a policy structure that operates at multiple levels, from local zoning laws to national-level subsidy programs. The U.S. Farm Bill (FB) has become what is referred to as an *omnibus* (or all-encompassing) piece of legislation that largely influences how, where, and why food is produced and distributed; these policies cover an increasingly broad suite of programs and purposes. For example, the 1933 FB, titled the Agricultural Adjustment Act of 1933, aimed specifically to provide relief for farmers in debt and increase agricultural revenue. Its stated purpose is as follows:

> "To relieve the existing national economic emergency by increasing agricultural purchasing power, to raise revenue for extraordinary expenses incurred by reason of such emergency, to provide emergency relief with respect to agricultural indebtedness, to provide for the orderly liquidation of joint-stock land banks, and for other purposes." (Agricultural Adjustment Act, 1933)

Thus, it was a reactionary policy to an ongoing economic crisis. The most recent version of the FB passed in 2018, states its purpose as the following:

> "To provide for the reform and continuation of agricultural and other programs of the Department of Agriculture through fiscal year 2023, and for other purposes." (Agricultural Improvement Act, 2018)

This most recent FB reflects a broader purpose than 1933, maintaining and updating the status quo of the U.S. agricultural system. The goal for "reform and continuation of agricultural programs" emphasizes the growing importance of these programs that regulate how the U.S. agri-food system operates. FB programs currently cover a wide variety of "Titles" or topics in the 2018 policy document; these Titles include: (1) Commodities, (2) Conservation, (3) Trade, (4) Nutrition, (5) Credit, (6) Rural Development, (7) Research, Extension, and related matters, (8) Forestry, (9) Energy, (10) Horticulture, (11) Crop Insurance, and (12) Miscellaneous. This 2018 FB proposed a budget for $428 billion for its 5-year life span, of which 76% is dedicated to Nutrition programs such as the Supplemental Nutrition Assistance Program (SNAP), and a mere 9% is dedicated to crop insurance, 7% for commodities, and 7% for conservation (McMinimy et al., 2019). The importance and composition of these Titles has substantially changed over time, ultimately defining and reinforcing the political structure of agricultural production in the U.S (for a more complete list, see McFadden and Hoppe, 2017, Appendix A).

FB programs have historically aimed to improve agricultural productivity and markets by controlling the supply of commodities. The Emergency Feed Grains Act of 1961 replaced market-oriented policies with direct federal government regulation; this put the federal government in greater control over the driving forces of the production (McGranahan et al., 2013). Following that, the well-known era of "fencerow to fencerow" production of the 1970s was defined by increased supply of agricultural commodities that captured economies of scale to combat high production costs. The "Russian Grain Robbery" of the mid-1970s—in which the Soviet government purchased over one fourth of U.S. wheat harvests to increase their own livestock production—challenged domestic demand for commodities, tripled wheat prices, and doubled corn and soy prices. This market spike led to the export of 80% of wheat in the U.S. to the Soviet Union (Luttrell, 1973). The then Secretary of Agriculture, Earl Butz, supported this international trade market as a way of boosting exports to foreign markets. Therefore, to combat the rise in commodity prices for the U.S., he encouraged farmers to increase their production, aiming to create immediate

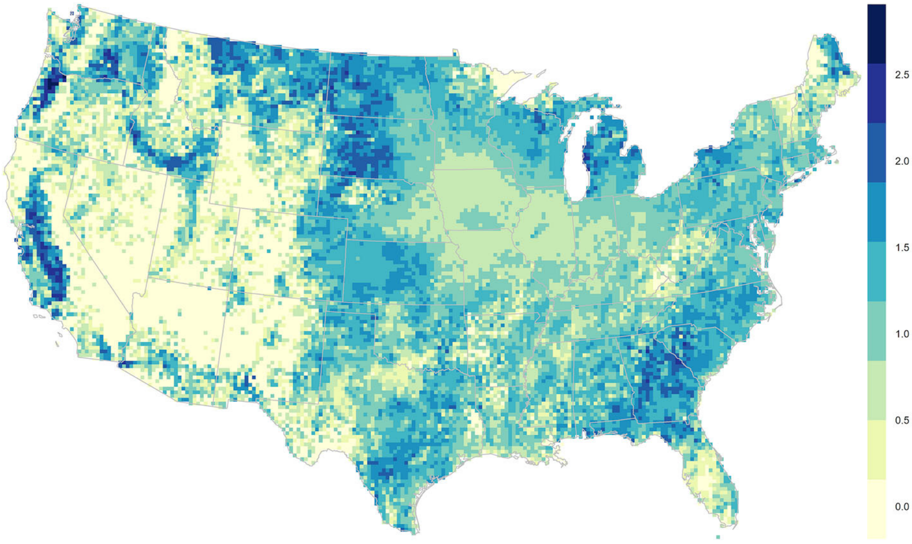


FIGURE 10 | Shannon's Diversity Index (SDI) of agricultural land use categories for each 20-km pixel in the U.S. in 2017. Light green indicates counties with a low SDI and dark blue indicates counties with a high SDI. Reprinted from Burchfield et al. (2019). Copyright (2019), with permission from Elsevier.

surpluses of commodity crops, particularly corn and soybeans (McGranahan et al., 2013). Although overall cropland cultivated did not immediately increase during this era, corn, soy, and wheat production noticeably expanded while production of other crops (e.g., sorghum, barley, oats) declined (see **Figure 9** above, whereby a vertical line at the year 1973 marks this transition). The Agricultural and Food Act of 1981 extended these federal support policies from the 1970s, leading to the 1980s Farm Crisis: the federal government made billions of dollars of payments to farmers growing commodity crops to reduce production, re-adjust commodity prices, and help farmers address rising debt (McGranahan et al., 2013). These federal regulations created incentives for specialized agricultural land use over the past 50 years currently still in effect.

Agricultural land reserve programs have played a role in influencing how and where commodities are produced. From the late 1950s through 1990, the federal government paid farmers to take productive cropland out of production as a means of supply control; this land had to be converted to grassland, trees, or other non-crop purposes (Olson, 2001). The Agricultural Act of 1956 established the Soil Bank Program to set aside 12 million hectares of land from commodity production to be used for wildlife habitat; however, the land enrolled in this early conservation reserve program was already low in productivity. Thus, this type of land reserve program helped regulate the amount of highly productive land used for commodity production by reducing the less productive land competing on the market with more productive land (McGranahan et al., 2013). Meanwhile, in conjunction with technological advances made during the Green Revolution of the 1950s and '60s, productivity of major crops increased on this high-quality land. In 1985, the Conservation Reserve Program (CRP) was established under the Food Security Act of 1985 with aims to reduce soil erosion on highly erodible cropland and reduce off-farm sedimentation, as well as *decrease* commodity surpluses and increase farm income. Further, the "swamp buster" provision was added for environmental protection by disincentivizing farmers from producing agricultural commodities on wetlands after 1985, as this conversion made them ineligible for federal support (Daniels, 1988). While the 1956 Soil Bank Act did not limit the amount of land that could be taken out of production, the 1985 CRP provision limited this amount of land to no more than 25% of a county's total cropland base; this helped minimize large-scale economic impacts on commodity prices and agri-businesses. However, ongoing commodity price support programs have continued to compete with CRP enrollment. Thus, while CRP enrollment has continued since 1985, it has not effectively targeted the most sensitive and erodible land or out-competed other financial incentives for farmers to produce subsidized commodity surpluses (Isik and Yang, 2004; Johnson et al., 2016).

In addition to incentivizing commodity production, FB programs have limited diversification on agricultural lands that are supported by federal subsidies. In the 1985 FB, acreage designated to commodity production was limited by the Acreage Limitation Program (ALP) and Paid Land Diversion Program (PLD); to receive subsidy payments, certain commodities could only be planted on a set amount of acreage. As of the 1996 FB, "production flexibility contracts" (a.k.a. "Freedom to Farm") replaced ALP and PLD to allow farmers to plant different crops other than previously stipulated commodities to increase planting flexibility while still receiving federal support (Willis and O'Brien, 2002). Producers could plant 100% of their contract acreage to a different crop, including grazing or hay production. However, this flexibility was limited; fruit and vegetable production (other than lentils, mung beans, or dry peas) was prohibited, unless a history of double-cropping fruits or vegetables had been established (ERS, 1996). As of 2002, this planting flexibility was replaced with direct

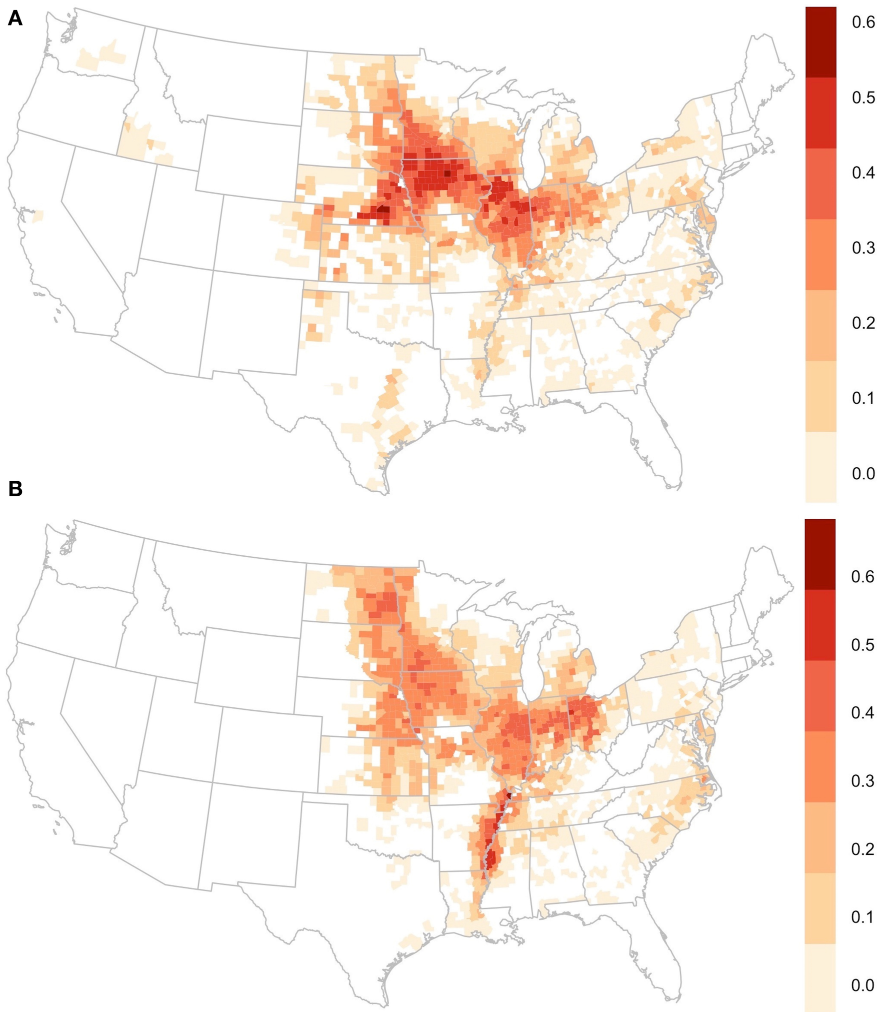


FIGURE 11 | (A) Percent of total county land cultivated with corn in 2017. **(B)** Percent of total county land cultivated with soybeans in 2017. Data: USDA NASS Survey.

payments to farmers for specific crop types and payment rates, regardless of farmer need (Willis and O'Brien, 2002). By 2014, direct payment subsidies were cut from the FB, replaced by several risk management programs (discussed below), but these recent changes do not undo historical incentives for land use specialization.

Further, commodity support programs are only accessible to certain farmers and favor certain types of production. Historically and at present, these programs are only eligible for established base acres. Base acres are defined as farm-level acreage for certain commodities based on the historical average acreage of that commodity; these are the acres eligible for commodity program payments. Therefore, program payments are determined by what *has been* grown on these base acres rather than what is currently being grown. Base acres were established in the 2002 Farm Bill and reflect planted acreage from 1998 to 2001 until the recent opportunity from the 2014 Farm Bill to re-allocate acres based on 2009 to 2012 planting (Farm Bureau, 2016). However, this reallocation did not allow *new* base acres to be designated—only the *adjustment* of designated acres to different commodities. Since base acres are linked to the farm itself, not the farmer, this omits land recently converted to commodity production to be supported by FB commodity payments (Farm Bureau, 2016). This further incentivizes keeping land previously managed for intensive commodity production in the same type of production. Thus, farmers with certain acreage could receive payments for wheat production but not currently produce wheat; contrastingly, acreage under current wheat cultivation *without* base acreage designation could not receive program support. In fact, differences in base acres and *actual* average acreage planted for covered commodities are largely observed across the U.S., maligning the risk mitigation potential of Commodity Title programs with risk experienced by farmers (Newton, 2017). These base acreage designations have not been updated in the 2018 FB, but base acres out of commodity production in the past 10 years are now ineligible for program payments; instead, these base acres can be enrolled in conservation programs, such as the Conservation Stewardship Program (Newton, 2017).

Current Titles established under the 2018 FB reflect past influences of federal agricultural policies and reinforce federal support and influence over the U.S. food system. Although all Titles may influence farmer decision-making and agricultural land use in some way, the Commodity, Credit, Trade, and Crop Insurance Titles (designated as "commodity-focused" Titles hereafter) cover many of the programs that serve to directly mitigate risk through insurance, provide financial assistance and disaster relief through loans and subsidies, and influence market demand through international trade regulations. These Titles are major drivers of the types of commodities produced, as well as where, why, and how this production occurs in present day.

Of these commodity-focused titles, the Commodity Title is the arguably the most influential Title for regulating commodity production and influencing farmer decisions. Commodity programs effectively provide support for market fluctuations and risk associated with commodity production, comprising the majority of influence over agricultural land use. Two main programs under this title include the Price Loss Coverage (PLC) program and the Agricultural Risk Coverage (ARC) program and are administered through the Farm Service Agency (FSA). The PLC, based on a certain crop-year price, pays farmers with historical base acres eligible for covered commodities when the market-based effective price falls below the effective reference price—a price determined by the 2014 FB that allows for market fluctuations (ERS, 2019b). ARC pays farmers with historical base acres when the actual yield (distinguished between irrigated and non-irrigated acres) and prices for their county's average per-acre crop year revenue falls below the guaranteed level for each covered commodity. Commodities covered by both of these programs are defined as wheat, oats, barley, corn, grain sorghum, rice, soybeans, sunflower seed, rapeseed, canola, safflower, flaxseed, mustard seed, crambe and sesame seed, dry peas, lentils, small chickpeas, large chickpeas, and peanuts. As of the 2018 FB, farmers can switch between PLC and ARC programs with greater flexibility. Other programs include the Non-insured Crop Disaster Assistance Program (NAP), Non-recourse Marketing Assistance Loan Program (MAL), and the Dairy Margin Coverage Program (DMC). NAP provides risk protection for crops not covered under the Federal Crop Insurance Program. MAL offers farmers short-term loans when market prices are at their lowest (during harvest time) to allow them to wait and sell their commodity when prices improve. Eligible commodities for MAL include wheat, corn, sorghum, barley, oats, upland and extra-long-staple cotton, long- and medium-grain rice, soybeans/other oilseeds, certain pulses, peanuts, sugar, honey, wool, and mohair. DMC offers coverage for dairy producers when the margin between the price of all milk and the average feed price is below a producer-determined threshold to help manage the fluctuations of the dairy market (ERS, 2019b). These programs largely aim to mitigate risk for farmers, as opposed to control supply of commodities.

Other commodity-focused Titles serve different yet complementary purposes. The Crop Insurance Title updates, modifies, and enacts the Federal Crop Insurance Program (FCIP) whereby farmers can access subsidies to protect against yield, crop revenue, and whole-farm revenue (WFA) losses (Johnson and Monke, 2019). Yield and crop revenue insurance coverage is crop-specific, whereby WFA covers the expected income of an entire farm to support more diversified systems. These insurance products are administered through the Risk Management Agency (RMA) and coverage extends across row crops, livestock, dairy, organic production, other specialty crops, grazing land, etc. (ERS, 2019b). The Trade Title reinforces global markets for U.S. grown crops and largely influences international food prices for U.S. farmers (ERS, 2019b; Johnson and Monke, 2019). Finally, the Credit Title provides direct government loans to farmers and ranchers through the FSA to support beginning, socially disadvantaged, and veteran producers (ERS, 2019b; Johnson and Monke, 2019).

As the structure of each FB has changed over time, the number of crops and commodities included in commodity-focused Titles and programs has increased. **Figure 12** illustrates the *distinct* number of crops and commodities in such Titles of each FB over time. This numeric measure helps illustrate both the broadening scope of the policy itself, as well as the diversity of crops included within FB programs that aim to support and regulate their production.

The 1933 FB only mentions eight distinct crops and animal products (cotton, wheat, rice, corn, tobacco, hogs, milk, and fruit groves/orchards) in its entire 25 pages, demonstrating its limited and reactionary purpose. Contrastingly, the 2018 FB mentions 52 distinct crops across 529 pages—a product of a gradual expansion in scope and influence over time. The highest number of crops mentioned is 81 in both the 2002 and 2008 FBs. Crops classified as fruits or vegetables were not recognized or mentioned in the documents until the 1980s; crops for biofuel or organic production were not introduced until the late 1980s, as well. Further, while the number of crops and commodities within the FB increased from the 1970s onward, the composition of U.S. crop acreage became increasingly less diverse (as seen in **Figure 9** above); these political and ecological changes occurred in tandem, suggesting that the increasing scope of the FB supported such specialization.

DISCUSSION

We discuss the implications of these results in the context of recent literature and the concern for transitioning the U.S. agricultural system toward greater sustainability. The discussion is structured to mirror the results section and contextualize the above data trends. We conclude with recommendations within the broader framework of sustainable agricultural transitions and future research.

Cropland, Farm Size, and Productivity

In recent decades, U.S. agricultural production has reaped the benefits of industrialization and mechanization to support exponential increases in yield of major crops (Reganold et al., 2011; Aguilar et al., 2015; Pellegrini and Fernández, 2018). Although total land area devoted to agriculture is declining

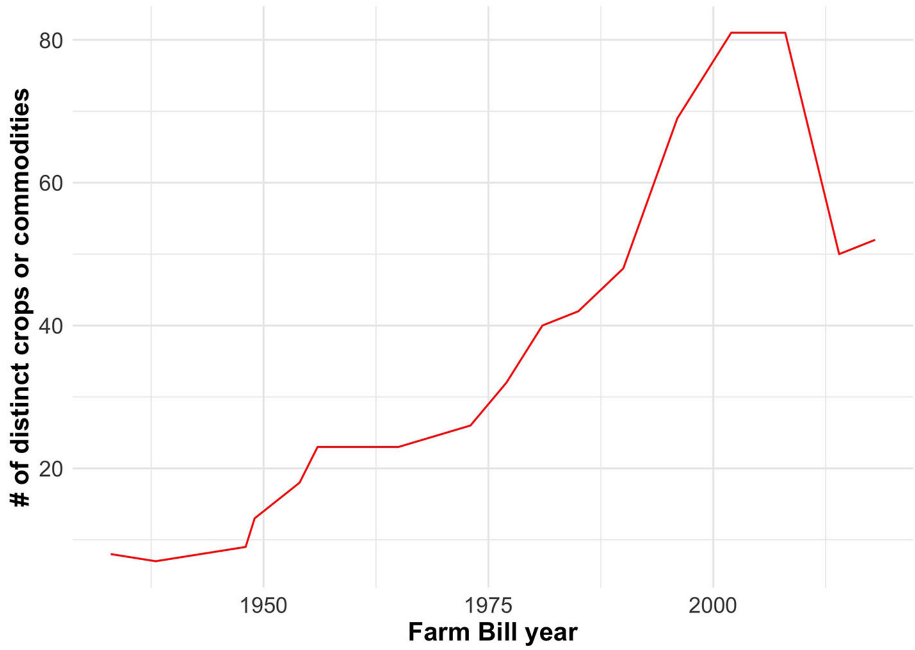


FIGURE 12 | Number of distinct crops or commodities included in the Farm Bill Commodities, Trade, Credit, and Crop Insurance Titles (i.e., commodity-focused Titles). Data: U.S. Farm Bills.

nationally (yet expanding globally, see Ramankutty et al., 2018), crop production is heavily concentrated in certain areas. Larger farms are consolidating, and competition for farmland among farmers is increasing (USDA NASS, 2019b). These large-scale farms are comprising more and more of U.S. cropland and are out-competing smaller operations (Paul et al., 2004; MacDonald and Hoppe, 2017); this consolidation is driven by historical patterns of land dispossession and predominantly White landownership (Dunbar-Ortiz, 2014; Ayazi and Elsheikh, 2015; Horst and Marion, 2019), as well as farmers expanding through part-ownership and operating rented land (Hart, 1991). At the same time, larger farms have brought economies of *scale* that boost productivity (Paul et al., 2004) and benefit from economies of *size* that make it profitable to expand farm size per unit of output (Duffy, 2009). Agglomeration of agricultural production around similar land uses and crop types reflects the pressure for farms to consolidate input investments, share information, and overcome the scalar thresholds of market competition.

While biophysical differences and political incentives influence regional specialization of crop production (Hart, 1978, 1986, 2001), county-level dominance of cropland in areas such as the Heartland, Basin and Range, and Mississippi Portal signifies the simplification and intensification of agricultural landscapes. The Corn Belt, originating from a landscape of mixed farming and agricultural experimentation, has become highly specialized for surplus commodity production (Hart, 1986; Hudson, 1994). The location of farms and cropland in the Heartland has remained relatively stable over the past several decades, indicating that the highest quality and most productive agricultural lands have stayed in agriculture throughout the region (Hart, 1986, 1991; Drummond et al., 2012). Other regions across the western U.S. have seen fluctuations in amount and location of cropland due to greater climatic, economic, and technological variability, as well as changing FB policies (Hart, 2001; Drummond et al., 2012). National evidence of productivity growth, particularly in the Midwest, indicate that farm consolidation is a substantial factor in the exponential increase of aggregate TFP, alongside technological innovation (Key, 2019).

Technological advances in seed genomics, fertilizers, chemicals, and mechanization have revolutionized agriculture in the U.S., but they have also introduced complicated ecological consequences. The introduction of herbicide-resistant (HR) genetically engineered crops in 1996 made the broad-spectrum application of glyphosate possible. Glyphosate-resistant HR crops have necessarily increased the application rates of herbicides and pesticides, introducing resistance in weed and insect populations; meanwhile, populations of beneficial species are decreasing (Benbrook, 2012; Pimentel and Burgess, 2014). Innovations in low-cost synthetic fertilizers in the 1950s and '60s made integrated crop-livestock farming and nutrient recycling biologically obsolete (Davis et al., 2012). Farmer reliance on synthetic fertilizers has increased due to soil fertility declines, yet evidence suggests that synthetic nitrogen depletes soil organic matter, a key indicator of soil health (Mulvaney et al., 2009). Labor efficiency increased with mechanization, and synthetic fertilizers and chemical inputs became increasingly available; meanwhile, specialization of crop and livestock production became more economically viable and efficient. Agricultural research has enabled corn, soy, and wheat to be highly productive per acre harvested. In the 2017/18 season, corn and soy provided $232 and $287 net returns per acre, respectively, and wheat provided $98 per acre (Ash et al., 2018). Yields of these crops and commodities have seen exponential increases prior to and following the Green Revolution in certain areas (e.g., the Corn Belt) yet have begun to plateau in others (e.g., fringes of the Corn Belt) (Hart, 1986; Ray et al., 2012; Pellegrini and Fernández, 2018). These advances led to increasing economies of scale, captured in the growth of farm size, shifts in farm infrastructure toward specialization, and a rapid decline in the number of farms across the U.S. (Hart, 1986; Dimitri et al., 2005).

Trends in national cropland reflect a "land-sparing" approach—less land used more *intensively* for increasing productivity and specialization—compared to a "land-sharing" approach—more land used more *extensively* to manage greater diversity of land use (Phalan et al., 2014). These different approaches to land management have been hotly debated regarding conservation and long-term sustainability (Fischer et al., 2008, 2014). As the U.S. trends toward greater specialization in agricultural production, this puts greater pressure on effective biodiversity conservation of non-agricultural land. Furthermore, this specialization holds implications for the sensitivity and resilience of agricultural production within an increasingly uncertain climate (Ortiz-Bobea et al., 2018) and increasing reliance on external mechanization (Rada and Fuglie, 2019). Such changes could increase farmer debt and put greater pressure on rural economies. These implications heighten concern over the long-term management of the ecological health of agricultural land within the context of increasing input use, machinery, and decreasing intra-crop and inter-crop species diversity within and across farms.

Crop Composition

In the U.S., the diversity of agricultural crops cultivated has decreased since the 1970s with wide regional differences. Regions that are most productive for dominant crops (i.e., corn and soybeans) maintain the least crop species diversity. Certain areas, such as Mississippi Portal Region, have maintained higher crop species diversity, whereby other areas, such as the Heartland region, have become largely optimized for a select few crops and commodities through decreasing diversity (Hart, 1986; Aguilar et al., 2015; Baines, 2015; Auch et al., 2018). Similarly, on a global scale, agricultural land has become dominated by a less diverse portfolio of crops (Martin et al., 2019).

Effects of declining crop species diversity raise concerns over the long-term health of agricultural ecosystems, as well as the stability of agricultural economies over time. Crop species diversity can be assessed at an on-farm and landscape level and holds different implications for land management. Increasing crop species diversity at a landscape level through compositional heterogeneity (i.e., the distinct number of crop types across a landscape) may have significant beneficial impacts on yield of major crops like corn and soy (Burchfield et al., 2019). Increasing configurational heterogeneity (i.e., the spatial arrangement of crop types and land uses) can boost pollinators and plant reproduction for small-scale farms (Hass et al., 2018). Further, increasing farm-scale diversity can improve the resilience and stability of agricultural production over time (Abson et al., 2013). Although some U.S. regions are much less diverse than others, maintaining crop diversity at local, national, and global scales is of great importance to achieve and maintain food security for the future (Massawe et al., 2016).

Managing on-farm and landscape-scale crop species diversity comes with a suite of considerations. Assuming that farmers aim to reduce risk in their operations, diverse cropping systems and practices have been positively linked to increased mean income and reduced income variance over time (Di Falco and Perrings, 2003). Crop diversity is known to enhance ecosystem services (ES) such as soil health, pest management, and water quality (Tscharntke et al., 2005, 2012; Hendrickx et al., 2007; Meehan et al., 2011; Bommarco et al., 2013; McDaniel et al., 2014; Landis, 2017), but these ecological benefits must also complement, if not enhance, other benefits for farmer livelihoods. Increasing crop diversity through practices such as crop rotation (over several seasons), intercropping (within one season), non-crop vegetation (such as filter strips or wildlife habitat), or integrated pest management pose challenges and barriers to their adoption; these include learning new management skills, balancing the potential risk on yield of major crops, or accessing appropriate machinery or technology to implement them effectively (Way and van Emden, 2000; Hooper et al., 2005; Pridham and Entz, 2008; NRC, 2010). Furthermore, these incentives and disincentives are filtered through federal agricultural policies that offer competing financial support. Biodiversity management on farms and across landscapes must be contextualized through such overlapping political, ecological, and social constraints.

Policy Structure

Federal agricultural policy has increased in scope since 1933 and maintains considerable influence. In fact, through this increase, the federal government is the primary source of supplemental income for farmers through subsidy payments (O'Connor, 2012). While the purpose of the FB has changed significantly since 1933, the incentive structure has not, prioritizing commodity production over both conservation practices (Lehner and Rosenberg, 2018) and agricultural diversification, even when the cost of production has exceeded farmer revenue (Hart, 1986). Even though the number of crops indicated in each commodity-focused FB Title has increased, the national crop portfolio has become increasingly less diverse. This misalignment between the diversity of crops regulated or supported by FB programs and the non-diversity of U.S. crop production highlights how policy ultimately promotes specialized commodity production. While environmental concerns arise over such land use trends, the implications of these federal policies are mixed.

Increasing federal control over and support of agricultural production has been debated in recent literature, particularly if and how it may promote or inhibit greater sustainability for both farmer livelihoods and ecological health. Evidence supports that U.S. agricultural subsidies are less accessible to smaller, organic, or diversified farming operations, fail to encourage conservation practices, promote commodity specialization (Bruckner, 2016), and systemically privilege

White landowners over marginalized farmers and farmworkers (Dunbar-Ortiz, 2014; Ayazi and Elsheikh, 2015; Minkoff-Zern and Sloat, 2017). While subsidies and financial assistance may help mitigate risk associated with crop diversification for farmers, it has also been shown to *discourage* diversification and support specialized commodity production (Di Falco and Perrings, 2005). Since crop insurance helps mitigate the need for income variation, farmers may rely less on diversifying their farming systems to reduce this risk (O'Donoghue et al., 2009). Growing federal support for risk mitigation programs—such as ARC, PLC, and crop insurance programs—further decouples farmer decision-making from environmental risk. Although crop insurance enrollment does not lead to greater nutrient use through fertilizers and other chemicals (Weber et al., 2016), recent studies have shown that crop insurance increases irrigation withdrawals across the U.S. by motivating farmers to grow more water-intensive crops (Deryugina and Konar, 2017). Furthermore, farmers enrolled in crop insurance were found to experience greater yield sensitivity of corn and soy in extreme heat than those not insured; thus, crop insurance could provide a disincentive to take adaptive measures against climate-related impacts (Annan and Schlenker, 2015).

Despite these limitations, removing or decreasing federal agricultural assistance as an alternative is associated with several tradeoffs. In fact, this reduction may actually support farm consolidation. Large farms can more easily access crop insurance (due to access to greater capital) than small and medium size farms (Bruckner, 2016; Graddy-Lovelace and Diamond, 2017); this reinforces barriers for disadvantaged, small-scale, or aspiring farmers (Calo and De Master, 2016; Rosenberg and Stucki, 2017; Horst and Marion, 2019). Examples of subsidy reduction outside of the U.S. exhibit mixed results. Subsidy removal in Canada has been associated with increased specialization of production (Bradshaw, 2004), while New Zealand has seen increased farm diversification and off-farm income for farmers (Vitalis, 2007). Some argue that focusing the political debate around agricultural subsidies distracts policymakers from intervening in agricultural markets in necessary yet beneficial ways (Graddy-Lovelace and Diamond, 2017). Therefore, increased agricultural subsidies do not presume to move *away* from agricultural sustainability, but rather the type and incentive of such policies should be questioned.

CONCLUSION

Overarchingly, the U.S. agricultural system has gradually transitioned toward a regulated and specialized system, recognized through consolidation of U.S. farms and the homogenization of crop production. Fewer and fewer farms own more and more land, and these farms continue to produce a select few crops within highly mechanized processes. These changes emphasize productivity and efficiency, despite increasing concern for biodiversity loss. Further, even though the Farm Bill has increased in scope, the underlying structures incentivizing and reinforcing agricultural specialization have not changed.

While we do not attempt to assess the current sustainability of U.S. agriculture within the NRC's definition, historical data trends accentuate the priorities of the production system writ large. Through substantial gains in productivity and specialization of commodities across the U.S., past and current agricultural land use largely reflect two of the sustainability objectives: (1) satisfying human food, feed, fiber, and biofuel needs; and (2) sustaining the economic viability of agriculture. However, the prioritization of sufficient production and its economic viability has come at the cost of the other outlined objectives: (3) enhancing environmental quality and the resource base; and (4) enhancing the quality of life for farmers, farm workers, and society as a whole. Intensive commodity production has concentrated in space and contributes to biodiversity loss and declining agroecosystem health. These systems often fail to promote farming that *harnesses* and *enhances* ES provisioning and are increasing reliance on external inputs instead. Meanwhile agricultural policies are not equally as advantageous or accessible to *all* producers, exacerbating social inequities and disadvantaging new or diverse farmers. The imbalance of these objectives heightens concern over the robustness of the system. Decreasing trends in crop diversity may contribute to decreased resistance and resilience to shocks and stressors associated with a changing climate and changing environments, and the adaptability needed to address urgent changes may be limited by an increasingly regulatory policy structure.

Within the NRC framework of change, *both* incremental and transformative approaches to change are necessary to promote more sustainable agricultural systems. For large-scale landscape transformations to occur, agricultural research and technological innovation must focus on commercial grain producers; this is how the majority of the agricultural land is used. To implement transformative change without destabilizing crop markets would be difficult. However, given how large these agricultural landscapes are, *any* change in their compositional (increased complexity of different land cover types) and configurational (increased complexity of spatial patterning of cover types) heterogeneity can produce important changes in biodiversity for local or global conservation (Fahrig et al., 2011); changes outside of these markets will not have the largest transformative impact. Therefore, incremental approaches could best support technological advancements and innovations already available for land management by building off current research and enhancing adoption for existing conservation alternatives. Transformative change could target restrictive policies—such as updating base acreage designations or reducing barriers for non-White or small-scale farmers—to encourage more flexible and diverse programs that support commodity production. Federal agricultural policy at present fails to effectively *promote* diversification or conservation practices; whether increased or decreased federal support will do so is currently debated. Yet, a more diverse and socially inclusive suite of programs can help support more diverse systems in which these commodities are grown, promoting technological innovations that can reduce the impacts of agricultural landscape simplification. If large farms and corporate entities remain consistently advantaged over small farms and businesses, then alternative agricultural management schemes will be limited.

We have built upon the NRC (2010) report discussing the complicated nature of evaluating sustainability within agricultural systems. By utilizing national-level data to look at trends of land use and policy over time, we inform and update previous research to remain contextually relevant for policy decisions and assess U.S. trends writ large. Agricultural transformations toward sustainability do not fit within the dichotomy of conventional or sustainable systems. Rather, considering drivers and constraints across multiple scales helps identify realistic pathways of change. For a more sustainable future, both incremental and transformative changes are needed to address the proximate and ultimate conditions of the current state of agricultural landscapes. Although crop composition, productivity, and farm consolidation trends vary regionally, agricultural policy is regulated at a federal level. Therefore, we call for federal agricultural policies to more appropriately address the current drivers of on-farm and landscape simplification, as well as the overlapping factors of sustainability from the local to global scale to contextualize the feasibility of agricultural transitions.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. These data can be found here: https://quickstats.nass.usda.gov/; https://nationalaglawcenter.org/farmbills/; https://www.ers.usda.gov/data-products/major-land-uses/; https://pubs.er.usgs.gov/publication/ds948; https://www.ers.usda.gov/data-products/agricultural-productivity-in-the-us/. All R scripts will be provided by the authors upon reasonable request.

AUTHOR CONTRIBUTIONS

KS and EB contributed conception and design of the study. All authors organized and visualized data. KS led the writing of the manuscript, while all authors contributed to manuscript revision, and have read and approved the submitted version.

FUNDING

This research was supported by funds from EB's start-up funds from the Department of Environment and Society at Utah State University (USU), USU's Climate Adaptation Science Program through the National Science Foundation under Grant No. 1633756, and the Utah Agricultural Experiment Station under Grant No. UTA01422.

ACKNOWLEDGMENTS

We would like to thank Stan Rhodes for his assistance with clarifying our policy analysis approach and Emma Jones for downloading and organizing data.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.00098/full#supplementary-material

REFERENCES

Abson, D. J., Fraser, E., and Benton, T. G. (2013). Landscape diversity and the resilience of agricultural returns: a portfolio analysis of land-use patterns and economic returns from lowland agriculture. *Agric. Food Secur.* 2:2. doi: 10.1186/2048-7010-2-2

Aguilar, J., Greta, G. G., Hendrickson, J. R., Archer, D. W., Forcella, F., and Liebig, M. A. (2015). Crop species diversity changes in the United States: 1978-2012. *PLoS ONE* 10:e0136580. doi: 10.1371/journal.pone.0136580

Annan, F., and Schlenker, W. (2015). Federal crop insurance and the disincentive to adapt to extreme heat. *Am. Econ. Rev.* 105, 262–266. doi: 10.1257/aer.p20151031

Arora, G., and Wolter, P. (2018). Tracking land cover change along the western edge of the U.S. corn belt from 1984 through 2016 using satellite sensor data: observed trends and contributing factors. *J. Land Use Sci.* 13, 59–80. doi: 10.1080/1747423X.2018.1466001

Ash, M., Bond, J. K., Capehart, T., Childs, N. W., McConnell, M., Meyer, L., et al. (2018). *An Exploration of Crop Markets: A Deeper Look Into the USDA Crop Baseline Projections*. FDS-18J-01. U.S. Department of Agriculture, Economic Research Service.

Auch, R. F., and Karstensen, K. A. (2015). *Status and Trends of Land Change in the Midwest-South Central United States-1973 to (2000).* U.S. Geological Survey Professional Paper 1794-C. Land Cover Trends (Washington, DC: USGS).

Auch, R. F., Xian, G., Laingen, C. R., Sayler, K. L., and Reker, R. R. (2018). Human drivers, biophysical changes, and climatic variation affecting contemporary cropping proportions in the northern prairie of the U.S. *J. Land Use Sci.* 13, 32–58. doi: 10.1080/1747423X.2017.1413433

Ayazi, H., and Elsheikh, E. (2015). *The Farm Bill: Corporate Power and Structural Racialization in the United States Food System*. Berkeley, CA: Haas Institute for a Fair and Inclusive Society at the University of California Berkeley. Available online at: https://belonging.berkeley.edu/global-justice/glocal-food-systems/farm-bill-report-corporate-power-and-structural-racialization-us-food-system

Baines, J. (2015). Fuel, feed and the corporate restructuring of the food regime. *J. Peasant Stud.* 42, 295–321. doi: 10.1080/03066150.2014.970534

Barnard, C. H., and Hexem, R. (1988). *Major Statistical Series of the U.S. Department of Agriculture: Land Values and Land Use*, AH-671, Vol. 6. US Department of Agriculture, Economic Research Service.

Benbrook, C. M. (2012). Impacts of genetically engineered crops on pesticide use in the U.S. - The first sixteen years. *Environ. Sci. Eur.* 24:24. doi: 10.1186/2190-4715-24-24

Benton, T. G., Vickery, J. A., and Wilson, J. D. (2003). Farmland biodiversity: is habitat heterogeneity the key? *Trends Ecol. Evol.* 18, 182–188. doi: 10.1016/S0169-5347(03)00011-9

Bianchi, F. J. J. A., Booij, C. J. H., and Tscharntke, T. (2006). Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proc. R. Soc. B Biol. Sci.* 273, 1715–1727. doi: 10.1098/rspb.2006.3530

Bigelow, D. P., and Borchers, A. (2017). *Major Uses of Land in the United States (2012)*. EIB-178. U.S. Department of Agriculture, Economic Research Service.

Bommarco, R., Kleijn, D., and Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends Ecol. Evol.* 28, 230–238. doi: 10.1016/j.tree.2012.10.012

Bommarco, R., Vico, G., and Hallin, S. (2018). Exploiting ecosystem services in agriculture for increased food security. *Glob. Food Secur.* 17, 57–63. doi: 10.1016/j.gfs.2018.04.001

Bradshaw, B. (2004). Plus c'est la même chose? questioning crop diversification as a response to agricultural deregulation in Saskatchewan, Canada. *J. Rural Stud.* 20, 35–48. doi: 10.1016/S0743-0167(03)00033-0

Bruckner, T. (2016). Agricultural subsidies and farm consolidation. *Am. J. Econ. Sociol.* 75, 623–648. doi: 10.1111/ajes.12151

Brym, Z. T., and Reeve, J. R. (2016). "Agroecological principles from a bibliographic analysis of the term agroecology," in *Sustainable Agriculture Reviews*, Vol. 19, ed E. Lichtfouse (Cham: Springer International Publishing), 203–231.

Burchfield, E. K., Nelson, K. S., and Spangler, K. A. (2019). The impact of agricultural landscape diversification on U.S. crop production. *Agric. Ecosyst. Environ.* 285:106615. doi: 10.1016/j.agee.2019.106615

Calo, A., and De Master, K. (2016). After the incubator: factors impeding land access along the path from farmworker to proprietor. *J. Agric. Food Syst. Commun. Dev.* 6, 1–17. doi: 10.5304/jafscd.2016.062.018

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., et al. (2012). Biodiversity loss and its impact on humanity. *Nature* 486, 59–67. doi: 10.1038/nature11148

Cassman, K. G. (1999). Ecological intensification of cereal production systems: yield potential, soil quality, and precision agriculture. *Proc. Natl. Acad. Sci. U.S.A.* 96, 5952–5959. doi: 10.1073/pnas.96.11.5952

Cozen, M. (2010). *The Making of the American Landscape*. Routledge. Available online at: https://ebookcentral-proquest-com.dist.lib.usu.edu/lib/usu/detail.action?docID=1702299

Daniels, T. L. (1988). America's conservation reserve program: rural planning or just another subsidy? *J. Rural Stud.* 4, 405–411. doi: 10.1016/0743-0167(88)90008-3

Davis, A. S., Hill, J. D., Chase, C. A., Johanns, A. M., and Liebman, M. (2012). Increasing cropping system diversity balances productivity, profitability and environmental health. *PLoS ONE* 7:e47149. doi: 10.1371/journal.pone.0047149

Deryugina, T., and Konar, M. (2017). Impacts of crop insurance on water withdrawals for irrigation. *Adv. Water Resour.* 110, 437–444. doi: 10.1016/j.advwatres.2017.03.013

Di Falco, S., and Perrings, C. (2003). Crop genetic diversity, productivity and stability of agroecosystems. a theoretical and empirical investigation. *Scott. J. Polit. Econ.* 50, 207–216. doi: 10.1111/1467-9485.5002006

Di Falco, S., and Perrings, C. (2005). Crop biodiversity, risk management and the implications of agricultural assistance. *Ecol. Econ.* 55, 459–466. doi: 10.1016/j.ecolecon.2004.12.005

Dimitri, C., Effland, A. B. W., and Conklin, N. C. (2005). *The 20th Century Transformation of U.S. Agriculture and Farm Policy*. USDA ERS. Available online at: http://ageconsearch.umn.edu/bitstream/59390/2/eib3.pdf

Drummond, M. A., Auch, R., Karstensen, K. A., Sayler, K., Taylor, J., and Loveland, T. R. (2012). Land change variability and human-environment dynamics in the United States great plains. *Land Use Policy* 29, 710–723. doi: 10.1016/j.landusepol.2011.11.007

Drummond, M. A., and Loveland, T. R. (2010). Land-use pressure and a transition to forest-cover loss in the Eastern United States. *Bioscience* 60, 286–298. doi: 10.1525/bio.2010.60.4.7

Duffy, M. (2009). Economies of size in production agriculture. *J. Hunger Environ. Nutr.* 4, 375–392. doi: 10.1080/19320240903321292

Dunbar-Ortiz, R. (2014). *An Indigenous Peoples' History of the United States*. Boston, MA: Beacon Press.

ERS (1996). *1996 FAIR Act Frames Farm Policy for 7 Years.* Agricultural Outlook Supplement April (1996). Washington, DC: ERS.

ERS (2000). *Farm Resource Regions. Agricultural Information Bulletin 760*. Washington, DC: USDA Economic Research Service. Available online at: https://www.ers.usda.gov/webdocs/publications/42298/32489_aib-760_002.pdf?v=42487

ERS (2019a). *Agricultural Productivity in the U.S.* USDA Economic Research Service. Available online at: https://www.ers.usda.gov/data-products/agricultural-productivity-in-the-us/

ERS (2019b). *Agriculture Improvement Act of 2018: Highlights and Implications*. Washington, DC: US Department of Agriculture, Economic Research Service. Avaliable online at: https://www.ers.usda.gov/agriculture-improvement-act-of-2018-highlights-and-implications/

Fahrig, L., Baudry, J., Brotons, L., Burel, F. G., Crist, T. O., Fuller, R. J., et al. (2011). Functional landscape eterogeneity and animal biodiversity in agricultural landscapes: heterogeneity and biodiversity. *Ecol. Lett.* 14, 101–112. doi: 10.1111/j.1461-0248.2010.01559.x

Falcone, J. A. (2015). *U.S. Conterminous Wall-to-Wall Anthropogenic Land Use Trends (NWALT), 1974-2012*. U.S. Geological Survey Data Series 948, 33 p. plus appendixes 3-6 as separate files. doi: 10.3133/ds948

Farm Bureau (2016). *Options for Updating Bases and Yields*. Farm Bureau: 2018 Farm Bill. Washington, DC: Farm Bureau. Avaliable online at: https://www.fb.org/files/2018FarmBill/Options_for_Updating_Bases_and_Yields.pdf

Fischer, J., Abson, D. J., Butsic, V., Chappell, M. J., Ekroos, J., Hanspach, J., et al. (2014). Land sparing versus land sharing: moving forward. *Conserv. Lett.* 7, 149–157. doi: 10.1111/conl.12084

Fischer, J., Brosi, B., Daily, G. C., Ehrlich, P. R., Goldman, R., Goldstein, J., et al. (2008). Should agricultural policies encourage land sparing or wildlife-friendly farming? *Front. Ecol. Environ.* 6, 380–385. doi: 10.1890/070019

Francis, C. A., Lieblein, G., Gliessman, S. R., Breland, T. A., Creamer, N., Harwood, R., et al. (2003). Agroecology: the ecology of food systems. *J. Sustain. Agric.* 22, 99–118. doi: 10.1300/J064v22n03_10

Graddy-Lovelace, G., and Diamond, A. (2017). From supply management to agricultural subsidies-and back again? the US farm bill & agrarian inviability. *J. Rural Stud.* 50, 70–83. doi: 10.1016/j.jrurstud.2016.12.007

Gustafson, E. J. (1998). Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems* 1:15. doi: 10.1007/s100219900011

Harlan, J. R. (1975). Our vanishing genetic resources. *Science* 188, 617–621. doi: 10.1126/science.188.4188.617

Hart, J. F. (1978). Cropland concentrations in the South. *Ann. Assoc. Am. Geogr.* 68, 505–517. doi: 10.1111/j.1467-8306.1978.tb01213.x

Hart, J. F. (1986). Change in the corn belt. *Geogr. Rev.* 76:51. doi: 10.2307/214784

Hart, J. F. (1991). Part-ownership and farm enlargement in the Midwest. *Ann. Assoc. Am. Geogr.* 81, 66–79. doi: 10.1111/j.1467-8306.1991.tb01679.x

Hart, J. F. (2001). Half a century of cropland change. *Geogr. Rev.* 91:525. doi: 10.2307/3594739

Hart, J. F. (2004). *The Changing Scale of American Agriculture*. Charlottesville, VA: University of Virginia Press.

Hass, A. L., Kormann, U. G., Tscharntke, T., Clough, Y., Baillod, A. B., Sirami, C., et al. (2018). Landscape configurational heterogeneity by small-scale agriculture, not crop diversity, maintains pollinators and plant reproduction in Western Europe. *Proc. R. Soc. B Biol. Sci.* 285:20172242. doi: 10.1098/rspb.2017.2242

Heal, G., Walker, B., Levin, S., Arrow, K., Dasgupta, P., Daily, G., et al. (2004). Genetic diversity and interdependent crop choices in agriculture. *Resour. Energ. Econ.* 26, 175–184. doi: 10.1016/j.reseneeco.2003.11.006

Hendrickx, F., Maelfait, J. P., Van Wingerden, W., Schweiger, O., Speelmans, M., Aviron, S., et al. (2007). How landscape structure, land-use intensity and habitat diversity affect components of total arthropod diversity in agricultural landscapes: agricultural factors and arthropod biodiversity. *J. Appl. Ecol.* 44, 340–351. doi: 10.1111/j.1365-2664.2006.01270.x

Hillier, J., Hawes, C., Squire, G., Hilton, A., Wale, S., and Smith, P. (2009). The carbon footprint of food crop production. *Int. J. Agric. Sustain.* 7, 107–118. doi: 10.3763/ijas.2009.0419

Homer, C., Dewitz, J., Yang, L., Jin, S., Danielson, P., Coulston, J. W., et al. (2015). Completion of the 2011 National land cover database for the conterminous United States - representing a decade of land cover change information. *Photogramm. Eng. Remote Sensing* 81, 345–354. doi: 10.14358/PERS.81.5.345

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E. K., Hungate, B. A., Matulich, K. L., et al. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature* 486, 105–108. doi: 10.1038/nature11118

Hooper, D. U., Chapin, F. S., Ewel, J. J, Hector, A., Inchausti, P., Lavorel, S., et al. (2005). Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecol. Monogr.* 75, 3–35. doi: 10.1890/04-0922

Hoppe, R. A. (2014). *Structure and Finances of US Farms: Family Farm Report, 2014 Edition. Economic Information Bulletin No. (EIB-132)*. Economic Research Service.

Horrigan, L., Lawrence, R. S., and Walker, P. (2002). How sustainable agriculture can address the environmental and human health harms of industrial agriculture. *Environ. Health Perspect.* 110, 445–456. doi: 10.1289/ehp.02110445

Horst, M., and Marion, A. (2019). Racial, ethnic and gender inequities in farmland ownership and farming in the US. *Agric. Hum. Values* 36, 1–16. doi: 10.1007/s10460-018-9883-3

Hudson, J. C. (1994). *Making the Corn Belt: A Geographical History of Middle-Western Agriculture*. Bloomington, IN: Indiana University Press.

Hudson, J. C., and Laingen, C. R. (2018). *American Farms, American Food: A Geography of Agriculture and Food Production in the United States*. Lanham, MD: Lexington Books.

Isik, M., and Yang, W. (2004). An analysis of the effects of uncertainty and irreversibility on farmer participation in the conservation reserve program. *J. Agric. Resour. Econ.* 29:18. doi: 10.22004/ag.econ.31113

Johnson, K. A., Dalzell, B. J., Donahue, M., Gourevitch, J., Johnson, D. L., Karlovits, G. S., et al. (2016). Conservation Reserve Program (CRP) lands provide ecosystem service benefits that exceed land rental payment costs. *Ecosyst. Serv.* 18, 175–185. doi: 10.1016/j.ecoser.2016.03.004

Johnson, R., and Monke, J. (2019). *What Is the Farm Bill?* CRS Report RS22131. Washington, DC: Congressional Research Service.

Key, N. (2019). Farm size and productivity growth in the United States corn belt. *Food Policy* 84, 186–195. doi: 10.1016/j.foodpol.2018.03.017

Khoury, C. K., Bjorkman, A. D, Dempewolf, H., Ramirez-Villegas, J., Guarino, L., Jarvis, A., et al. (2014). Increasing homogeneity in global food supplies and the implications for food security. *Proc. Natl. Acad. Sci. U.S.A.*111, 4001–4006. doi: 10.1073/pnas.1313490111

Krippendorf, K. (2004). *Content Analysis: An Introduction to Its Methodology, 2nd Edn*. Thousand Oaks, CA: SAGE Publications, Inc. Available online at: https://www.jstor.org/stable/2288384?origin=crossref

Laingen, C. R. (2017). Creating a dynamic regional model of the US corn belt. *Int. J. Appl. Geospat. Res.* 8, 19–29. doi: 10.4018/IJAGR.2017100102

Landis, D. A. (2017). Designing agricultural landscapes for biodiversity-based ecosystem services. *Basic Appl. Ecol.* 18, 1–12. doi: 10.1016/j.baae.2016.07.005

Lehner, P. H., and Rosenberg, N. A. (2018). Farm bill to help farmers weather climate change. *J. Food Law Policy* 14, 39–54. Available online at: https://ssrn.com/abstract=3219464

Luttrell, C. B. (1973). *The Russian Wheat Deal-Hindsight vs. Foresight*. Federal Reserve Bank of St. Louis 55. doi: 10.20955/r.55.2-9.jiy

MacDonald, J. M., and Hoppe, R. A. (2017). *Large Family Farms Continue to Dominate, US Agricultural Production. Statistic: Farm Economy, Amber Waves*. Washington, DC: Economic Research Service.

Martin, A. R., Cadotte, M. W., Isaac, M. E., Milla, R., Vile, D., and Violle, C. (2019). Regional and global shifts in crop diversity through the anthropocene. *PLoS ONE* 14:e0209788. doi: 10.1371/journal.pone.0209788

Massawe, F., Mayes, S., and Cheng, A. (2016). Crop diversity: an unexploited treasure trove for food security. *Trends Plant Sci.* 21, 365–368. doi: 10.1016/j.tplants.2016.02.006

McDaniel, M. D., Tiemann, L. K., and Grandy, A. S. (2014). Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics? A meta-analysis. *Ecol. Appl.* 24, 560–570. doi: 10.1890/13-0616.1

McFadden, J. R., and Hoppe, R. A. (2017). The *Evolving Distribution Of Payments From Commodity, Conservation, And Federal Crop Insurance Programs*. Economic Information Bulletin Number 184. USDA Economic Research Service.

McGranahan, D. A., Brown, P. W., Schulte, L. A., and Tyndall, J. C. (2013). A historical primer on the US farm bill: supply management and conservation policy. *J. Soil Water Conserv.* 68, 67A−73A. doi: 10.2489/jswc.68.3.67A

McMinimy, M. A., Angadjivand, S., Aussenberg, R. A., Billings, K. C., Bracmort, K., Casey, A. R., et al. (2019). *The 2018 Farm Bill (P.L. 115-334): Summary and Side-by-Side Comparison*. Washington, DC: Congressional Research Service.

Meehan, T. D., Werling, B. P., Landis, D. A., and Gratton, C. (2011). Agricultural landscape simplification and insecticide use in the Midwestern United States. *Proc. Natl. Acad. Sci. U.S.A.* 108, 11500–11505. doi: 10.1073/pnas.1100751108

Minkoff-Zern, L. A., and Sloat, S. (2017). A new era of civil rights? latino immigrant farmers and exclusion at the United States department of agriculture. *Agric. Hum.Values* 34, 631–643. doi: 10.1007/s10460-016-9756-6

Mulvaney, R. L., Khan, S. A., and Ellsworth, T. R. (2009). Synthetic nitrogen fertilizers deplete soil nitrogen: a global dilemma for sustainable cereal production. *J. Environ. Qual.* 38, 2295–2314. doi: 10.2134/jeq2008.0527

National Agricultural Law Center (2019). *United States Farm Bills*. National Agricultural Law Center. Available online at: https://nationalaglawcenter.org/farmbills/

Newton, J. (2017). *Modernizing Base Acres*. Market Intel. Farm Bureau. Available online at: https://www.fb.org/market-intel/modernizing-base-acres

NRC (2010). *Toward Sustainable Agricultural Systems in the 21st Century*. National Research Council of the National Academies. Washington, DC: The National Academies Press.

O'Connor, A. (2012). Fence row to fence row: a examination of federal commodity subsidies. *Kan. J. Law Pub. Policy*. 21, 432–450.

O'Donoghue, E. J., Roberts, M. J., and Key, N. (2009). Did the federal crop insurance reform act alter farm enterprise diversification? *J. Agric. Econ.* 60, 80–104. doi: 10.1111/j.1477-9552.2008.00166.x

Olson, A. (2001). Federal farm programs - past, present and future-will we learn from our mistakes? *Great Plains Nat. Resour.* J. 1:6.

Ortiz-Bobea, A., Knippenberg, E., and Chambers, R. G. (2018). Growing climatic sensitivity of US agriculture linked to technological change and regional specialization. *Sci. Adv.* 4:eaat4343. doi: 10.1126/sciadv.aat4343

Paul, C. M., Nehring, R., Banker, D., and Somwaru, A. (2004). Scale economies and efficiency in US agriculture: are traditional farms history? *J. Product. Anal.* 22:22. doi: 10.1007/s11123-004-7573-1

Pellegrini, P., and Fernández, R. J. (2018). Crop intensification, land use, and on-farm energy-use efficiency during the worldwide spread of the green revolution. *Proc. Natl. Acad. Sci. U.S.A.* 115, 2335–2340. doi: 10.1073/pnas.1717072115

Phalan, B., Green, R., and Balmford, A. (2014). Closing yield gaps: perils and possibilities for biodiversity conservation. *Philos. Trans. R. Soc. B Biol. Sci.* 369:20120285. doi: 10.1098/rstb.2012.0285

Pimentel, D., and Burgess, M. (2014). "Chapter 2: environmental and economic costs of the application of pesticides primarily in the United States," in *Integrated Pest Management,* eds D. Pimentel and R. Peshin (Dordecht: Springer), 47–71.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., et al. (1995). Environmental and economic costs of soil erosion and conservation benefits. *Science* 267, 1117–1123. doi: 10.1126/science.267.5201.1117

Pridham, J. C., and Entz, M. (2008). Intercropping spring wheat with cereal grains, legumes, and oilseeds fails to improve productivity under organic management. *Agron. J.* 100:1436. doi: 10.2134/agronj2007.0227

Pywell, R. F., Heard, M. S., Woodcock, B. A., Hinsley, S., Ridding, L., Nowakowski, M., et al. (2015). Wildlife-friendly farming increases crop yield: evidence for ecological intensification. *Proc. R. Soc. B Biol. Sci.* 282:20151740. doi: 10.1098/rspb.2015.1740

R Core Team (2020). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing. Available online at: https://www.R-project.org/

Rabalais, N.N., Turner, R.E., and Wiseman, W.J. Jr. (2002). Gulf of Mexico Hypoxia, A.K.A "The Dead Zone." *Ann. Rev. Ecol. Syst.* 33, 235–363. doi: 10.1146/annurev.ecolsys.33.010802.150513

Rada, N. E., and Fuglie, K. O. (2019). New perspectives on farm size and productivity. *Food Policy* 84, 147–152. doi: 10.1016/j.foodpol.2018.03.015

Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M., et al. (2018). Trends in global agricultural land use: implications for environmental health and food security. *Ann. Rev. Plant Biol.* 69, 789–815. doi: 10.1146/annurev-arplant-042817-040256

Ray, D. K., Ramankutty, N., Mueller, N. D., West, P. C., and Foley, J. A. (2012). Recent patterns of crop yield growth and stagnation. *Nat. Commun.* 3:1293. doi: 10.1038/ncomms2296

Reganold, J. P., Jackson-Smith, D., Batie, S. S., Harwood, R. R., Kornegay, J. L., Bucks, D., et al. (2011). Transforming, US agriculture. *Science* 332, 670–671. doi: 10.1126/science.1202462

Reitsma, K. D., Dunn, B. H., Mishra, U., Clay, S. A., DeSutter, T., and Clay, D. E. (2015). Land-use change impact on soil sustainability in a climate and vegetation transition zone. *Agron. J.* 107, 2363–2372. doi: 10.2134/agronj15.0152

Rindfuss, R. R., Walsh, S. J., Turner, B. L., Fox, J., and Mishra, V. (2004). Developing a science of land change: challenges and methodological issues.

Proc. Natl. Acad. Sci. U.S.A. 101, 13976–13981. doi: 10.1073/pnas.0401545101

Robinson, R. A., and Sutherland, W. J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *J. Appl. Ecol*. 39, 157–176. doi: 10.1046/j.1365-2664.2002.00695.x

Rosenberg, N., and Stucki, B. W. (2017). The butz stops here: why the food movement needs to rethink agricultural history. *J. Food. Law. Policy*. 13, 12–25.

Sayler, K. L., Acevedo, W., and Taylor, J. L. (2016). *Status and Trends of Land Change in the Eastern United States-1973 to 2000*. U.S. Geological survey professional paper 1794-D. Land Cover Trends. Washington, DC: USGS.

Sleeter, B. M., Sohl, T. L., Loveland, T. R., Auch, R. F., Acevedo, W., Drummond, M. A., et al. (2013). Land-cover change in the conterminous United States from 1973 to 2000. *Glob. Environ. Change* 23, 733–748. doi: 10.1016/j.gloenvcha.2013.03.006

Sohl, T., Reker, R., Bouchard, M. A., Sayler, K., Dornbierer, J., Wika, S., et al. (2016). Modeled historical land use and land cover for the conterminous United States. *J. Land Use Sci*. 11, 476–499. doi: 10.1080/1747423X.2016.1147619

Taylor, J. L., Acevedo, W., Auch, R. F., and Drummond, M. A. (2015). *Status and Trends of Land Change in the Great Plains of the United States-1973 to (2000)*. U.S. Geological Survey Professional Paper 1794-B. Land Cover Trends. Washington, DC: USGS.

Tilman, D. (1999). Global environmental impacts of agricultural expansion: the need for sustainable and efficient practices. *Proc. Natl. Acad. Sci. U.S.A*. 96, 5995–6000. doi: 10.1073/pnas.96.11.5995

Tscharntke, T., Clough, Y., Wanger, T. C., Jackson, L., Motzke, I., Perfecto, I., et al. (2012). Global food security, biodiversity conservation and the future of agricultural intensification. *Biol. Conserv*. 151, 53–59. doi: 10.1016/j.biocon.2012.01.068

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., and Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecol. Lett*. 8, 857–874. doi: 10.1111/j.1461-0248.2005.00782.x

USDA (1982). *1982 Census Publications*. USDA Census of Agriculture Historical Archive. Avaliable online at: http://usda.mannlib.cornell.edu/usda/AgCensusImages/1982/01/51/121/Table-01.pdf

USDA (1992). *1992 Census Publications*. USDA Census of Agriculture Historical Archive. Avaliable online at: http://usda.mannlib.cornell.edu/usda/AgCensusImages/1992/01/51/1575/Table-06.pdf

USDA NASS (2019a). *2017 Census of Agriculture Report: United States Summary and State Data. AC-17-A-51. Geography Area Series, Part 51*. Washington, DC: United States Department of Agriculture.

USDA NASS (2019b). *Farms and Land in Farms: 2018 Summary*. USDA.

USDA NASS (2019c). *USDA National Agricultural Statistics Service Quick Stats*. Available online at: https://quickstats.nass.usda.gov/

Vitalis, V. (2007). Agricultural subsidy reform and its implications for sustainable development: the New Zealand experience. *Environ. Sci*. 4, 21–40. doi: 10.1080/15693430601108086

Way, M. J., and van Emden, H. F. (2000). Integrated pest management in practice - pathways towards successful application. *Crop Prot*. 19, 81–103. doi: 10.1016/S0261-2194(99)00098-8

Weber, J. G., Key, N., and O'Donoghue, E. (2016). Does federal crop insurance make environmental externalities from agriculture worse? *J. Assoc. Environ. Resour. Econ*. 3, 707–742. doi: 10.1086/687549

Willis, B., and O'Brien, D. (2002). *Summary and Evolution of U.S. Farm Bill Commodity Titles*. Fayetteville, AK: National Ag Law Center.

Wright, C. K., and Wimberly, M. C. (2013). Recent land use change in the western corn belt threatens grasslands and wetlands. *Proc. Natl. Acad. Sci. U.S.A*. 110, 4134–4139. doi: 10.1073/pnas.1215404110

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., and Swinton, S. M. (2007). Ecosystem services and dis-services to agriculture. *Ecol. Econ*. 64, 253–260. doi: 10.1016/j.ecolecon.2007.02.024

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The reviewer RA and handling editor declared their shared affiliation at the time of the review.

PERSPECTIVE
published: 19 August 2020
doi: 10.3389/fsufs.2020.00114

Beef Production in the Southwestern United States: Strategies Toward Sustainability

Sheri Spiegal[1]*, Andres F. Cibils[2], Brandon T. Bestelmeyer[1], Jean L. Steiner[2], Richard E. Estell[1], David W. Archer[3], Brent W. Auvermann[4], Stephanie V. Bestelmeyer[5], Laura E. Boucheron[6], Huiping Cao[7], Andrew R. Cox[8], Daniel Devlin[9], Glenn C. Duff[2,10], Kristy K. Ehlers[11], Emile H. Elias[1,12], Craig A. Gifford[13], Alfredo L. Gonzalez[1], John P. Holland[14], Jenny S. Jennings[4], Ann M. Marshall[11], David I. McCracken[14], Matthew M. McIntosh[2], Rhonda Miller[15], Mark Musumba[2], Robert Paulin[16], Sara E. Place[17], Matthew Redd[18], C. Alan Rotz[19], Cindy Tolle[20] and Anthony Waterhouse[14]

[1] USDA-ARS Jornada Experimental Range, Las Cruces, NM, United States, [2] Department of Animal and Range Sciences, New Mexico State University, Las Cruces, NM, United States, [3] USDA-ARS Northern Great Plains Research Laboratory, Mandan, ND, United States, [4] Texas A&M AgriLife Research, Amarillo, TX, United States, [5] Asombro Institute for Science Education, Las Cruces, NM, United States, [6] Klipsch School of Electrical and Computer Engineering, New Mexico State University, Las Cruces, NM, United States, [7] Department of Computer Science, New Mexico State University, Las Cruces, NM, United States, [8] Chihuahuan Desert Rangeland Research Center, New Mexico State University, Las Cruces, NM, United States, [9] Kansas Center for Agricultural Resources and the Environment & Kansas Water Resources Institute, Kansas State University, Manhattan, KS, United States, [10] Clayton Livestock Research Center, New Mexico State University, Las Cruces, NM, United States, [11] BlueSTEM AgriLearning Center, El Reno, OK, United States, [12] USDA Southwest Climate Hub, Las Cruces, NM, United States, [13] Extension Animal Sciences and Natural Resources, New Mexico State University, Las Cruces, NM, United States, [14] Hill & Mountain Research Centre, Scotland's Rural College, Perthshire, United Kingdom, [15] Department of Animal Science, Texas A&M, College Station, TX, United States, [16] Corta Madera Ranch, Pine Valley, CA, United States, [17] ELANCO, Greenfield, IN, United States, [18] The Nature Conservancy, Dugout Ranch and Canyonlands Research Center, Monticello, UT, United States, [19] USDA-ARS Pasture Systems and Watershed Management Research Unit, State College, PA, United States, [20] Evergreen Ranching & Livestock, Custer, SD, United States

Edited by:
Mark A. Drummond,
United States Geological Survey (USGS), United States

Reviewed by:
Christopher Laingen,
Eastern Illinois University,
United States
Jennifer Tucker,
The University of Georgia,
United States

***Correspondence:**
Sheri Spiegal
sheri.spiegal@usda.gov

Specialty section:
This article was submitted to Land, Livelihoods and Food Security, a section of the journal Frontiers in Sustainable Food Systems

Received: 30 August 2019
Accepted: 25 June 2020
Published: 19 August 2020

Citation:
Spiegal S, Cibils AF, Bestelmeyer BT, Steiner JL, Estell RE, Archer DW, Auvermann BW, Bestelmeyer SV, Boucheron LE, Cao H, Cox AR, Devlin D, Duff GC, Ehlers KK, Elias EH, Gifford CA, Gonzalez AL, Holland JP, Jennings JS, Marshall AM, McCracken DI, McIntosh MM, Miller R, Musumba M, Paulin R, Place SE, Redd M, Rotz CA, Tolle C and Waterhouse A (2020) Beef Production in the Southwestern United States: Strategies Toward Sustainability. Front. Sustain. Food Syst. 4:114. doi: 10.3389/fsufs.2020.00114

From grazing lands to meat packing, beef production systems in the United States are striving to meet global demands without compromising environmental quality or local profitability. These challenges and opportunities are manifest in four US regions connected ecologically and socially through beef production: the American Southwest, the Ogallala Aquifer region, the Northern Plains, and the Upper Midwestern Corn Belt. Most calves raised on extensive, arid Southwestern ranches are exported to the Ogallala Aquifer region for finishing on grains that are grown either locally on Ogallala Aquifer water or imported from the Upper Midwest. Changes in climate, vegetation, and human demographics threaten the sustainability of the regionally-interconnected system. Heritage cattle genetics, precision ranching, and alternative supply chain options are three strategies that show promise for addressing these sustainability threats, but major knowledge gaps exist. For instance, while environmentally-friendly landscape use by Raramuri Criollo, a heritage cattle type, has been identified in several arid rangeland settings, little is known about their performance in conventional feed yards. While precision agriculture is already prevalent in croplands, less is known about how such technologies can be cost effective in arid rangelands. Moreover, many perceive grass-finishing on rangeland as environmentally friendly and beneficial for local agricultural communities, but tradeoffs involving greenhouse gas emissions, increased rangeland use, and disruption of cattle feeding systems of the Ogallala Aquifer region must be assessed. Here we introduce a USDA-NIFA Coordinated Agricultural Project designed to fill these knowledge gaps and advance sustainability of beef production linked to the US Southwest. With a boundary-spanning approach of education, participatory research, and extension, the project is identifying tradeoffs of the three strategies with explicit attention to pericoupling (i.e., socioeconomic and environmental interactions) of regions connected by beef production and full consideration of the coupled ecological and social systems within those regions.

Keywords: Southwestern United States, rangelands, sustainable agricultural systems, Coordinated Agricultural Project, pericoupling framework

INTRODUCTION

Humans have used livestock grazing to adapt to arid landscapes for millennia (Clutton-Brock, 1989), but as livestock production has become embedded in a complex transnational meat supply chain, new strategies are needed to ensure sustainable production into the future. In the United States, about 25,000 cattle ranches are located in the arid and semi-arid Southwest[1]. These ranches produce ∼6% of the cows that provide calves for the US beef industry, making Southwestern ranching essential not only to local communities, economies, and landscapes, but to the nation's overall beef supply, as well (Havstad et al., 2018; USDA-NASS, 2020). However, the fragility of the predominant supply chain emanating from the Southwest coupled with increasing heat and drought are threatening the capacity of Southwestern ranchers to produce beef sustainably (Gershunov et al., 2013; Polley et al., 2013; Havstad et al., 2018; McIntosh et al., 2019; Hendrickson, 2020).

Most calves weaned on the cow-calf ranches of the Southwest are exported to the Ogallala Aquifer region[2] for backgrounding, grain finishing, and meat sales (Johnson and Becker, 2009; Buhnerkempe et al., 2013; Blank et al., 2016). The Ogallala Aquifer region also imports grain from the Upper Midwest[3] to meet feeding quotas not filled by local feed production (Gottschalk, 2007; Guerrero et al., 2013). Problems in one link of this inter-regional supply chain can compromise resilience of the entire chain. Moreover, interventions designed to solve problems in one region affect, and are affected by, ecological and socioeconomic dynamics in connected regions. Therefore, to foster beef production that is truly sustainable – that is, that satisfies dietary demand, protects environmental quality, and ensures economic security and good quality of life for producers and society (National Research Council, 2010; Kleinman et al., 2018) – we must understand the performance of beef production in multiple realms and in the multiple regions connected by supply and demand (Liu, 2017).

With these goals in mind, three strategies show promise for improving sustainability of beef production originating in the US Southwest and the regions connected to it: heritage cattle genetics, precision ranching, and alternative supply chain options. Here we summarize the major challenges to the sustainability of Southwest beef production, provide rationale for evaluating these three strategies as ways to address the challenges, and report early results of our multi-disciplinary, multi-year approach to understanding the benefits and drawbacks associated with each strategy (**Figure 1**). Our approach was funded in 2019 as a 5-year Coordinated Agricultural Project (CAP) by the United States Department of Agriculture – National Institute of Food and Agriculture (NIFA-AFRI #2019-69012-29853, www.swbeef.org). Here we report results of the first year of the "Sustainable Southwest Beef CAP."

[1]We define the US Southwest as the states of New Mexico, Arizona, Nevada, Utah, California (**Figure 1**).

[2]We define the Ogallala Aquifer region as parts of Texas, New Mexico, Oklahoma, Kansas, Nebraska, Colorado, Wyoming, and South Dakota (**Figure 1**).

[3]We define the Upper Midwest as the "Corn Belt" which covers Indiana, Illinois, Iowa, Missouri, eastern Nebraska, and eastern Kansas.

SUSTAINABILITY CHALLENGES FOR SOUTHWEST BEEF PRODUCTION

Sustainability Challenges on Pasture and Ranch Scales

Similar to other arid landscapes worldwide, range pastures of the American Southwest tend to be large and heterogeneous. Frequent use of particular locations by cattle can result in perennial grass loss (Bestelmeyer et al., 2018), soil degradation (Nash et al., 2003), and increased dust emissions (Baddock et al., 2011) – all of which diminish cattle weight gains (Holechek, 1992). Manipulating fencing, water locations, and timing of use are common approaches to improving livestock distribution in rangelands (Heitschmidt and Taylor, 1991; Owens et al., 1991). These interventions, however, can be cost-prohibitive to establish and maintain in arid systems (Hunt et al., 2007).

High input costs coupled with external market forces contribute to rates of return varying from *net losses* to only +3% on annual investment in the ranches of the American Southwest – significantly lower than the 6% received by US agriculture on the whole (Torell et al., 2001; USDA-ERS, 2016). Looking ahead, these economic stresses are projected to intensify as the Southwest continues to experience higher temperatures, increasing frequency and intensity of heat waves, and more frequent droughts (Gershunov et al., 2013; Briske et al., 2015; USGCRP, 2017). These novel climate exposures are predicted to affect ecosystems and economics through diminished rangeland carrying capacities, increased site vulnerability to soil degradation, compromised regional feed and pasture forage production, and intensified animal heat stress (Havstad et al., 2018).

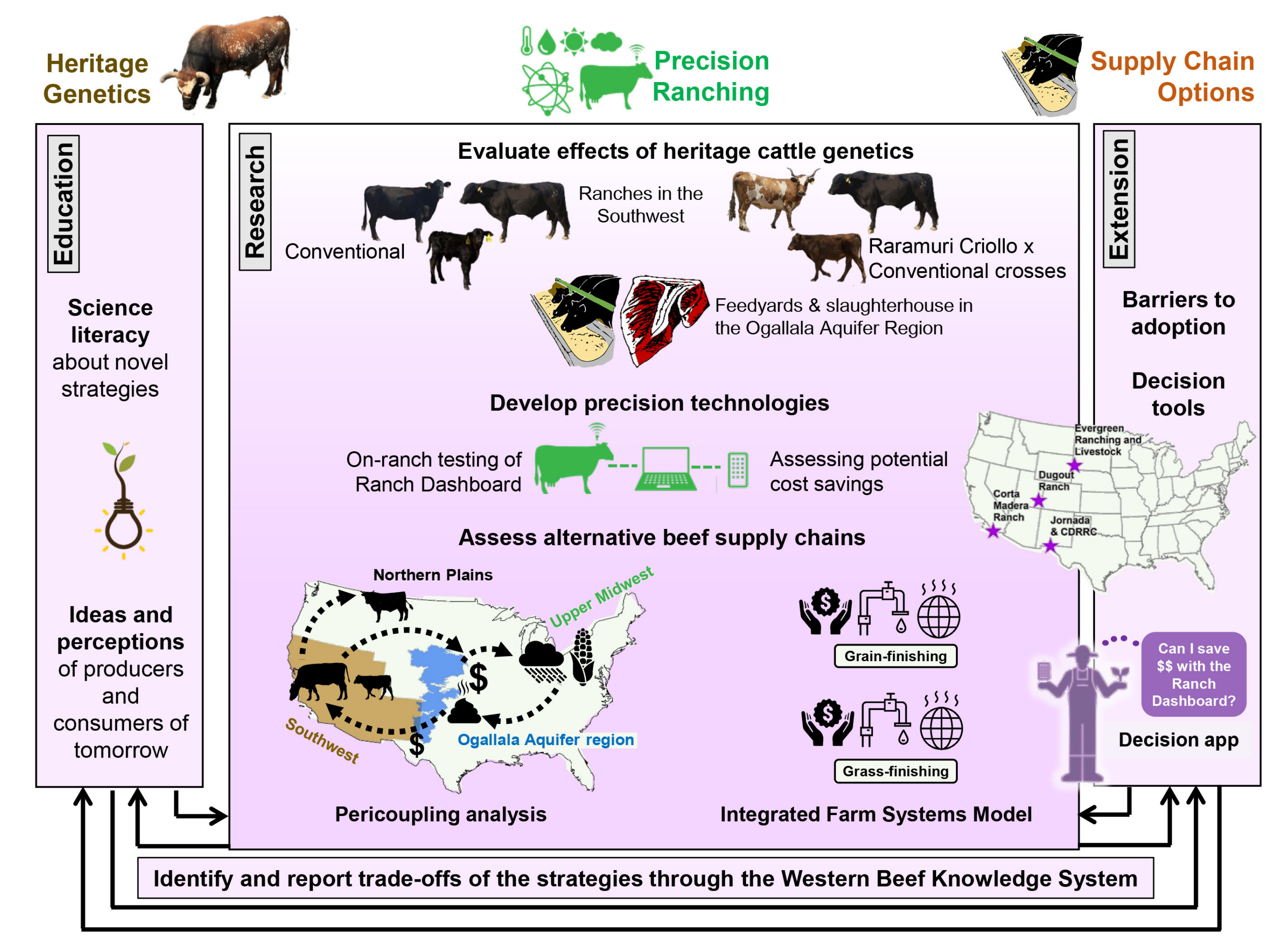


FIGURE 1 | The Sustainable Southwest Beef CAP is investigating three strategies with potential to improve sustainability of beef production originating from the American Southwest, using a boundary-spanning approach of education, participatory research, and extension.

Sustainability Challenges on the Supply Chain Scale

Looking beyond ranch gates, the specialization and concentration of US beef cattle and cattle feed production has greatly increased efficiency in terms of cost per unit of product of beef (Dimitri et al., 2005; Capper, 2011); however, it has also contributed to a host of environmental, economic, and societal concerns, including compromised environmental quality and quality of life for communities near concentrated feedlot manure (Casey et al., 2006), as well as vulnerabilities in supply chains. For instance, occupancy restrictions in meat processing plants experienced in the spring of 2020 due to COVID-19 have resulted in cattle remaining in feedyards longer, and fewer conventional cuts being available in supermarkets, affecting the ranches upstream (Peel et al., 2020; Texas A&M, 2020). The lack of typical beef cuts and volume in supermarkets resulted, for many Americans, in expanded interest in the provenance of beef and local beef products (Atkins, 2020; Emmert, 2020; Nagus, 2020). While it is too early to predict long-term effects at the writing of this article, it is possible that investment in alternative, local supply chains may ultimately affect the long-term economic sustainability of conventional grain finishing (Hobbs, 2020).

STRATEGIES TOWARD SUSTAINABILITY: NEW RESEARCH AND EARLY RESULTS

Heritage Cattle

The Raramuri Criollo biotype has undergone 500 years of adaptation to the harsh conditions of the Sierra Tarahumara in northern Mexico with minimal genetic influence of improved beef breeds (Estell et al., 2012; Anderson et al., 2015). Based on past research, Raramuri Criollo appear to experience less heat stress on hot summer days (Nyamuryekung'e et al., 2017) and have been anecdotally observed to forage more on low-quality grasses and shrubs than conventional beef breeds (Anderson et al., 2015). In addition, during seasons when green forage is relatively scarce and patchily distributed, Raramuri Criollo have been found to achieve greater distribution than conventional cattle types (Peinetti et al., 2011; Spiegal et al., 2019).

To date, grass finishing has been the primary option for Southwestern producers raising Raramuri Criollo, which can be finished on grass but get passed over at auctions in the conventional production chain due to color and shape non-conformity (Enyinnaya, 2016; Torell et al., in review). Another option is cross-breeding the heritage type with beef breeds used conventionally, thereby maintaining the potential economic and environmental benefits of Raramuri Criollo cows while producing more widely marketable offspring (Martínez-Cordova et al., 2014; Mcintosh et al., 2018).

While grass finishing and grain-finishing cattle with Raramuri Criollo genetics show promise for economic and environmental sustainability, especially under warmer and drier conditions, more information is needed before adoption of Raramuri Criollo genetics can be widely recommended. To fill these information gaps, a long-term breed comparison study was initiated in March 2020 on the New Mexico State University (NMSU) Chihuahuan Desert Rangeland Research Center (CDRRC) in Las Cruces, New Mexico. Four large pastures were dedicated to the respective cow-calf herds – two pastures for a heritage herd, and two for a conventional Brangus herd. External inputs and outputs are being quantified to assess differences in ranch total factor productivity (Ramankutty et al., 2018) between herds, and vegetation and soils are being monitoring to assess the ecological effects. To understand the processes driving production and ecological outcomes, cattle movements are being monitored in real time (see below), and costs and returns are being measured, including supplement intake, percent calf crop, and kilograms of calf weaned.

The feedlot and finishing performance will be compared between the heritage crossbred calves and conventional beef calves at research facilities in the Ogallala Aquifer region. Calves for this component of the study are being raised on cooperating ranches in southern New Mexico, southeastern California, and southeastern Utah (stars in **Figure 1**). The first calf crop is scheduled to be transported to Clayton, New Mexico for wheat pasture backgrounding and eventual finishing at Clayton, New Mexico, and Texas A&M Agrilife Research facilities in Bushland, Texas in fall 2020, and repeated the following 2 years. Slaughtered cattle will be subjected to beef quality tests including consumer taste panels at Texas A&M University in College Station, Texas in 2021, 2022, and 2023.

This breed comparison leverages one of 18 coordinated experiments in the Long-Term Agroecosystem Research (LTAR) network Common Experiment, contributing to a national assessment of the benefits and drawbacks of adopting "aspirational" management approaches on farms and ranches nationwide (Spiegal et al., 2018). This experiment is also part of an international network of long term grazing studies comparing the environmental footprint of Criollo vs. improved beef breeds at sites in Mexico and Argentina.

Precision Ranching

Sensor-driven precision farming, already mainstream in intensive animal agriculture systems (Neethirajan, 2017), can also help ranchers in the warming and drying American Southwest make rapid decisions to sustain animal health and forage resources. Real-time analysis of shifts in animal movement patterns associated with declining forage, inadequate or faulty water supply, birth, or predation helps ranchers to intervene rapidly, effectively providing a type of early warning system addressing multiple sustainability problems.

Importantly, these technologies can help reduce economic and environmental costs of ranching in extensive, arid lands. Based on calculations for the 780-km^2 USDA-ARS Jornada Experimental Range, wireless sensors indicating water levels in troughs could save 388–478 h of driving time and 742–956 gallons of fuel, which translates into $7,800–$10,000 in annual cost savings, 6.6–8.5 metric tons of avoided CO_2 emissions, and more time for pursuing other endeavors. On the other hand, investments in the system such as installation, maintenance, and time spent learning to use the technology can reduce overall cost-effectiveness of adoption.

To investigate the potential of these technologies in extensive arid landscapes, we are developing a precision ranching system able to log, transmit, and analyze animal, weather, and water sensor data in real time via a long-range, low power wireless area network (LoRa WAN), to be tested at five participating ranches (**Figure 1**). Cost inputs and savings from this technology will be assessed via enterprise budgets (Torell et al., 2014), and a survey instrument will be used to determine user perceptions regarding the usefulness of all aspects of system implementation. With this understanding of cost savings and feedback from participating ranchers, a market-ready product should be available within 6 years.

During the first year of the project, we built a pilot model and are testing it at the NMSU CDRRC, where the long-term breed comparison study was initiated. The GPS collars, watering tank, and rainfall sensors have been collecting data since March 2020 (**Supplement 1**). Initial testing and calibration of components of the precision ranching system at CDRRC is allowing us to gauge its usefulness and is helping our team identify and carefully document potential challenges of using LoRa WAN on extensive cattle ranches with sparse communication networks. Understanding these technological hurdles will be critically important as we roll out the precision ranching system on cooperating commercial ranches in the near future.

Supply Chain Options

Amid concerns about food safety and environmental impacts of beef supply chains, the market share for alternative beef products – natural, certified organic, grass-fed – has been growing in recent decades (Tonsor et al., 2009; Mathews and Johnson, 2013; Food Marketing Institute, 2017), and societal interest in locally-sourced food appears to be growing rapidly during the ongoing COVID-19 pandemic. Thus, ranchers who grass-finish Raramuri Criollo cattle are part of a larger community in the American Southwest that has adopted grass finishing for a variety of reasons (Barnes, 2011).

During the past year of engaging with Southwestern producers who grass finish cattle, we have come to identify two main approaches: (1) finishing locally on arid ranches, and (2) exporting weaned calves to the Northern Plains[4] (the

[4]We define the Northern Plains as North Dakota, South Dakota, Minnesota, Iowa, and Nebraska.

"Follow the Green" production system). Much is unknown about the rate of adoption of these approaches, their ecological and economic outcomes, or how those outcomes compare with those of grain finishing systems – especially as the Ogallala Aquifer region's backgrounding and feedlot industries face threats of aquifer depletion (McGuire, 2017) and the expanding impacts of the COVID-19 pandemic (Hendrickson, 2020). Therefore, we are working to create a knowledge base for producers, consumers, regional planners, and policy makers involved with Southwest beef production so they can compare grass finishing vs. grain finishing under various scenarios of change. Our primary analytical tools are the Integrated Farm System Model (IFSM; Rotz et al., 2019) and a multi-regional "pericoupling" analysis (Liu, 2017).

The IFSM uses production inputs in the farms and ranches of a given supply chain to estimate the environmental and economic outcomes of that supply chain (including energy use; carbon, phosphorus, and reactive nitrogen footprints; water consumption; production costs; and net returns). We are using IFSM to compare economic and environmental outcomes in six supply chains: Follow the Green with and without Raramuri Criollo, Grass-Finishing in the Southwest with and without Raramuri Criollo, and Grain-Finishing in the Ogallala Aquifer Region with and without Raramuri Criollo. We are gathering information on inputs from five ranchers and two feedyard operators formally participating in project research, as well as from other producers engaged through the CAP's extension efforts (see below). Ultimately the simulated environmental and economic effects will provide a measure of the long-term sustainability of the six supply chains, so that tradeoffs can be quantified and compared.

All six production systems being simulated in the IFSM originate with calves born on Southwestern ranches. The weaned calves are then exported to other regions (Follow the Green, Grain-Finishing), or are held back from those regions (Grass-Finishing in the Southwest). Given these inter-regional connections, we aim to understand how dynamics in one region affect the dynamics of the others, and vice versa. To that end, we are conducting a "pericoupling" analysis (Liu, 2017) to characterize the socioeconomic and environmental interactions among the regional systems linked via beef production under both the current system and a plausible near-future scenario (da Silva et al., 2019). Using the pericoupling framework, we are addressing the following questions about the connected regions under both the current and future scenarios (**Table 1**):

1. What are the flows of resources among four regions connected through beef production (the Southwest, Ogallala Aquifer region, Upper Midwest, and Northern Plains)?
2. What agents bring forth the connections (pericouplings) between the regions?
3. What are the causes of the pericouplings between the regions?
4. What are the major effects of the pericouplings on each region?

During the past year, we have built our pericoupling database with agro-economic datasets that span national, state, and regional levels, as well as results from IFSM simulations, and information from our integrated extension and education activities. Preliminary results are in **Table 1**.

TABLE 1 | Preliminary results of a pericoupling analysis to assess linkages of four regions affected by beef production in the American Southwest, under the current system and a plausible near-future scenario.

Scenario	Current	Future
Flows	Weaned calves: Almost all calves weaned on Southwestern ranches are exported to the Ogallala Aquifer region for finishing[a]. Feed grain: A large proportion of the grains used in finishing in the Ogallala Aquifer region are imported from the Upper Midwest. Cattle payments: Ogallala Aquifer region pays the Southwest via calf purchases. Manure nutrients: The Ogallala Aquifer region takes responsibility for managing manure nutrients of calves imported from the Southwest.	Weaned calves: Half of the calves weaned on Southwestern ranches are exported to the Ogallala Aquifer region for grain finishing, a quarter are exported to the Northern Plains for grass finishing, and a quarter are retained in the Southwest for grass finishing. Feed grain: Amount imported by the Ogallala Aquifer region from the Upper Midwest decreases. Cattle payments: Ogallala Aquifer region and Northern Plains pay the Southwest via calf purchases. Money received via calf purchases is retained in the Southwest. Manure nutrients: The Ogallala Aquifer region and Northern Plains are responsible for managing manure nutrients for imported calves; more manure nutrients are retained on Southwestern rangelands.
Agents	—Ranchers, brokers, vertically integrated feedyards/packers, major beef retailers, policy-makers, consumers, niche marketing cooperatives—	
Causes	Consumers' sustained demand for marbled beef. Location of major meat packers and vertical integration of animal production in US. Economies of scale for grain finishing.	Consumer concerns about grain-finishing supply chain; increased and sustained demand for alternatives. Continued social distancing as experienced in spring 2020. Reduced availability of Ogallala Aquifer water for backgrounding and feedlots. Input cost savings and/or government cost-sharing via heritage genetics and precision ranching.
Effects	"Brittle" food system. In Ogallala Aquifer region: Calves imported from the Southwest support employment in backgrounding and grain-finishing industries. Use of aquifer water for backgrounding and finishing calves from the Southwest. Declining water table levels. In Upper Midwest: The Ogallala Aquifer region is a market for grain.	In Ogallala Aquifer region: Reductions in: imported grains, imported beef cattle manure loads, employment in backgrounding and grain-finishing industries, aquifer water use. In the Southwest: Increases in: range use, hay demand, local revenue, opportunities for niche marketing. Longer methanogenic rumination. In Upper Midwest: Grain market disruption. In Northern Plains: Increased range use and possibly increased demand for feeder calves to utilize Ogallala Aquifer region packing quotas.

[a] *See* ***Supplement 2*** *for initial data analysis to estimate cattle flows from the US Southwest to the Ogallala Aquifer region.*

BOUNDARY-SPANNING APPROACH

We designed the Sustainable Southwest Beef CAP to span boundaries between science and decision-making in order to improve actions in both realms (*sensu* Bednarek et al., 2018). A central pillar is participatory research: All research is being conducted at least in part on commercial ranches, with direct involvement of ranch operators. This involvement, from study design to execution to data interpretation, is ensuring tight linkages between science and real-world challenges and opportunities in Southwestern beef production. The boundary-spanning approach was adopted, in part, to ensure a realistic understanding of opportunities for, and barriers to, adoption of the strategies under investigation.

To understand more about the potential for adoption of the strategies under investigation, during the past year, knowledge co-production/extension partners in the Sustainable Southwest Beef CAP – from New Mexico State University, the USDA Southwest Climate Hub, and Texas A&M AgriLife – engaged with producers from the Southwest and the regions pericoupled to the Southwest through beef production. Central tools have been on-ranch demonstrations, in-person events, podcasts, and surveys. For instance, the project team hosted an event for ∼125 ranchers, feedlot operators and others connected to the beef cattle industry at the 2020 Southwest Beef Symposium in Amarillo, Texas, where initial rancher perceptions of the three strategies were collected. Cattle producers ($n = 36$) from 26 counties across seven states completed the CAP's "baseline" survey (Elias et al., in review). In response to a question about which topic of the project would be most immediately applicable to their operation, about a quarter indicated that precision ranching technology is most applicable, another quarter selected range finishing in the Southwest and other supply chain options, and another quarter chose the overall integrated approach of the CAP as most applicable. Ten percent of respondents indicated that Spanish/heritage breed cattle would be most applicable. We will compare baseline data with surveys at the end of the 5-year project to detect changes in perceptions about the strategies.

In partnership with the knowledge co-production/extension and research teams, the Asombro Institute for Science Education in New Mexico and the BlueSTEM Agri-Learning Center in Oklahoma have developed lessons and teacher trainings to increase science literacy, advance knowledge about difficult decision-making technology in agriculture, and garner feedback about the strategies under investigation from the agricultural professionals of tomorrow. The integration of K-12 activities into the other components of the CAP emphasizes collaboration, interdisciplinary thinking, and strong communication skills (Bestelmeyer et al., 2015).

In the first year of the project, the education team developed a 1-h classroom lesson and a field trip activity to introduce lower elementary students to Raramuri Criollo (https://asombro.org/wp-content/uploads/Criollo.pdf). Lessons were based on the Sustainable Southwest Beef CAP project and aligned with the Next Generation Science Standards (NGSS), making them relevant to teachers in New Mexico and 19 other states using these standards. Lessons were developed and pilot tested with more than 200 2nd and 3rd grade students in the fall 2020 semester.

School closings in the spring 2020 semester halted classroom lessons, field trips, and teacher trainings. The education team therefore pivoted toward developing an interactive learning experience that could be done by students learning from home. "Solving the Beef" (https://asombro.org/solvingthebeef/) is a game that encourages players in competing teams to develop creative solutions for sustainable beef production and marketing given a set of scenarios and constraints. It is built around engineering design principles from NGSS. Though Solving the Beef was developed as an adaptation to social distancing, it can also be played in a traditional classroom or after-school setting. The game will be expanded by adding additional scenarios as results from the Southwest Beef CAP are published. Moreover, the game will allow the education team to collect ideas from students – the producers and consumers of tomorrow – to feed back to the research component of the project.

ASSESSING AND COMMUNICATING TRADEOFFS

In addition to peer-reviewed and popular press articles, an interactive repository is being built to house and communicate the integrated knowledge developed by the Sustainable Southwest Beef CAP. The "Western Beef Knowledge System" is being designed to aid decision-making around beef production and consumption, with geographically-specific information for producers about the potential benefits and drawbacks of adopting the strategies under investigation, and for consumers seeking locally-tailored guidance on how they can purchase beef that aligns with their stated values. We have also developed short factsheets for use by regional planners and other policy makers, as they evaluate incentives for adoption of the strategies and understand the inter-regional effects of alternative beef supply chains (https://southwestbeef.org/factsheets).

Ultimately, our goal is to apply new, integrated knowledge to advance sustainability of US beef production. Adoption of animal genetics suited to a hotter, drier climate, precision technologies that provide affordable and timely information for ranch management, and alternative marketing options all have potential to improve economic, environmental, and societal outcomes. However, when making significant changes in an agricultural system, full consideration of the regions pericoupled through production is necessary to achieve desired outcomes. With our boundary-spanning approach, we aim to illuminate these inter-regional connections, and identify viable pathways to improve sustainability for beef producers, beef consumers, and the rangelands cherished by Americans nationwide.

AUTHOR CONTRIBUTIONS

All authors contributed equally to the ideas in this article. SS was lead writer with contributions from, and editing by, AFC, BB, JS, RE, DA, CR, and MMu. MMc created artwork for **Figure 1**.

FUNDING

The Sustainable Southwest Beef Coordinated Agricultural Project was funded by the USDA National Institute of Food and Agriculture, Agriculture and Food Research Initiative's Sustainable Agricultural Systems (SAS) program. Grant #2019-69012-29853. This research was also a contribution from the Long-Term Agroecosystem Research (LTAR) network. LTAR is supported by the United States Department of Agriculture.

ACKNOWLEDGMENTS

Many thanks to Dr. Jack Liu for sharing his expertise for our pericoupling analysis. Thanks also to Dr. David Anderson of Texas A&M for providing data on interstate cattle movements into Texas and inspected by the Texas Animal Health Commission, as discussed in the **Supplementary Information**.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.00114/full#supplementary-material

REFERENCES

Anderson, D. M., Estell, R. E., Gonzalez, A. L., Cibils, A. F., and Torell, L. A. (2015). Criollo cattle: heritage genetics for arid landscapes. *Rangelands* 37, 62–67. doi: 10.1016/j.rala.2015.01.006

Atkins, R. (2020). *New Mexico Cattle Ranchers Busy Due to COVID-19. KRQE.* Available online at: https://www.krqe.com/health/coronavirus-new-mexico/new-mexico-cattle-ranchers-busy-due-to-covid-19/ (accessed July 22, 2020).

Baddock, M. C., Zobeck, T. M., Van Pelt, R. S., and Fredrickson, E. L. (2011). Dust emissions from undisturbed and disturbed, crusted playa surfaces: cattle trampling effects. *Aeolian Res.* 3, 31–41. doi: 10.1016/j.aeolia.2011.03.007

Barnes, M. K. (2011). Low-input grassfed livestock production in the American West: case studies of ecological, economic, and social resilience. *Rangelands* 33, 31–40. doi: 10.2111/1551-501X-33.2.31

Bednarek, A. T., Wyborn, C., Cvitanovic, C., Meyer, R., Colvin, R. M., Addison, P. F. E., et al. (2018). Boundary spanning at the science–policy interface: the practitioners' perspectives. *Sustainabil. Sci.* 13, 1175–1183. doi: 10.1007/s11625-018-0550-9

Bestelmeyer, B. T., Peters, D. P., Archer, S. R., Browning, D. M., Okin, G. S., Schooley, R. L., et al. (2018). The grassland–shrubland regime shift in the Southwestern United States: misconceptions and their implications for management. *BioScience* 68, 678–690. doi: 10.1093/biosci/biy065

Bestelmeyer, S. V., Elser, M. M., Spellman, K. V., Sparrow, E. B., Haan-Amato, S. S., and Keener, A. (2015). Collaboration, interdisciplinary thinking, and communication: new approaches to K−12 ecology education. *Front. Ecol. Environ.* 13, 37–43. doi: 10.1890/140130

Blank, S. C., Saitone, T. L., and Sexton, R. J. (2016). Calf and yearling prices in the western United States: Spatial, quality, and temporal factors in satellite video auctions. *J. Agric. Resource Econ.* 41, 458–480. doi: 10.22004/ag.econ.246175

Briske, D. D., Joyce, L. A., Polley, H. W., Brown, J. R., Wolter, K., Morgan, J. A., et al. (2015). Climate-change adaptation on rangelands: linking regional exposure with diverse adaptive capacity. *Front. Ecol. Environ.* 13, 249–256. doi: 10.1890/140266

Buhnerkempe, M. G., Grear, D. A., Portacci, K., Miller, R. S., Lombard, J. E., and Webb, C. T. (2013). A national-scale picture of US cattle movements obtained from interstate certificate of veterinary inspection data. *Prev. Vet. Med.* 112, 318–329. doi: 10.1016/j.prevetmed.2013.08.002

Capper, J. L. (2011). The environmental impact of beef production in the United States: 1977 compared with 2007. *J. Anim. Sci.* 89, 4249–4261. doi: 10.2527/jas.2010-3784

Casey, K. D., Bicudo, J. R., Schmidt, D. R., Singh, A., Gay, S. W., Gates, R. S., et al. (2006). "Air quality and emissions from livestock and poultry production/waste management systems," in *Animal Agriculture and the Environment National Center for Manure and Animal Waste Management White Papers*, eds. J. M. Rice, D. F. Caldwell, and F. J. Humenik (St. Joseph, MI: American Society of Agricultural and Biological Engineers), 1–40.

Clutton-Brock, J. (1989). "Introduction," in *The Walking Larder: Patterns of Domestication, Pastoralism, and Predation,* ed. J. Clutton-Brock (London: Unwin Hyman).

da Silva, R. F. B., Batistella, M., Palmieri, R., Dou, Y., and Millington, J. D. (2019). Eco-certification protocols as mechanisms to foster sustainable environmental practices in telecoupled systems. *For. Policy Econ.* 105, 52–63. doi: 10.1016/j.forpol.2019.05.016

Dimitri, C., Effland, A. B., and Conklin, N. C. (2005). *The 20th Century Transformation of US Agriculture and Farm Policy*. Washington, DC: US Department of Agriculture, Economic Research Service.

Emmert, M. (2020). *As COVID-19 Pandemic Rocks Meatpacking Industry*, *Iowans Turn to Local Farmers to Buy Meat*. Des Moines, IA: Des Moines Register.

Enyinnaya, J. C. (2016). *A Case Study of the Economics of Criollo Cattle Production in the Deserts of Southern New Mexico*. Las Cruces, NM: New Mexico State University.

Estell, R. E., Havstad, K. M., Cibils, A. F., Fredrickson, E. L., Anderson, D. M., Schrader, T. S., et al. (2012). Increasing shrub use by livestock in a world with less grass. *Rangel. Ecol. Manag.* 65, 553–562. doi: 10.2111/REM-D-11-00124.1

Food Marketing Institute (2017). *Power of Meat*: *An in-Depth Look at Meat Through the Shoppers' Eyes*. Washington, DC: North American Meat Institute.

Gershunov, A., Rajagopalan, B., Overpeck, J., Guirguis, K., Cayan, D., Hughes, M., et al. (2013). "Future climate: projected extremes," in *Assessment of Climate Change in the Southwest United States* (Washington, DC: Island Press; Springer), 126–147. doi: 10.5822/978-1-61091-484-0_7

Gottschalk, A. (2007). "*The impacts of the US corn/ethanol policy on the US cattle* industry," in *Proceedings, The Range Beef Cow Symposium XX,* (Fort Collins, Colorado).

Guerrero, B., Amosson, S., and McCollum, T. (2013). *The impact of the beef industry in the Southern Ogallala Region.* College Station, Texas: Texas A&M Agrilife Extension, AG-001.

Havstad, K. M., Brown, J. R., Estell, R., Elias, E., Rango, A., and Steele, C. (2018). Vulnerabilities of Southwestern US Rangeland-based animal agriculture to climate change. *Clim. Change* 148, 371–386. doi: 10.1007/s10584-016-1834-7

Heitschmidt, R. K., and Taylor, C. A. (1991). "Livestock production," in *Grazing: an Ecological Perspective*, eds. R. K. Heitschmidt and J. W. *Stuth (Portland: Timber Press)*, 161–177.

Hendrickson, M. K. (2020). Covid lays bare the brittleness of a concentrated and consolidated food system. *Agric. Hum. Values*. 12:1–12. doi: 10.1007/s10460-020-10092-y

Hobbs, J. E. (2020). Food supply chains during the COVID-19 pandemic. *Can. J. Agric. Econ. Revue Can. D'agroeconomie*. 1–6. doi: 10.1111/cjag.12237

Holechek, J. L. (1992). Financial benefits of range management practices in the Chihuahuan Desert. *Rangelands* 14, 279–284.

Hunt, L. P., Petty, S., Cowley, R., Fisher, A., Ash, A. J., and MacDonald, N. (2007). Factors affecting the management of cattle grazing distribution in northern Australia: preliminary observations on the effect of paddock size and water points. *Rangel. J.* 29, 169–179. doi: 10.1071/RJ07029

Johnson, R., and Becker, G. S. (2009). *Livestock Marketing and Competition Issues. CRS Report for Congress 2009.* Washington, DC: US Government Printing Office.

Kleinman, P. J. A., Spiegal, S., Rigby, J. R., Goslee, S., Baker, J., Bestelmeyer, B. T., et al. (2018). Advancing sustainable intensification of U.S. *agriculture through long-term research. J. Environ. Qual.* 47, 1412–1425. doi: 10.2134/jeq2018.05.0171

Liu, J. (2017). Integration across a metacoupled world. *Ecol. Soc.* 22:29. doi: 10.5751/ES-09830-220429

Martínez-Cordova, Y., García-Galicia, I. A., and Rodríguez-Almeida, F. A. (2014). "Características de la carne de ganado Criollo de Chihuahua, Angus × Criollo y Hereford × Angus finalizados en corral," in *XLI Reunión de la Asociación Mexicana para la Producción Animal y Seguridad Alimentaria A.C. (AMPA) y VII Reunión Nacional de Sistemas Agro y Silvopastoriles* (Mérida), 5.

Mathews, K. H., and Johnson, R. J. (2013). *Alternative Beef Production Systems: Issues and Implications*. USDA ERS. LDPM-218-01.

McGuire, V. L. (2017). *Water-Level and Recoverable Water in Storage Changes, High Plains Aquifer, Predevelopment to 2015 and 2013–15*. Scientific Investigations Report 2017-5040. US Geological Survey.

McIntosh, M. M., Cibils, A. F., Estell, R. E., Soto-Navarro, S. A., Gonzalez, A. L., Nyamuryekung,'E., et al. (2018). "Weight gain and behavior of Raramuri Criollo versus Corriente steers developed on Chihuahuan Desert rangeland," in *71st Annual Society for Range Management Meeting Abstracts* (Sparks, NV: Society for Range Management).

McIntosh, M. M., Holechek, J. L., Spiegal, S. A., Cibils, A. F., and Estell, R. E. (2019). Long-term declining trends in Chihuahuan Desert forage production in relation to precipitation and ambient temperature. *Rangel. Ecol. Manag.* 72, 976–87. doi: 10.1016/j.rama.2019.06.002

Nagus, C. (2020). As beef shortages take place due to COVID-19 outbreak, local rancher sees sales surge. *KMOV4*. Available online at: https://www.krqe.com/health/coronavirus-new-mexico/new-mexico-cattle-ranchers-busy-due-to-covid-19/ (accessed July 22, 2020).

Nash, M. S., Jackson, E., and Whitford, W. G. (2003). Soil microtopography on grazing gradients in Chihuahuan Desert grasslands. *J. Arid Environ.* 55, 181–192. doi: 10.1016/S0140-1963(02)00251-3

National Research Council (2010). *Toward Sustainable Agricultural Systems in the 21st Century*. Washington, DC: The National Academies Press.

Neethirajan, S. (2017). Recent advances in wearable sensors for animal health management. *Sens. Bio-Sens. Res.* 12, 15–29. doi: 10.1016/j.sbsr.2016.11.004

Nyamuryekung'e, S., Cibils, A. F., Estell, R. E., Gonzalez, A. L., Cano, D. E., and Spiegal, S. (2017). "Behavior of *Raramuri Criollo* vs. Angus-crossbred cows in relation to desert summer ambient heat conditions," in *70th Annual Society for Range Management Meeting Abstracts* (St. George, UT: Society for Range Management).

Owens, M. K., Launchbaugh, K. L., and Holloway, J. W. (1991). Pasture characteristics affecting spatial distribution of utilization by cattle in mixed brush communities. *J. Range Manag.* 44, 118–123. doi: 10.2307/4002308

Peel, D. S., Dustin, A., Randy, B., Kenneth, B., Don, C., Amy, H., et al. (2020). *Economic Damages to the U.S. Beef Cattle Industry Due to COVID-19. Oklahoma Cooperative Extension Service* Available online at: https://extension.okstate.edu/coronavirus/media/beef-economic-damages-full-report-covid-19.pdf (accessed May 26, 2020).

Peinetti, H. R., Fredrickson, E. L., Peters, D. P., Cibils, A. F., Roacho-Estrada, J. O., and Laliberte, A. S. (2011). Foraging behavior of heritage versus recently introduced herbivores on desert landscapes of the American Southwest. *Ecosphere* 2:art57. doi: 10.1890/ES11-00021.1

Polley, H. W., Briske, D. D., Morgan, J. A., Wolter, K., Bailey, D. W., and Brown, J. R. (2013). Climate change and North American rangelands: trends, projections, and implications. *Rangeland Ecol. Manag.* 66, 493–511. doi: 10.2111/REM-D-12-00068.1

Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M., et al. (2018). Trends in global agricultural land use: implications for environmental health and food security. *Annu. Rev. Plant Biol.* 69, 789–815. doi: 10.1146/annurev-arplant-042817-040256

Rotz, C. A., Asem-Hiablie, S., Place, S., and Thoma, G. (2019). Environmental footprints of beef cattle production in the United States. *Agric. Syst.* 169, 1–13. doi: 10.1016/j.agsy.2018.11.005

Spiegal, S., Bestelmeyer, B. T., Archer, D. W., Augustine, D. J., Boughton, E. H., Boughton, R. K., et al. (2018). Evaluating strategies for sustainable intensification of US agriculture through the Long-Term Agroecosystem Research Network. *Environ. Res. Lett.* 13:034031. doi: 10.1088/1748-9326/aaa779

Spiegal, S., Estell, R. E., Cibils, A. F., James, D. K., Peinetti, H. R., Browning, D. M., et al. (2019). Seasonal divergence of landscape use by heritage and conventional cattle on desert rangeland. *Rangeland Ecol. Manag.* 72, 590–601. doi: 10.1016/j.rama.2019.02.008

Texas A&M (2020). COVID-19 "The Texas A&M system responds - episode 7," in *The Age of the Coronavirus, Chancellor John Sharp Seeks to Answer the Question: "Where's the beef?" May 20, 2020 Interview of Dr. David Anderson*. College Station, TX: The Texas A&M University System.

Tonsor, G. T., Schroeder, T. C., Pennings, J. M. E., and Mintert, J. (2009). Consumer valuations of beef steak food safety enhancement in Canada, Japan, Mexico, and the United States. *Can. J. Agric. Econ. Revue Canadienne D'agroeconomie* 57, 395–416. doi: 10.1111/j.1744-7976.2009.01158.x

Torell, L. A., Rimbey, N. R., Tanaka, J. A., and Bailey, S. A. (2001). "The lack of a profit motive for ranching: implications for policy analysis," in *Proceedings of a Symposium Sponsored by the Western Coordinating Committee. Current Issues in Rangeland Resource Economics,* (Kailua-Kona, Hawaii, USA). Available online at: http://www.publiclandsranching.org/htmlres/pdf/ref_torell_lack_profit_motive.pdf (accessed November 16, 2013).

Torell, L. A., Rimbey, N. R., Tanaka, J. A., Taylor, D. T., and Wulfhorst, J. D. (2014). Ranch level economic impact analysis for public lands: a guide to methods, issues, and applications. *J. Rangeland Appl.* 1, 1–13.

USDA-ERS (2016). *Farm Income and Wealth Statistics*. Available online at: https://www.ers.usda.gov/data-products/farm-income-and-wealth-statistics/charts-and-maps-of-us-farm-balance-sheet-data.aspx (accessed July 22, 2020).

USDA-NASS (2020). *Beef Cow Inventory by State, Survey 2010-2020*. Available online at: https://quickstats.nass.usda.gov/results/C28763F7-8229-331B-93E9-AB82F81216E8 (accessed May 24, 2020).

USGCRP (2017). "Climate science special report: fourth national climate assessment," *Volume I*. eds. D. J. Wuebbles, D. W. Fahey, K. A. Hibbard, B. C. Dokken, B., Stewart, and T. K. Maycock (Washington, DC: Global Change Research Program).

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 08 September 2020
doi: 10.3389/fenvs.2020.00149

Check for updates

Demonstration and Testing of the Improved Shelterbelt Component in the Holos Model

Roland Kröbel[1]*, Julius Moore[1], Yu Zhao Ni[2], Aaron McPherson[1], Laura Poppy[2], Raju Y. Soolanayakanahally[2,3], Beyhan Y. Amichev[4], Tricia Ward[2], Colin P. Laroque[4], Ken C. J. Van Rees[4] and Fardausi Akhter[2]*

[1] Lethbridge Research and Development Centre, Agriculture and Agri-Food Canada, Lethbridge, AB, Canada, [2] Indian Head Research Farm, Agriculture and Agri-Food Canada, Indian Head, SK, Canada, [3] Saskatoon Research and Development Centre, Agriculture and Agri-Food Canada, Saskatoon, SK, Canada, [4] Department of Soil Science, University of Saskatchewan, Saskatoon, SK, Canada

The shelterbelt component of Canada's whole-farm model Holos was upgraded from an age-determined to a circumference-determined (at breast height) calculation using a multi-stem averaging approach. The model interface was developed around the idea that a shelterbelt could have multiple rows, and a variable species composition within each row. With this, the model calculates the accumulated aboveground carbon in the standing biomass and a lookup table of modeled tree growth is used to add estimates of the belowground carbon. Going from an initial interface that asks for the current state, the model also incorporates an option of past and future shelterbelt plantings. In order to test the model's suitability, we measured diverse shelterbelts (evergreen, deciduous, shrub type) in southern Saskatchewan, Canada representing commonly planted woody species. By making use of Caragana, Green Ash, Colorado Spruce, Siberian Elm, and a mixed Caragana/Green Ash tree row, we tested how many tree circumference measurements would be required to yield a representative average. Later, these results were incorporated in the Holos model to estimate the accumulated above- and below-ground carbon in each shelterbelt type.

Keywords: shelterbelts, agroforestry, Holos model, carbon sequestration, allometric modeling of carbon

OPEN ACCESS

Edited by:
Thomas Barfoed Randrup, Swedish University of Agricultural Sciences, Sweden

Reviewed by:
Michael W. Strohbach, Technische Universität Braunschweig, Germany
Edward W. Bork, University of Alberta, Canada

****Correspondence:***
Roland Kröbel
roland.kroebel@canada.ca
Fardausi Akhter
fardausi.akhter@canada.ca

Specialty section:
This article was submitted to Land Use Dynamics, a section of the journal Frontiers in Environmental Science

Received: *31 October 2019*
Accepted: *04 August 2020*
Published: *08 September 2020*

Citation:
Kröbel R, Moore J, Ni YZ, McPherson A, Poppy L, Soolanayakanahally RY, Amichev BY, Ward T, Laroque CP, Van Rees KCJ and Akhter F (2020) Demonstration and Testing of the Improved Shelterbelt Component in the Holos Model. Front. Environ. Sci. 8:149. doi: 10.3389/fenvs.2020.00149

INTRODUCTION

Global food consumption causes roughly one third of the global human induced greenhouse gas emissions (GHG), with agriculture directly contributing 23% (Intergovernmental Panel on Climate Change [IPCC], 2019), with the latter splitting approximately equally into CO_2 (from deforestation and other land use change), CH_4 (peatlands, rice cropping, and ruminant livestock), and N_2O (from crop production). With growing food demand and a still increasing global population at about ∼1% per year in 2015–2020 (United Nations [UN], 2019), these contributions are expected to continually increase (European Environment Agency, 2015). In Canada, the national inventory report (following IPCC guidelines) estimates that agriculture contributes 8.4% to the national GHG budget, with N_2O from cropping contributing 5.3% of the total national emissions and CH_4 contributing 13% of the total, since the fugitive emission from oil and gas, as well as from landfills, also contribute to the latter share (Baah-Acheamfour et al., 2017; Environment and Climate Change Canada [ECCC], 2019).

Due to the commitments in the Paris climate accord (United Nations Framework Convention on Climate Change [UNFCCC], 2019), Canada is committed to reducing its emissions from 730 Mt CO_2 eq. in 2005 to 511 Mt CO_2 eq. by 2030 (a reduction of 304 Mt CO_2 eq. from an unmitigated emissions scenario of 815 Mt CO_2 eq.), with an estimated contribution from agriculture of −2 Mt CO_2 eq. (Environment and Climate Change Canada [ECCC], 2020). Yet, the apparent temperature driven decomposition of soil carbon (Gregorich et al., 2017) calls into question the ability of Canadian soils to store more carbon in the future.

Regardless, planting and growing trees is touted as one of the most viable options to capture CO_2 from the atmosphere (Bastin et al., 2019), as they pose a longer-term storage of atmospheric carbon with the potential for further processing and carbon sequestration. Canada's agriculture landscape stretched over 158.7 million acres in 2016 (∼64.2 million ha) (StatsCan, 2019), and 1.7 million acres have planted shelterbelts (Toensmeier, 2016). Udawatta and Jose (2012) reported that shelterbelts could sequester up to 105 Mg C/ha in the aboveground shelterbelt biomass. When belowground biomass and soil carbon are added, shelterbelts have the potential to sequester a significant amount of atmospheric carbon per unit of land compared to other agricultural practices (despite some potential initial losses in soil carbon). In fact, the Canadian federal government once invested heavily into the planting of field, livestock, and farmyard shelterbelts which were intended to reduce wind speed and wind-derived soil erosion (Howe, 1986; Kulshreshtha et al., 2011) and enhance microclimate for crops (Kort, 1988; Kort et al., 2012) and animal production (Prairie Farm Rehabilitation Administration [PFRA], 1980; Poppy, 2003). Most of these shelterbelts have reached the end of their lifetime (Waldron and Hildahl, 1974; Rural Development Institute Shelterbelt Survey, 2014) and are removed for the sake of mechanization and production maximization (Waldron and Hildahl, 1974; Schroeder et al., 2011; Rempel, 2013; Rural Development Institute Shelterbelt Survey, 2014; Ha et al., 2019).

While many scientists have pointed out the various advantages of shelterbelts (Agriculture and Agri-Food Canada [AAFC], 2018), the question remains how Canadian landowners can be encouraged to maintain existing shelterbelts and increase their numbers beyond the current state (Rempel et al., 2017). A common pathway is regulatory or incentivizing policies, but information provision and education are important possibilities as well (Agriculture and Agri-Food Canada [AAFC], Agri-Environment Service Branch [AESB], and Agroforestry Development Centre [ADC] (2010). Stange and Jackson, 2015; Ward and de Gooijer, 2017; Stevenson, 2018). Simulation models are frequently used to test the accuracy and applicability of our gained scientific understanding of natural processes, and yet, such models, when packaged in appropriate software solutions, can be used to educate learners and practitioners in- and outside of academia. This is especially pertinent as the impact of our land use is increasingly felt in the environment (see for instance DiBartolomeis et al., 2019) and the importance of learning the complexities of land use/environment interactions for better decision-making wanes in comparison to perceived economic forcing. However, to gain an understanding of interactions, trade-offs, and the ripple effect of the various greenhouse gas sources and sinks (where the management of one can alter the other), interdisciplinary collaboration is required, to cover each and every aspect of farming systems and their potential interactions with the environment. Subsequently, a systems analysis approach has to be applied to ensure that identified impacts and benefits are not offset through some other, not considered, process or farming system component. For Canada and its farming systems, an initial step has been accomplished with the (whole-farm) Holos model (Little et al., 2008).

The Holos model, Agriculture and Agri-Food Canada's whole-farm model, is based on the conceptualization of a virtual farm approach proposed by Janzen et al. (2006), and was published in its first iteration as Holos Classic (Little et al., 2008). That first iteration aimed to educate Canadian farmers about the magnitude of GHG emission sources on their farm, and about potential mitigation options they could employ, one of which was the planting of shelterbelts as a way to use sequestered carbon (in tree and shrub biomass and select surface and soil C pools) as an offset against the emissions of other GHG. User feedback triggered the development of a subsequent research version of the model (Holos version 2 and 3), which added management flexibility to the interface, making the model's results more locally specific.

The Holos model has been utilized in several exemplary whole-farm analyses, ranging from understanding the general GHG impact of representative beef and dairy farm systems (Beauchemin et al., 2010; McGeough et al., 2012), while others tested real farm data (Church et al., 2015; Alemu et al., 2017a), or evaluated the effect of management practices on the whole-farm emissions (Alemu et al., 2017b; Guyader et al., 2017; Little et al., 2017). The model was utilized to investigate it's capability for calculating tree biomass in farming systems (Amadi et al., 2016; Mayrinck et al., 2019), and was also adopted to assess farming systems in Norway (Bonesmo et al., 2012, 2013; Skjelvåg et al., 2012; Gülzari et al., 2017, 2018; Samsonstuen et al., 2019) and Bulgaria (Petkova, 2012).

A renewed effort in model development followed (Kröbel et al., 2012), and the addition of a new carbon modeling approach (Kröbel et al., 2016) required a ground-up rebuild of the model and thus offered the opportunity to update algorithms and processes in the model, and to redesign the interface in a (non-scientific) user-friendly fashion. As part of this renewal, we started updating the old shelterbelt calculations methods which used allometric calculations based on age (Kort and Turnock, 1999) with the findings of Amichev et al. (2017), who redesigned the allometric calculations based on tree diameter/circumference. These are being incorporated in the new Holos model version 4, together with a stakeholder driven interface design. This paper reports on the practical testing of the interface, parameter requirements, and the underlying equations, which is being conducted using measurements from actual shelterbelts in the vicinity of Indian Head, SK, Canada.

The goal of this manuscript was to:

(i) update the carbon accumulation calculations for tree biomass in the Holos whole-farm model using circumference instead of age driven algorithms, as suggested by Amichev et al. (2017);
(ii) develop a user friendly software interface that is simple to use and yet offers sufficient flexibility to reflect the diversity and complexity of existing shelterbelts;
(iii) test the model's applicability by estimating the carbon storage of existing (measured) shelterbelts and determine the minimum requirements for model inputs;
(iv) compare the model's outputs with literature derived measurements or modeled outputs from process models.

MATERIALS AND METHODS

Site Description

We measured five field shelterbelts consisting of four single and one mixed species near Indian Head, SK, Canada (**Figure 1**). These sites are part of a larger project where the role of shelterbelts and other field boundary habitats (such as, natural field boundaries and road allowances, wetlands, etc.) on crop yield and quality as well as biodiversity and soil health are examined in large-scale monoculture agricultural landscape in Saskatchewan.

For the Holos model interface, we only used planted field shelterbelts to calculate the accumulated carbon in these shelterbelts. The shelterbelts consist of Caragana (*Caragana arborescens*), Green Ash (*Fraxinus pennsylvanica*), Colorado Spruce (*Picea pungens*), Siberian Elm (*Ulmus pumila*), and mixed Caragana/Green Ash tree plantings. The details including shelterbelt age and characteristics, soil classification, soil texture, and adjacent crop rotations are provided in **Table 1**.

Measurement Data

We measured tree stem circumference at 1.3 m height aboveground for single stem trees (Colorado Spruce). When multiple stems are present per single tree (Green Ash and Siberian Elm), a cumulative circumference was estimated using:

$$Cumulative\ circumference = \sqrt{\sum DBH_i^2 \times Pi} \qquad (1)$$

where,

Cumulative circumference is the calculated circumference of all stems together (cm).

DBH_i is the diameter (cm) of a stem at breast height (1.3 m).

Pi is the mathematical constant 3.14159.

For Caragana, we measured circumference of all stems of the shrub at 30 cm height aboveground and then calculated the cumulative circumference using the same formula as described above. For both trees and Caragana shrubs, we measured the circumference of every tree and/or shrub for 100 m length starting at one end of the shelterbelt. We recorded missing and dead trees for the shelterbelt mortality calculation (**Table 2**).

To establish a recommendation in the model's interface as to how many woody plants would have to be measured by the model user to achieve a representative average, we applied two methodologies. Using the thinnest and thickest stem of each shelterbelt, we calculated the variance and the average for use in the Student's *t*-test for each respective shelterbelt. With this, we identified the number of required samples for being within a range of 5, 10, 15, and 20% of the actual measured mean with a probability of 80, 90, 95, and 99%, respectively. However, as potential model users (e.g., landowner) are more likely to measure groups of trees in close vicinity rather than observing statistical necessities, we further investigated how closely a rolling average of 3, 5, 10, 15, and 20 trees would approach the average of all measured trees.

The Holos Model (v. 4)

The Holos model is Agriculture and Agri-Food Canada's whole-farm model, designed to answer "what if?" questions with respect to a landowner's management decision effects with regards to the farm's overall GHG budget. For this purpose, the model includes (in version 3) 18 major crops, as well as detailed estimates for beef, dairy, swine, and poultry, and more rudimentary estimates for other livestock. For the calculations of GHG from the different farm components, IPCC Tier 1 emission factors were employed initially, but for soil carbon, soil N_2O, as well as beef and dairy enteric CH_4 emissions, Tier 2 factors were implemented based on peer-reviewed publications. The model's methodology (used equations and publication sources) is freely available upon request[1].

The model's underlying principle of the 'virtual farm' was initiated by Janzen et al. (2006), and resulted in the development of Holos Classic (available upon request) and subsequently Holos version 3 (download[2]). Both models simulate the emissions of a whole farm for 1 year, with the first version offering pre-defined mitigation strategies, and the latter offering a monthly time step for better livestock herd management input. In the outputs, the model lists emissions of N_2O, CH_4, and CO_2, and also converts all emissions into CO_2-equivalents. In the CO_2 emissions, rough estimates for machinery use and irrigation are incorporated; however, the bulk of CO_2 emissions come from upstream emissions (emissions created outside of the systems for inputs that are required for the operation of the system) in fertilizer, pesticide, and electricity production. Agricultural soils are considered to remain in equilibrium until certain management practices occur that cause pre-determined carbon changes (reduction of tillage or summer fallow, and switching from annual to perennial cropping), subsequently output as carbon offsets (negative CO_2 emissions). The offsets also include user defined planted shelterbelts [aboveground C accumulation estimates based on Kort and Turnock (1999)].

Kröbel et al. (2012) argued that a whole-farm model should consider more than just GHG emission estimates, as many practices that aim to lower greenhouse gas emissions may inadvertently cause other impacts on the environment that may be as or even more undesirable. Starting with the implementation of a carbon budget module (Kröbel et al., 2016), the model is undergoing a transformation toward multi-year simulations,

[1] aafc.holos.aac@canada.ca

[2] https://www.agr.gc.ca/eng/scientific-collaboration-and-research-in-agriculture/agricultural-research-results/holos-software-program/?id=1349181297838

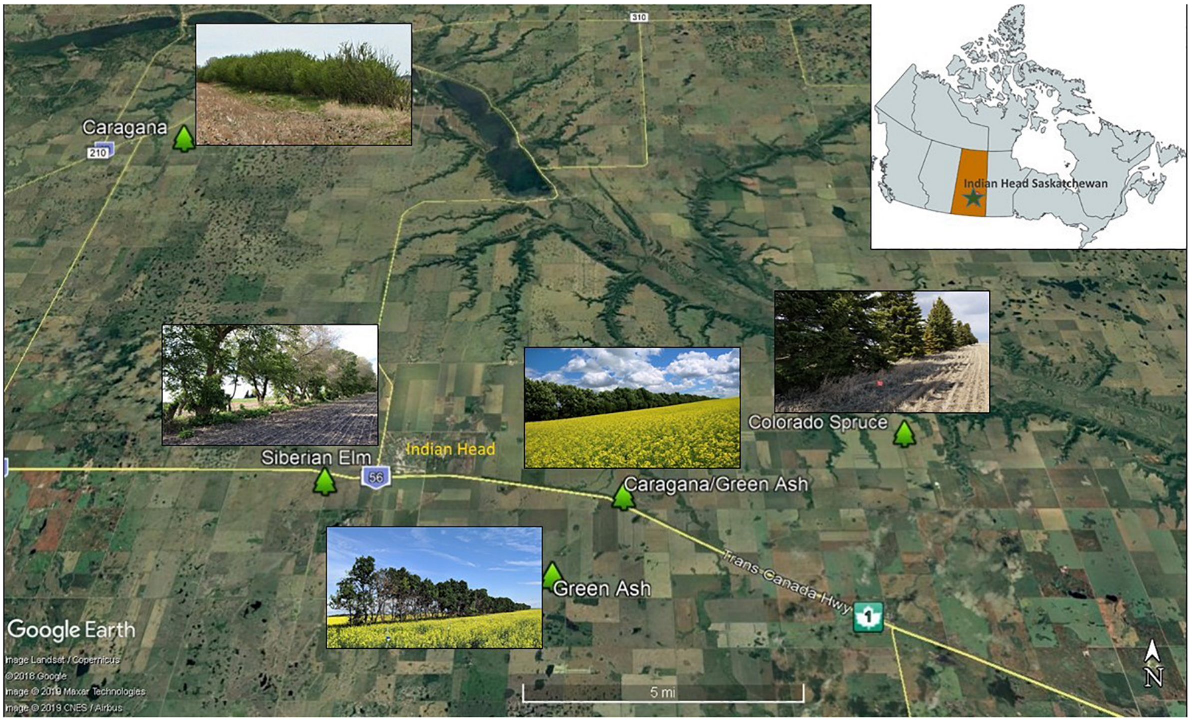


FIGURE 1 | Physical location map of measured shelterbelts near Indian Head, SK, Canada. Inset-map of Canada with a green asterisk showing geographical location of Indian Head, SK, Canada.

and many of its components (cropping, pasture, beef, dairy, and shelterbelts) are being updated in the process. The model is being written in C# (C Sharp) using an agile software development approach, and accordingly, the interface development is being stakeholder driven (through online meetings with potential end users who provided feedback on multiple iterations of the interface).

As the model's primary users are scientists (the model is used in several AAFC and university led projects), farmers (small enterprises have used the model to direct-market their product as carbon neutral), and policy makers (the carbon credit program of Alberta province is based on Holos), a versatile interface is needed to fit the different requirements by the users and their potential knowledge level of the required inputs. For farmers, time requirements and simplicity are important factors. Regardless, for some operations they prefer a lot of detail to represent their farm as best as possible (e.g., beef producers asked for more detailed herd management inputs). The level of details required is similar to scientists who will use specific measurements to feed into the model. In both cases, however, in order to conduct a whole-farm analysis, some input requirements may be missing, which is where the model database attempts to provide representative average values to the model users. These are then also required by policy makers who do not have access to any farm level data, and who use the model to assess the effect of policy initiatives onto average (representative) farm systems.

Therefore, the interface of Holos version 4 is structured into three main stages: a current state, a timeline (for past and future states), and a detail input. The first stage provides an estimate of current GHG emission and potential offsets, while the timeline allows looking back (and forward) to see how patterns evolved over time (Kröbel et al., 2016), this includes the emissions estimates of nitrous oxide, methane, carbon dioxide, and ammonia, as well as leaching of nitrate. Both stages ask for long-term averages to ease user requirements and to better demonstrate the effect of management choices. The last stage allows the input of annually specific values (e.g., based on measurements) for more detailed investigations.

With respect to the shelterbelt component (**Figure 2**), the model asks for the number of rows, the species within each row, the length of each row, and the number of trees/shrubs in each row to calculate the present carbon storage in the shelterbelt as a way to 'offset' other emission sources. In the 'timeline' stage, additional input parameters are required: the planting year of the shelterbelt and the number of trees planted. Thus, the user can explore how carbon was accumulated in the shelterbelt over time, and the user can explore which species to plant in a new (renewed) shelterbelt when, for instance, focusing on carbon capture. In the detail input, number of trees, their circumference, and the row length can be adjusted for each individual year, which would allow for more locally specific and accurate estimates (if available).

Throughout the development, model stakeholders were engaged during online meetings to review the progress on the

TABLE 1 | Shelterbelt age, characteristics, soil classification and texture of studied shelterbelts near Indian Head, SK, Canada.

Tree type	Shelterbelt description		Soil classification and texture	Notes
Colorado Spruce	*Year planted*	1982	Ih6T Black soil (*Rego Black Chernozem*). Clay to heavy clay surface texture	Healthy and dense with occasional gaps due to mortality from past flooding.
	Shelterbelt length (m)	716		
	Shelterbelt width (m)	7.7		
	Average DBH (cm)	27.4		
	Mortality rate (%)	14.8		
Green Ash	*Year planted*	1992	Eg1 Black soils (*Orthic Black Chernozem*). Loam to clay loam surface textures.	Poor health condition due to spray damage caused by application of glyphosate in the adjacent crop; high porosity on lower half.
	Shelterbelt length (m)	706		
	Shelterbelt width (m)	4		
	Average DBH (cm)	18.8		
	Mortality rate (%)	35.7		
Caragana (C)/ Green Ash (GA)	*Year planted*	1990	Eg1 Black soils (*Orthic Black Chernozem*). Loam to clay loam surface textures.	Caragana created a dense understory and formed a healthy contiguous mixed species shelterbelt.
	Shelterbelt length (m)	1293		
	Shelterbelt width (m)	5		
	Average DBH (cm)	13 (C)* 19.1 (GA)*		
	Mortality rate (%)	24.5 (C)* 77.6 (GA)*		
Siberian Elm	*Year planted*	~1990	Ox10 Black soil (*Orthic Black Chernozem*). Loam surface texture.	The trees are pruned heavily and understory vegetation is mostly cleared. Trees are healthy with occasional gaps throughout the length.
	Shelterbelt length (m)	1109		
	Shelterbelt width (m)	3		
	Average DBH (cm)	31.8		
	Mortality rate (%)	15.8		
Caragana	*Year planted*	1996	Ba4 Black soils (*Rego Black Chernozem*) with clay loam surface textures.	The shelterbelt is healthy and contiguous.
	Shelterbelt length (m)	752		
	Shelterbelt width (m)	5		
	Average DBH (cm)	11.0		
	Mortality rate (%)	18.1		

**A simulated 'mortality' was used to modify the total number of shelterbelt trees/shrubs.*

interface development, with a total of 4 online meetings taking place. Only the final results (**Figure 2**) are shown.

Equations and Calculation Procedures of the New Shelterbelt Component

To update the allometric equation from Kort and Turnock (1999), we adopted the relationships identified by Amichev et al. (2017), derived from a dataset of measured shelterbelts in the province of Saskatchewan, Canada. The updated relationships use the circumference of the tree trunk (measured at 1.30 m height outside tree bark) to calculate the aboveground accumulated carbon in the living tree biomass (Amichev et al., 2017), rather than using the age of the tree (Kort and Turnock, 1999), a much less reliable method and a difficult to ascertain value in hindsight.

$$C_{tree} = Carbon_{concentration(trees)} * \left(a \times \frac{tree\,circumference}{3.14159}\right)^b \tag{2}$$

$$tree\,circumference = \sqrt{\sum (circumference_i)^2} \tag{3}$$

where,

C_{tree} Above-ground C stocks per tree (kg C tree^{-1}).

$Carbon_{concentration(trees)}$ Carbon concentration of all tree parts (kg kg^{-1}) set to 0.5 kg kg^{-1} (Kurz et al., 2009).

a Coefficient a (**Table 3**).

b Coefficient b (**Table 3**).

tree circumference cumulative tree stem circumference (cm) at 1.30 m tree height (breast height) measured outside tree bark.

$circumference_i$ circumference (cm) at breast height of each individual stem i (i = 1,2,. . .,n) of a tree with multiple stems.

Using Eqs 2 and 3 to calculate the carbon accumulation in a single tree, row length and planting density will provide the number of trees to be considered (**Table 3**). To drive the model's calculations, our team attempted to limit inputs to data that every-day-users can easily obtain. To start assessing the current state, the row length, number of trees (or average spacing), and average circumference (at 1.30 m breast height for trees and 30 cm height for Caragana) are required. With this information, the model calculates the currently accumulated carbon in the aboveground biomass of a single tree, which is used for a series of look-up values in the shelterbelt database.

Lookup Database for Past and Future Shelterbelt Growth and C Additions

As the allometric equations cannot be used to calculate the belowground biomass of a tree, we are relying on previous (3PG) model simulations (Amichev et al., 2016) for an estimate of tree age specific above-/below-ground biomass fractions to

TABLE 2 | Calculated circumference on the basis of cumulative basal area (mm), using measurements from different shelterbelts near Indian Head, SK, Canada.

Tree #	Caragana	Green Ash	Siberian Elm	Colorado Spruce	Caragana/ Green Ash	Tree #	Caragana	Green Ash	Siberian Elm	Caragana/ Green Ash	Tree #	Caragana	Caragana/ Green Ash	Tree #	Caragana/ Green Ash
1	216.0	0.0	136.3	105.2	0.0	***51***	329.9	51.5	110.5	214.0	***101***	232.5	738.1	***151***	383.5
2	86.5	0.0	165.7	0.0	526.9	***52***	331.8	46.2	84.0	321.7	***102***	398.9	417.5	***152***	873.1
3	236.1	0.0	0.0	69.7	145.0	***53***	633.0	83.8	93.3	555.4	***103***	359.3	301.2	***153***	492.3
4	170.3	47.1	121.6	67.9	616.2	***54***	312.0	49.5	0.0	561.0	***104***	0.0	273.7	***154***	199.6
5	269.5	0.0	152.1	72.3	367.7	***55***	358.7	34.6	102.0	538.3	***105***	621.9	464.3	***155***	601.3
6	213.9	0.0	121.1	98.3	366.1	***56***	258.7	66.9	91.8	335.2	***106***	347.1	455.8	***156***	695.5
7	0.0	64.8	120.9	96.8	396.4	***57***	374.1		40.0	436.7	***107***	432.0	438.3	***157***	414.7
8	270.7	59.7	88.0	0.0	690.8	***58***	0.0			104.9	***108***	439.8	445.9	***158***	265.7
9	242.0	0.0	100.0	97.1	147.3	***59***	428.5			642.4	***109***	157.4	329.5	***159***	443.9
10	407.9	45.6	49.0	94.2	370.0	***60***	452.3			134.0	***110***	214.1	985.3	***160***	424.1
11	106.9	44.0	109.0	37.6	296.9	***61***	382.6			625.2	***111***	383.0	126.5	***161***	666.9
12	0.0	60.7	76.0	95.5	622.0	***62***	387.6			123.0	***112***	483.9	197.3	***162***	478.3
13	446.4	66.0	52.0	0.0	484.2	***63***	591.6			656.8	***113***	217.4	481.8	***163***	565.4
14	304.0	0.0	119.6	63.4	328.0	***64***	286.4			404.5	***114***	256.2	678.9	***164***	473.0
15	295.5	34.2	101.0	84.8	460.4	***65***	336.0			385.1	***115***	0.0	422.1	***165***	628.1
16	322.4	73.2	90.0	93.9	187.7	***66***	346.9			416.6	***116***	0.0	551.7	***166***	631.9
17	92.0	60.0	124.5	72.3	421.2	***67***	408.9			510.0	***117***	159.3	326.1	***167***	405.3
18	86.0	68.3	0.0	92.0	971.1	***68***	645.8			360.7	***118***	336.7	707.4	***168***	470.7
19	140.3	64.7	114.6	77.0	431.1	***69***	427.6			0.0	***119***	355.4	163.2	***169***	600.1
20	302.8	80.8	135.8	92.4	251.9	***70***	275.2			521.5	***120***	549.4	420.6	***170***	574.4
21	421.3	33.6	129.0	0.0	325.2	***71***	259.6			490.0	***121***	504.5	104.4	***171***	492.4
22	0.0	67.4	99.0	108.7	469.5	***72***	371.8			406.6	***122***	0.0	813.7	***172***	293.1
23	299.3	45.9	47.0	82.3	175.9	***73***	398.2			159.8	***123***	0.0	517.3	***173***	658.2
24	352.4	72.6	95.0	86.1	224.8	***74***	486.2			110.6	***124***	0.0	124.5	***174***	605.8
25	279.3	0.0	160.7	83.9	615.8	***75***	0.0			0.0	***125***	391.7	307.0	***175***	225.1
26	330.2	62.4	119.0	116.2	0.0	***76***	416.8			776.0	***126***	0.0	399.0	***176***	482.9

(Continued)

TABLE 2 | Continued

Tree #	Caragana	Green Ash	Siberian Elm	Colorado Spruce	Caragana/ Green Ash	Tree #	Caragana	Green Ash	Siberian Elm	Caragana/ Green Ash	Tree #	Caragana	Caragana/ Green Ash	Tree #	Caragana/ Green Ash
27	241.7	64.4	115.0	91.4	611.4	***77***	330.5			98.9	***127***	0.0	479.9	***177***	552.9
28	281.8	63.1	39.0		271.8	***78***	469.8			207.1	***128***	0.0	534.6	***178***	509.0
29	326.2	79.4	119.4		958.2	***79***	503.2			173.9	***129***	205.6	467.7	***179***	488.3
30	0.0	0.0	117.0		188.6	***80***	0.0			612.4	***130***	195.4	572.0	***180***	590.4
31	143.9	53.7	0.0		266.8	***81***	290.5			462.4	***131***	195.6	573.8	***181***	553.1
32	207.7	0.0	146.7		259.8	***82***	636.4			748.5	***132***	410.6	414.7	***182***	801.5
33	255.5	65.1	0.0		343.2	***83***	0.0			353.3	***133***	471.8	463.4	***183***	333.2
34	420.7	52.5	92.0		541.2	***84***	485.9			287.0	***134***	328.1	668.2	***184***	328.0
35	418.7	0.0	107.0		308.4	***85***	426.9			377.4	***135***	441.8	90.9	***185***	443.0
36	96.4	54.0	55.0		619.0	***86***	668.0			389.6	***136***	0.0	851.8	***186***	707.2
37	0.0	0.0	60.0		664.8	***87***	0.0			634.1	***137***	362.5	494.5	***187***	496.9
38	0.0	0.0	62.0		369.3	***88***	565.1			123.5	***138***	312.2	543.4	***188***	747.7
39	0.0	58.1	181.2		369.6	***89***	0.0			673.8	***139***		482.3	***189***	391.5
40	408.1	74.3	0.0		313.1	***90***	634.6			562.3	***140***		531.7	***190***	567.4
41	0.0	0.0	0.0		463.1	***91***	344.0			473.1	***141***		411.5	***191***	215.7
42	178.8	64.3	56.0		315.1	***92***	389.0			248.0	***142***		507.4	***192***	411.5
43	222.0	72.0	0.0		318.1	***93***	332.9			666.0	***143***		630.8	***193***	
44	149.7	0.0	69.0		486.2	***94***	458.8			513.2	***144***		342.4	***194***	
45	427.3	0.0	76.9		321.3	***95***	310.9			743.5	***145***		174.4	***195***	
46	265.9	0.0	72.9		585.8	***96***	218.7			686.8	***146***		85.8	***196***	
47	323.7	47.1	86.0		285.3	***97***	527.4			731.0	***147***		776.4	***197***	
48	316.0	0.0	84.0		252.1	***98***	0.0			340.6	***148***		433.5	***198***	
49	216.9	0.0	100.8		531.0	***99***	428.2			348.5	***149***		517.9	***199***	
50	284.2	59.0	0.0		615.8	***100***	632.6			432.2	***150***		556.0	***200***	

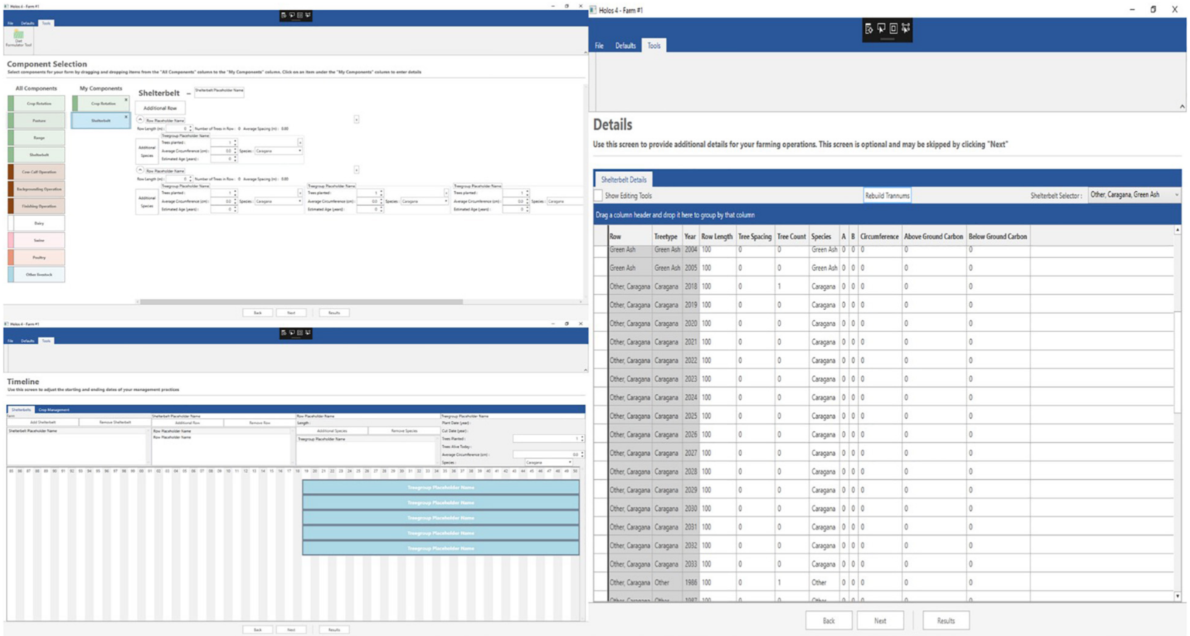


FIGURE 2 | Screenshots of the Holos model interface.

TABLE 3 | Coefficients for above-ground biomass estimation for shelterbelt tree species.

	a	*b*	[a]Diameter (cm)		Spacing (m)	Mortality (%)
			Minimum	Maximum		
White Spruce (*Picea glauca*)	0.0066	3.1832	1.3	38.0	0.5–4.0	0–66
Scots Pine (*Pinus sylvestris*)	0.43264	1.887	17.5	63.0	1.0–3.2	0–50
Manitoba Maple (*Acer negundo*)	0.29428	1.898	3.2	43.6	1.0–5.0	0–47
Green Ash (*Fraxinus pennsylvanica*)	0.20637	2.1217	10.9	37.0	1.0–5.0	0–68
Caragana (*Caragana arborescens*)	0.0284	2.576	5.3	24.2	0.4–2.4	0–29

[a]*Diameter is the estimated cumulative stem thickness of a multi-stem shelterbelt tree, accounting for the stem thickness of all individual stems, measured at 1.3 m height (breast height) along the stem (from ground level) for all tree species, except for Caragana, for which cumulative stem thickness is estimated from diameters measured at 30 cm height along the individual stems (from ground level), derived from Amichev et al. (2017).*

estimate the total tree biomass. This total tree biomass is used to calculate the total shelterbelt biomass, and using tree mortality (the ratio, as percentage, of missing/dead over planted trees) and age, are compared to a lookup table of average shelterbelt biomass amounts. The derived fraction of actual (at the time of observation) versus predicted carbon accumulation is used to back-estimate how the carbon accumulation progressed over time in the past, as well to forecast carbon accumulation into the future.

The average shelterbelt biomass and carbon amounts in the lookup table were previously determined with the tree growth (3PG model) and carbon dynamics (CBM-CFS3 model) models which were adapted for shelterbelt systems by Amichev et al. (2016). In that study, historic climate data were used along with extensive field data to parameterize both models for shelterbelt systems (Amichev et al., 2016). For carbon accumulations in the future, a high (A2) climate forcing scenario was used for the period 2016–2075 (Canadian Centre for Climate Modelling and Analysis [CCCMA], 2017, third generation Coupled Global Climate Model). The values in the lookup tables were used to interpolate the biomass and carbon amounts for any farm's unique shelterbelt design, based on previously generated data for different tree species, ages (1–60 years), spacing (2.0, 3.5, and 5.0 m), and mortality levels (0, 15, 30, and 50%) (Amichev et al., 2017).

The lookup tables in the database also used for mixed shelterbelts through simulated 'mortalities.' For example, the correct number of live trees for the first species ($N_1 = 43$) was calculated by modifying the total number of shelterbelt trees ($N_T = 188$) by a simulated 'mortality' of 77.6%, and for the second species ($N_2 = 145$) it was modified by a simulated 'mortality' of 24.5%. Two of the measured tree species in our measurement dataset are not covered by the database. We hence decided to summarize the estimation methods to create average estimates for coniferous and deciduous trees, respectively. This was done by using data looked up for White Spruce and Scots

Pine individually, and then calculate the average between the two for an average coniferous tree (to be used here for Colorado Spruce). Likewise, Manitoba Maple and Green Ash data were looked up and then were averaged for an average deciduous tree (to be used here for Siberian Elm).

The database contains shelterbelt data of each ecodistrict[3] of Saskatchewan (EcoRegions Working Group, 1989). We assorted and averaged these data according to the established Canadian plant hardiness zones[4] (McKenney et al., 2001) in order to allow the appropriate utilization in other provinces of Canada (with an assumed increase in error that cannot be corrected until more specific data become available). These established averages will serve as a representative growth curve that will be used in estimates over time to (back-) calculate the circumference over time as a fraction determined by the user-supplied current state (e.g., if the current circumference is 50% of the representative growth curve, all past carbon accumulation estimates looked up from the data tables will be reduced accordingly).

RESULTS AND DISCUSSION

Model Input Recommendations

In order to calculate the carbon accumulation in a present shelterbelt, the model requires the user to measure circumference(s) of their tree(s) as an input into the model. While measuring the whole length of a shelterbelt would certainly reacquaint anyone with the state and health of their shelterbelt, it appears an overly expansive ask for the use of a model. We hence set out to investigate what is the minimum required number of tree trunks that would be needed to be measured in order to properly assess the carbon accumulation in the shelterbelt with a degree of certainty (**Tables 4**, **5**).

The statistical analysis suggests that the inherent variability of tree growth within shelterbelts would require a large number of tree measurements to create a close estimate to the real average with a high confidence (**Table 4**). The requirements were much higher for Caragana and Siberian Elm, but considerably lower for Colorado Spruce and Green Ash. Thus, with an expectation that a maximum of 10 trees would be measured by a user on their own volition, an 80% probability would be achieved to be within 15 and 10% of the average for Caragana and Siberian Elm, respectively, while for Green Ash and Colorado Spruce the same number of measurements would give a 90% confidence estimate that is within 10 and 15% of the real mean (**Table 4**).

Assuming that a user would rather measure groups of trees than properly random sampled trees of within a shelterbelt, we calculated the rolling mean of cumulative circumferences within each shelterbelt dataset to see how often a randomly selected group would approximate the real average. Rolling means met the average of the measured tree shelterbelts within a 15% error range quite reliably with seven trees measured (Green Ash, Siberian Elm, and Colorado Spruce in 100, 81, and 100% of the cases), while for Caragana the error range increased to 30% (**Table 5**). Increasing the number of measured trees from 7 to 10 allowed to fit within the above mentioned error ranges more reliably, but did not effectively decrease the error range of the circumference. By increasing the sample size to measuring 15 or 20 trees for an estimation of the average shelterbelt circumference, would decrease the error range to 10% for Green Ash, Siberian Elm, and Colorado Spruce, while for Caragana, the error range would stay at 25%. Based on the findings observed in this study, we recommend that a user would need at least seven trees to measure circumferences of their trees as an input into the model.

Carbon Accumulation Estimates

When assessing the accumulation of carbon in the planted trees and their respective environment, it needs to be taken into account that their respective age is unequal (Caragana shelterbelt being the youngest at 24 years, and the Colorado Spruce shelterbelt being the oldest at 38 years). Furthermore, the single tree growth has to be put into the context of the complete shelterbelt, which requires considering trees that have not survived, carbon deposited through leaf litter, as well as the continuous loss of carbon from the soil (see **Figure 5**). Furthermore, different species have different growth rate patterns and their management (or lack thereof) determines how much of that potential can be realized (**Table 6**). In this sense, however, the results of these estimates are not representative, but rather meant to demonstrate the models capability to assess individual landowner's shelterbelts with sufficient certainty.

The Siberian Elm shelterbelt accumulated the most carbon of all shelterbelts, double than the mixed Caragana/Green Ash shelterbelt, triple of the Colorado Spruce shelterbelt, and more than 10 times the carbon accumulation of the pure Caragana shelterbelt (**Figure 3**). The growth rate of the Siberian Elm also caused to diminish the early growth carbon loss seen for other trees, thus turning the system quickly into a carbon sink after already 3 years (**Figures 4**, **5**). It is remarkable in this sense that the Siberian Elm had, in the total budget, the smallest aboveground fraction contributing to the total (68% of TEC), and the largest dead organic matter accumulation (19% of TEC).

The only shelterbelt somewhat competing with the Siberian Elm was the Caragana/Green Ash shelterbelt (**Table 6**), even though better management (tree survival) and thus realized growth potential meant that the Caragana in mix with Green Ash accumulated double the carbon of the pure Caragana shelterbelt (albeit with six additional growth years, and closer spacing), while the Green Ash accumulated roughly 30% more than the pure shelterbelt stand (**Table 6**). For Caragana, this difference was purely on the basis of surviving shrubs, as the actual carbon accumulation per shrub was equal (**Figure 3**). With respect to TEC, the Green Ash showed similar contributions patterns as the Siberian Elm (70 and 17% for aboveground and dead organic matter), but for the Caragana, a distinctly larger fraction of carbon is in the aboveground biomass (84%) and a much smaller fraction in the dead organic matter (9%).

For the Colorado Spruce shelterbelt, which was the oldest, but also the shelterbelt with the fewest trees, a considerable amount of carbon was still accumulated (**Table 6**). However, downscaled

[3]https://open.canada.ca/data/en/dataset/fe9fd41c-1f67-4bc5-809d-05b62986b26b

[4]http://sis.agr.gc.ca/cansis/nsdb/climate/hardiness/index.html

TABLE 4 | *t*-Test determined required number of samples for a representative estimation of average circumference for different shelterbelt species, based on measured shelterbelts from near Indian Head, SK, Canada.

Within % of mean	20%	15%	10%	5%
Probability level (%)				
Caragana (Caragana arborescens) – Average circumference: 345.1 mm – SD: 132.3 – Variance: 17490.3				
99%	98	162	229	396
95%	25	41	58	99
90%	11	18	26	44
80%	7	11	15	25
Green Ash (Fraxinus pennsylvanica) – Average circumference: 59.2 mm – SD: 12.6 – Variance: 158.9				
99%	30	50	71	121
95%	8	13	18	31
90%	4	6	8	14
80%	2	4	5	8
Siberian Elm (Ulmus pumila) – Average circumference: 99.8 mm – SD: 33.2 – Variance: 1103.7				
99%	69	113	161	277
95%	18	29	41	70
90%	8	13	18	31
80%	5	8	11	18
Colorado Spruce (Picea pungens) – Average circumference: 86.0 mm – SD: 16.7 – Variance: 280.0				
99%	43	71	101	174
95%	11	18	26	44
90%	5	8	12	20
80%	3	5	7	11
Mixed Caragana/Green Ash – Average circumference: 451.1 mm – SD: 188.0 – Variance: 35328.8				
99%	116	191	271	468
95%	29	48	68	117
90%	13	22	31	53
80%	8	12	17	30

on a per tree basis, a Siberian Elm tree stored about 25% more carbon than a Colorado Spruce (which had 8 years more to grow) (**Figure 3**). The Colorado Spruce also requires 15 years to become a carbon sink (**Figure 5**). This may be due to the fact that almost all the TEC is located in the living above- and below-ground biomass (84 and 15%, respectively), while there is almost no dead organic matter accumulated (**Figure 4**). In general, it takes a much longer time for dead organic matter of coniferous shelterbelts (i.e., fallen needles, branches, bark) to decompose and be added into the soil carbon pool, compared to deciduous shelterbelts; this prolonged time for the soil under a Colorado Spruce shelterbelt to act as carbon source is reflected in **Figure 4**.

All the shelterbelts measured in this study were from black soil zones in Saskatchewan, Canada, and were within 100 km distance. Crop management practices are representative of the region; however, there are noticeable differences in the management of shelterbelts among the sites studied. Based on the TEC measured per shelterbelts, Siberian Elm showed the highest potentials of carbon sequestration in this study. If carbon sequestration is the sole objective of a user, then this species would be the best candidate among the species studied. However, shelterbelts provide many other benefits that should not be ignored. While Caragana shelterbelt is found the least potential in terms of TEC, the species provides added benefits by fixing atmospheric nitrogen and a dense vegetation boundary line to protect the crops and soils from wind damage. Another shelterbelt, Colorado Spruce, is a tall evergreen tree and not only protects crops from wind damage but also provides essential habitat for wildlife. Regardless, based on the findings of this study, we are confident of the capability of the model to assess individual landowner's shelterbelts with sufficient certainty.

Biomass Contributions and Other Benefits of Shelterbelts

Historically shelterbelts were planted in the Canadian Prairies since 1903 to protect the soils and crops from wind damage and wind erosion as well as to provide shelter for livestock and farmyards from strong wind during cold winter and hot summer months (Mayrinck et al., 2019). However, with the changes in production technologies, such as adoption of zero tillage and cover crops, the emphasis on the benefits of shelterbelts has declined. Many large landowners view shelterbelts as a barrier in maneuvering large machinery. Large equipment takes a longer time to go around these non-crop areas during the short window of spring and fall farm operations (seeding, spraying, and harvesting). Small landowners, though more likely to retain shelterbelts than larger landowners, may view these areas as non-productive areas and often cleared and converted them

TABLE 5 | Estimating the accuracy of representative measurements of circumference (and derived averages) for different shelterbelt species, using measurements from near Indian Head, SK, Canada.

Average of:		**3 trees**	**5 trees**	**7 trees**	**10 trees**	**15 trees**	**20 trees**
Within range of:							
(%)	**(mm)**						
Caragana (Caragana arborescens)							
5%	17.3	14%	8%	13%	14%	15%	17%
10%	34.5	25%	21%	27%	22%	23%	29%
15%	51.8	34%	43%	44%	41%	39%	45%
20%	69.0	50%	63%	63%	59%	61%	59%
25%	86.3	62%	72%	74%	80%	81%	79%
30%	103.5	72%	83%	84%	91%	89%	95%
Green Ash (Fraxinus pennsylvanica)							
5%	3.0	43%	45%	55%	64%	77%	83%
10%	5.9	60%	82%	94%	100%	100%	100%
15%	8.9	80%	91%	100%	100%	100%	100%
20%	11.8	97%	100%	100%	100%	100%	100%
Siberian Elm (Ulmus pumila)							
5%	5.0	22%	20%	21%	23%	39%	52%
10%	10.0	41%	50%	50%	69%	76%	93%
15%	15.0	46%	64%	81%	87%	100%	100%
20%	20.0	59%	82%	93%	100%	100%	100%
25%	24.9	72%	93%	98%	100%	100%	100%
30%	29.9	85%	95%	100%	100%	100%	100%
Colorado Spruce (Picea pungens)							
5%	4.3	24%	47%	59%	64%	88%	100%
10%	8.6	62%	84%	88%	100%	100%	100%
15%	12.9	90%	100%	100%	100%	100%	100%
20%	17.2	95%	100%	100%	100%	100%	100%
Mixed Caragana/Green Ash							
5%	22.6	19%	23%	27%	27%	26%	25%
10%	45.1	35%	45%	51%	55%	59%	72%
15%	67.7	49%	64%	75%	83%	92%	95%
20%	90.2	66%	84%	87%	94%	100%	100%
25%	112.8	79%	90%	96%	99%	100%	100%
30%	135.3	87%	96%	99%	100%	100%	100%

into croplands to increase production areas. Such activities are responsible for decreasing shelterbelts in the prairie region.

Although shelterbelts may not seem important to many landowners for protecting soils and crops from strong winds, there is still a need to examine other benefits provided by these areas before removing them from farmlands. In a recent study, shelterbelts are shown to improve crop yield by modifying microclimate in the adjacent crops (Osorio et al., 2019). The increase in yield compensated for the footprint of the shelterbelt and yet boosted yield in soybeans and wheat. Shelterbelts provide critical semi-natural habitats to pollinators and other beneficial insects, birds, mammals, and other wildlife within large monoculture fields of agricultural crops (reviewed in Dix et al., 1995; Mize et al., 2008). Alongside the benefits mentioned above, shelterbelts can have a significant effect on mitigating GHG emissions from Canadian agricultural activity (Ward and de Gooijer, 2017). For agroforestry to be successful as a mitigation tool, the plant materials comprising the agroforestry practice must themselves have adaptive capacity to future shifts in conditions due to climate change (Lengnick, 2015). To optimize the potential of agroforestry as a GHG mitigation tool, species selection (e.g., growth speed and lifespan) will be important (Amadi et al., 2016).

Tree species currently used and potentially available for use in agroforestry have potential to be susceptible to erratic and extreme weather events, as well as climate-induced fluctuations in insects and pathogens (Fuhrer, 2003; Allen et al., 2010). For example, a primary species that was historically used in agroforestry plantings, Siberian Elm, is no longer recommended because it is a host for the banded elm bark beetle (*Scolytus schevyrewi*) (Negron et al., 2005) while the recommendation of Green Ash is becoming questionable with the emergence of the emerald ash borer (*Agrilus planipennis*) moving into new ecosystems across Canada.

On the other hand, hybrid poplar (*Populus* spp.) are widely planted as shelterbelts in the Canadian prairies (>5.68 million

TABLE 6 | Carbon accumulation estimates for different shelterbelt of varying age and mortality on a 100 m row length, using measurements from near Indian Head, SK, Canada (Note: the mixed shelterbelt has simulated "mortalities" used to avoid double-counting of live trees of the two species for one and the same planting location).

	Species		**Caragana**	**Green Ash**	**Siberian Elm (av. decid. tree)**	**Colorado Spruce (av. conif. tree)**	**MIXED: Caragana and Green Ash Caragana – Green Ash – Sum Total**		
Data	Age (1–60) =	yr	24	28	30	38	30	30	30
	DBH =	cm	11.0	18.8	31.8	27.4	13.0	19.1	n/a
	Spacing =	m	0.7	1.8	1.8	3.7	0.5	0.5	0.5
	Mortality (0–100) =	%	18.1	35.7	15.8	14.8	24.5	77.6	n/a
	Number of live trees	trees/ 100 m	113	36	48	23	145	43	188
Per-tree	(Abg) Above-ground Biom. C	kgC/tree	6.8	52.4	131.4	118.2	10.4	53.7	n/a
	(Bwg) Roots Biom. C		0.6	10.6	25.1	20.7	0.9	10.4	n/a
	(DOM) Dead Org. Matter C		–0.5	9.2	36.5	1.5	1.1	12.9	n/a
	(TEC) Total Ecosystem C		6.9	72.2	193.1	140.3	12.5	77.0	n/a
Per-shelterbelt	(Abg) Above-ground Biom. C	Mg C/100 m	0.8	1.9	6.3	2.7	1.5	2.3	3.8
	(Bwg) Roots Biom. C		0.1	0.4	1.2	0.5	0.1	0.4	0.6
	(DOM) Dead Org. Matter C		–0.1	0.3	1.8	0.0	0.2	0.6	0.7
	(TEC) Total Ecosystem C		0.8	2.6	9.3	3.2	1.8	3.3	5.1
Results	*Farm Potential (to projected average of the respective provincial cluster)	%	–45.9	20.9	125.6	–43.8	–21.4	277.7	n/a

**Farm potential is estimated from existing shelterbelts as percent increase (positive %) or percent decrease (negative %) of carbon stocks in the farm's shelterbelt, compared to the average shelterbelt carbon stocks for that location (i.e., cluster/soil zone look-up table values). It is estimated as: Potential (%) = 100 * (C_farm – C_cluster)/(C_cluster).*

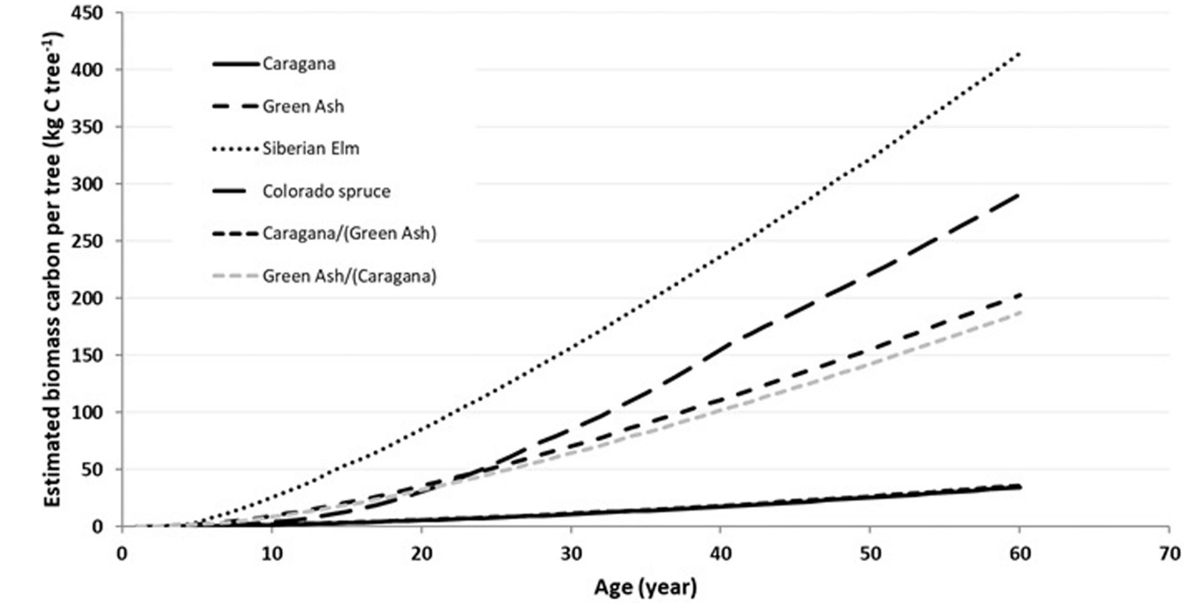


FIGURE 3 | Estimated carbon accumulation in the living biomass of trees grown near Indian Head, SK, Canada, using adjusted values from the representative estimates of Cluster BLK 3 (Amichev et al., 2017).

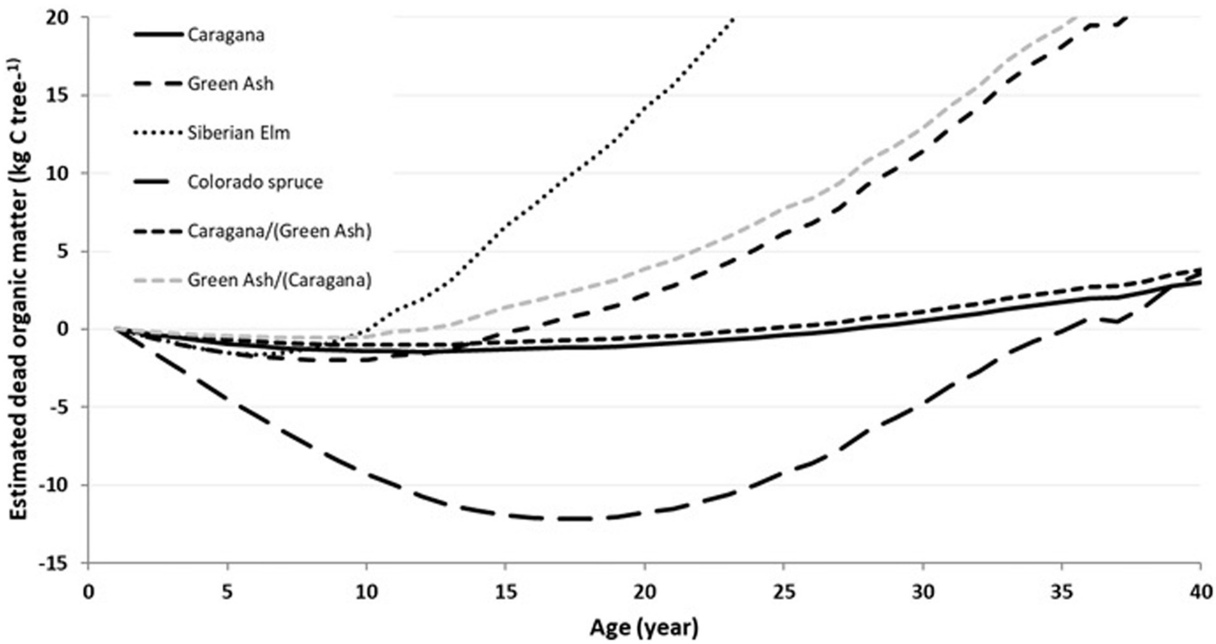


FIGURE 4 | Estimated carbon change in the dead organic matter underneath trees grown near Indian Head, SK, Canada, using adjusted values from the representative estimates of Cluster BLK 3 (Amichev et al., 2017).

trees, 4144 km in length; Amichev et al., 2017) selected to be cold hardy, drought tolerant, pest and disease resilient. At age 60 years, hybrid poplar attain 15–17 m in height with a mean aboveground biomass ranging from 397 to 634 OD Mg km^{-1} and DBH of 52–63 cm. In the current study, we did not measure the circumference of hybrid poplar for use in the Holos model due to the reason that all the shelterbelts included are field shelterbelts. The hybrid poplar shelterbelts available in the region are farmyard shelterbelts that serve a different function, such as protect farmhouse and livestock from wind and cold. For consistency purposes, we compared five field shelterbelts in this study. In the future we plan to include

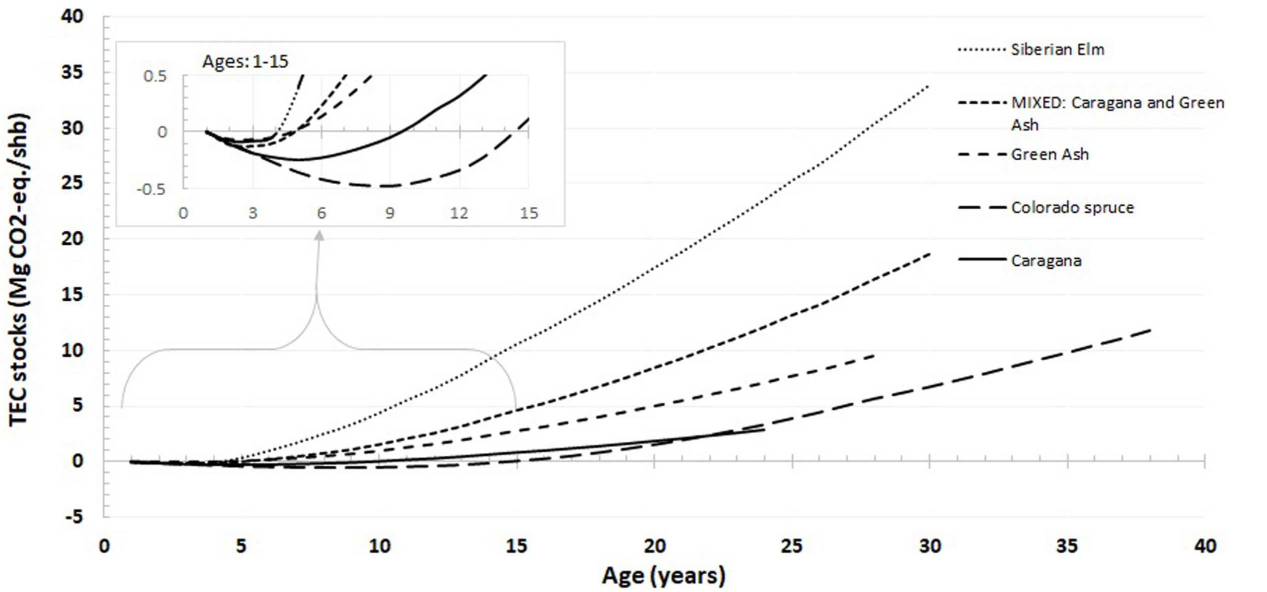


FIGURE 5 | Estimated Total Ecosystem Carbon (TEC) accumulation through different shelterbelts (of different age) near Indian Head, SK, Canada, using adjusted values from the representative estimates of Cluster BLK 3 (Amichev et al., 2017).

hybrid poplar in a broader context because when compared with the five shelterbelt tree/shrub species tested in the Holos model, hybrid poplar display inherent capacity to grow fast with the largest potential to sequester carbon (average aboveground biomass for Green Ash – 32 Mg km^{-1}; White Spruce – 41 Mg km^{-1}; hybrid poplar – 105 Mg km^{-1}; Kort and Turnock, 1999).

It will be essential to determine vulnerability of tree species under modeled climate change scenarios in order to position the necessary production and delivery of suitable plant materials and to provide science-based guidance for plant selection (Ward, 2016). Therefore, a key need is to test a range of woody plant germplasm to identify sources of germplasm that is adapted to both current and future conditions in Canada (Silim, 2004; Johnston et al., 2009). However, since research on climate change adapted plant materials is limited and due to the longevity of woody species one is at the mercy of using diversity as a key principle in developing climate change adapted agroforestry plantings (Schoeneberger et al., 2012) which is fundamentally selecting a variety of plant species that will succeed under shifting weather and climate change conditions. Equally, tree breeding programs can expand selection options, such as the trait-assisted selection from diverse set of germplasm collection (Soolanayakanahally, 2010; Keller et al., 2011) to generate woody feedstocks with high resource-use efficiencies (particularly, water and nutrients) for present and future climates.

CONCLUSION

The whole-farm model Holos was updated with a new allometric equation for its shelterbelt component to more accurately estimate the carbon accumulation in Canadian shelterbelts. Using measured shelterbelts near Indian Head, SK, Canada, the model calculated that a Siberian Elm shelterbelt accumulated most carbon, followed by a mixed Caragana/Green Ash shelterbelt, while single stands of Caragana, Green Ash or Colorado Spruce had poor carbon accumulation in comparison. However, these results are in dependence of actual shelterbelt management (and age) more so than the species selection.

Trees that have excelled in carbon accumulation in the past, may not perform as well in a changing climate, and are already under threat due to invading species. If shelterbelts are to be an active component in our Canadian climate commitments, investment will be needed both to build the genetic potential for continuing tree growth and the distribution of new cultivars across the Canadian landscape. The Holos model can be of assistance to showcase the carbon storage potential to model users, either in farming or policy making.

Going forward in an attempt to utilize shelterbelts as a potential 'negative emissions' sink for Canadian GHG reduction targets, the selection of species needs to be reassessed based on other factors than just the potential carbon accumulation. There has been little documented research on the ecological and economic benefits of shelterbelts in promoting crop productivity and ecological diversity in intensely cropped agricultural landscapes in Saskatchewan. There is a need to measure the benefits or services provided by field shelterbelts, such as increased pollination from native bees and predation of harmful pests by beneficial insects and birds, to determine whether it is advantageous for landowners to maintain these habitat areas on the landscape.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article.

AUTHOR CONTRIBUTIONS

RK and FA were the main authors. JM and AM conceived the software development. YN and LP conducted the shelterbelt measurements. BA guided and reviewed the model updated and provided data access. RS, TW, CL, and KV provided feedback and discussion throughout the work and for the manuscript. All authors contributed to the article and approved the submitted version.

FUNDING

This study was provided by Agriculture and Agri-Food Canada's (AAFC) Canadian Agriculture Partnership (CAP) program for the accomplishment of this work, while underlying data were originally funded through the Agricultural Greenhouse Gas Program (AGGP), also financed through AAFC's CAP program.

ACKNOWLEDGMENTS

We would like to acknowledge funding that contributed to the writing to this manuscript.

REFERENCES

Agriculture and Agri-Food Canada [AAFC] (2018). *Agriculture and Agri-Food Canada. Benefits of agroforestry*. Available online at: http://www.agr.gc.ca/eng/science-and-innovation/agricultural-practices/agroforestry/benefits-of-agroforestry/?id=1344633257343 (accessed August 15, 2019).

Agriculture and Agri-Food Canada [AAFC], Agri-Environment Service Branch [AESB], and Agroforestry Development Centre [ADC] (2010). *Prairie Shelterbelt Program: Frequently Asked Questions*. Ottawa: Government of Canada, 26.

Alemu, A. W., Amiro, B. D., Bittman, S., MacDonald, D., and Ominski, K. H. (2017a). Greenhouse gas emission of Canadian cow-calf operations: a whole-farm assessment of 295 farms. *Agricult. Syst.* 151, 73–83. doi: 10.1016/j.agsy.2016.11.013

Alemu, A. W., Janzen, H. H., Little, S., Hao, X. Y., Thompson, D. J., Baron, V., et al. (2017b). Assessment of grazing management on farm greenhouse gas intensity of beef production systems in the Canadian Prairies using life cycle assessment. *Agricult. Syst.* 158, 1–13. doi: 10.1016/j.agsy.2017.08.003

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., et al. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecol. Manag.* 259, 660–684. doi: 10.1016/j.foreco.2009.09.001

Amadi, C. C., Van Rees, K. C. J., and Farrell, R. E. (2016). Greenhouse gas mitigation potential of shelterbelts: estimating farm-scale emission reductions using the Holos model. *Can. J. Soil Sci.* 97, 353–367. doi: 10.1139/CJSS-2016-2017

Amichev, B. Y., Bentham, M. J., Kulshreshtha, S. N., Laroque, C. P., Piwowar, J. M., and Van Rees, K. C. J. (2017). Carbon sequestration and growth of six common tree and shrub shelterbelts in Saskatchewan, Canada. *Can. J. Soil Sci.* 97, 368–381. doi: 10.1139/cjss-2016-0107

Amichev, B. Y., Bentham, M. J., Kurz, W., Laroque, C. P., Kulshreshtha, S. N., Piwowar, J. M., et al. (2016). Carbon sequestration by white spruce shelterbelts in Saskatchewan, Canada: 3PG and CBM-CFS3 model simulations. *Ecol. Model.* 325, 35–46. doi: 10.1016/j.ecolmodel.2016.01.003

Baah-Acheamfour, M., Chang, S. X., Bork, E. W., and Carlyle, C. N. (2017). The potential of agroforestry to reduce atmospheric greenhouse gases in Canada Insight from pairwise comparisons with traditional agriculture, data gaps and future research. *Forest. Chronicle* 93, 180–189. doi: 10.5558/tfc2017-024

Bastin, J. F., Finegold, Y., Garcia, C., Mollicone, D., Rezende, M., Routh, D., et al. (2019). The global tree restoration potential. *Science* 365, 76–79. doi: 10.1126/science.aax0848

Beauchemin, K. A., Janzen, H. H., Little, S. M., McAllister, T. A., and McGinn, S. M. (2010). Life cycle assessment of greenhouse gas emissions from beef production in western Canada: a case study. *Agricult. Syst.* 103, 371–379. doi: 10.1016/j.agsy.2010.03.008

Bonesmo, H., Beauchemin, K. A., Harstad, O. M., and Skjelvåg, A. O. (2013). Greenhouse gas emission intensities of grass silage based dairy and beef production: a systems analysis of Norwegian farms. *Livestock Sci.* 152, 239–252. doi: 10.1016/j.livsci.2012.12.016

Bonesmo, H., Little, S. M., Harstad, O. M., Beauchemin, K. A., Skjelvåg, A. O., and Sjelmo, O. (2012). Estimating farm-scale greenhouse gas emission intensity of pig production in Norway. *Acta Agricult. Scand. Sec. A* 62, 318–325. doi: 10.1080/09064702.2013.77091

Canadian Centre for Climate Modelling and Analysis [CCCMA] (2017). *The Third Generation Coupled Global Climate Model*. British Columbia: CCCMA.

Church, J. S., Raymond, A. F., Moote, P. E., Van Hamme, J. D., and Thompson, D. J. (2015). Investigating the carbon footprint of cattle grazing the Lac du Bois Grasslands of British Columbia. *J. Ecosyst. Manag.* 15, 1–14.

DiBartolomeis, M., Kegley, S., Mineau, P., Radford, R., and Klein, K. (2019). An assessment of acute insecticide toxicity loading (AITL) of chemical pesticides used on agricultural land in the United States. *PLoS One* 14:e0220029. doi: 10.1371/journal.pone.0220029

Dix, M. E., Johnson, R. J., Harrell, M. O., Case, R. M., Wright, R. J., Hodges, L., et al. (1995). Influence of trees on abundance of natural enemies of insect pests: a review. *Agrofor. Syst.* 29, 303–311. doi: 10.1007/bf00704876

Environment and Climate Change Canada [ECCC] (2019). *Pollutant Inventories and Reporting Division*. National Inventory Report 1990–2017: Greenhouse Gas Sources and Sinks in Canada. Cat. No.: En81-4/1E-PDF. Atlantic: ECCC.

Environment and Climate Change Canada [ECCC] (2020). *Canadian Environmental Sustainability Indicators: Progress towards Canada's greenhouse gas emissions reduction target. Cat. No.: En4-144/48- 2019E-1-PDF*. Available online at: https://www.canada.ca/en/environment-climate-change/services/environmental-indicators/progress-towards-canada-greenhouse-gas-emissions-reduction-target.html (accessed August 14, 2019).

EcoRegions Working Group (1989). *Ecoclimatic Regions of Canada, First Approximation. Ecological Land Classification Series, No. 23*. Ottawa: Environment Canada.

European Environment Agency (2015). *Agriculture and Climate Change*. Available online at: https://www.eea.europa.eu/signals/signals-2015/articles/agriculture-and-climate-change (accessed January 30, 2015).

Fuhrer, J. (2003). Agroecosystem responses to combinations of elevated CO2, ozone, and global climate change. *Agricult. Ecosyst. Environ.* 97, 1–20. doi: 10.1016/s0167-8809(03)00125-7

Gregorich, E. G., Janzen, H. H., Ellert, B. H., Helgason, B. L., Qian, B., Zebarth, B. J., et al. (2017). Litter decay controlled by temperature, not soil properties, affecting future soil carbon. *Glob. Change Biol.* 23, 1725–1734. doi: 10.1111/gcb.13502

Gülzari, ŞÖ, Åby, B. A., Persson, T., Höglind, M., and Mittenzwei, K. (2017). Combining models to estimate the impacts of future climate scenarios on feed supply, greenhouse gas emissions and economic performance on dairy farms in Norway. *Agricult. Syst.* 157, 157–169. doi: 10.1016/j.agsy.2017.07.004

Gülzari, ŞÖ, Ahmadi, B. V., and Stott, A. W. (2018). Impact of subclinical mastitis on greenhouse gas emissions intensity and profitability of dairy cows in Norway. *Prevent. Vet. Med.* 150, 19–29. doi: 10.1016/j.prevetmed.2017.11.021

Guyader, J., Little, S., Kröbel, R., Benchaar, C., and Beauchemin, K. A. (2017). Comparison of greenhouse gas emissions from corn-and barley-based dairy production systems in Eastern Canada. *Agricult. Syst.* 152, 38–46. doi: 10.1016/j.agsy.2016.12.002

Ha, T. V., Amichev, B. Y., Belcher, K. W., Bentham, M. J., Kulshreshtha, S. N., Laroque, C. P., et al. (2019). Shelterbelt Agroforestry Systems Inventory and Removal Analyzed by Object-based Classification of Satellite Data in Saskatchewan, Canada. *Can. J. Remote Sens.* 45, 246–263. doi: 10.1080/07038992.2018.1540297

Howe, J. A. G. (1986). One hundred years of prairie forestry. *Prairie Forum* 11, 243–251.

Intergovernmental Panel on Climate Change [IPCC] (2019). *IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems*. Geneva: IPCC.

Janzen, H. H., Angers, D. A., Boehm, M., Bolinder, M., Desjardins, R. L., Dyer, J. A., et al. (2006). A proposed approach to estimate and reduce net greenhouse gas emissions from whole farms. *Can. J. Soil Sci.* 86, 401–418. doi: 10.4141/s05-101

Johnston, M. H., Campugna, M., Gray, P. A., Kope, H. H., Loo, J. A., Ogden, A. E., et al. (2009). *Vulnerability of Canada's Tree Species to Climate Change and Management Options for Adaptation: An Overview for Policy Makers And Practitioners*. Ottawa, ON: Canadian Council of Forest Ministry.

Keller, S. R., Soolanayakanahally, R. Y., Guy, R. D., Silim, S., Olson, M. S., and Tiffin, P. (2011). Climate-driven local adaptation of ecophysiology and phenology in balsam poplar, *Populus balsamifera* L. (Salicaceae). *Am. J. Bot.* 98, 99–108. doi: 10.3732/ajb.1000317

Kort, J. (1988). Benefits of windbreaks to field and forage crops. *Agricult. Ecosyst. Environ.* 22–23, 165–190. doi: 10.1016/0167-8809(88)90017-90015

Kort, J., Bank, G., Pomeroy, J., and Fang, X. (2012). Effects of shelterbelts on snow distribution and sublimation. *Agrofor. Syst.* 86, 335–344. doi: 10.1007/s10457-011-9466-9464

Kort, J., and Turnock, R. (1999). Carbon reservoir and biomass in Canadian prairie shelterbelts. *Agroforestry Systems* 44, 175–186.

Kröbel, R., Bolinder, M. A., Janzen, H. H., Little, S. M., Vandenbygaart, A. J., and Kätterer, T. (2016). Canadian farm-level soil carbon change assessment by merging the greenhouse gas model Holos with the Introductory Carbon Balance Model (ICBM). *Agricult. Syst.* 143, 76–85. doi: 10.1016/j.agsy.2015.12.010

Kröbel, R., Janzen, H. H., Beauchemin, K. A., Bonesmo, H., Little, S. M., and McAllister, T. A. (2012). A proposed approach to estimate and reduce the environmental impact from whole farms. *Acta Agricult. Scand. Sec. A* 62, 225–232. doi: 10.1080/09064702.2013.770912

Kulshreshtha, S., Van Rees, K., Hesseln, H., Johnston, M., and Kort, J. (2011). "Issues in agroforestry development on the canadian prairies," in *Handbook on Agroforestry: Management Practices and Environmental Impact*, Ed. L. Kellimore (Hauppauge, NY: Nova Science Publishers), 91–127.

Kurz, W. A., Dymond, C. C., White, T. M., Stinson, G., Shaw, C. H., Rampley, G. J., et al. (2009). CBM-CFS3: a model of carbon-dynamics in forestry and land-use change implementing IPCC standards. *Ecol. Model.* 220, 480–504. doi: 10.1016/j.ecolmodel.2008.10.018

Lengnick, L. (2015). *Resilient Agriculture: Cultivating Food Systems for A Changing Climate*. Gabriola Island, BC: New Society Publishers, 288.

Little, S. M., Benchaar, C., Janzen, H. H., Kröbel, R., McGeough, E. J., and Beauchemin, K. A. (2017). Demonstrating the Effect of Forage Source on the Carbon Footprint of a Canadian Dairy Farm Using Whole-Systems Analysis and the Holos Model: Alfalfa Silage vs. Corn Silage. *Climate* 5:87. doi: 10.3390/cli5040087

Little, S. M., Lindeman, J., Maclean, K., and Janzen, H. H. (2008). *Holos - A Tool to Estimate and Reduce GHGs from Farms. Methodology and Algorithms for Version* 1.1.x. Ottawa, ON: Agriculture & Agri-Food Canada, 158.

Mayrinck, R. C., Laroque, C. P., Amichev, B. Y., and Van Rees, K. (2019). Above- and below-ground carbon sequestration in shelterbelt trees in canada: a review. *Forests* 10:922. doi: 10.3390/f10100922

McGeough, E. J., Little, S. M., Janzen, H. H., McAllister, T. A., McGinn, S. M., and Beauchemin, K. A. (2012). Life cycle assessment of greenhouse gas emissions from dairy production in Eastern Canada: a case study. *J. Dairy Sci.* 95, 5164–5175. doi: 10.3168/jds.2011-5229

McKenney, D. W., Hutchinson, M. F., Kesteven, J. L., and Venier, L. A. (2001). Canada's plant hardiness zones revisited using modern climate interpolation techniques. *Can. J. Plant Sci.* 81, 129–143.

Mize, C. W., Brandle, J. R., Schoeneberger, M. M., and Bentrup, G. (2008). "Ecological Development and Function of Shelterbelts in Temperate North America," in *Toward Agroforestry Design-An Ecological Approach*, eds S. Jose and A. M. Gordon (New York, NY: Springer), 27–54. doi: 10.1007/978-1-4020-6572-9_3

Negron, J. F., Witcosky, J. J., Cain, R. J., LaBonte, J. R., Duerr, D. A., et al. (2005). The banded elm bark beetle: a new threat to elms in North American. *Am. Entomol.* 51, 84–94. doi: 10.1093/ae/51.2.84

Osorio, R. J., Barden, C. J., and Ciampitti, I. A. (2019). GIS approach to estimate windbreak crop yield effects in Kansas-Nebraska. *Agroforest Syst.* 93, 1567–1576. doi: 10.1007/s10457-018-0270-2

Petkova, M. (2012). *Bulgarian Research Program for Reducing Methane Production from Ruminants Through Innovation in Their Nutrition*. Bulgaria: Institute of Animal Science Kostinbrod 2232.

Poppy, L. (2003). *Shelter Your Livestock With Trees*. Indian Head, SK: Agriculture and Agri-Food Canada (PFRA).

Prairie Farm Rehabilitation Administration [PFRA] (1980). *Development evaluation study: PFRA Tree Nursery Program1902–1972*. Regina, SK: Government of Canada.

Rempel, J. C. (2013). *Costs, Benefits, and Barriers to the Adoption and Retention of Shelterbelts in Prairie Agriculture as Identified by Saskatchewan producers*. Master Thesis, University of Saskatchewan, Saskatchewan.

Rempel, J. C., Kulshreshtha, S. N., Amichev, B. Y., and Van Rees, K. C. J. (2017). Costs and benefits of shelterbelts: a review of producers' perceptions and mind map analyses for Saskatchewan. Canada. *Can. J. Soil Sci.* 97, 341–352.

Rural Development Institute Shelterbelt Survey (2014). *Prairie Producers' Use of and Attitudes Towards Shelterbelts*. Manitoba: Brandon University.

Samsonstuen, S., Åby, B. A., Crosson, P., Beauchemin, K. A., Bonesmo, H., and Aass, L. (2019). Farm scale modelling of greenhouse gas emissions from semi-intensive suckler cow beef production. *Agricult. Syst.* 176:102670. doi: 10.1016/j.agsy.2019.102670

Schoeneberger, M., Bentrup, G., de Gooijer, H., Soolanayakanahally, R., Sauer, T., Brandle, J., et al. (2012). Branching out: agroforestry as a climate change mitigation and adaptation tool for agriculture. *J. Soil Water Conserv.* 67:128A0136A.

Schroeder, B., Walker, D., Inouye, G., Poppy, L., and Lutz, J. (2011). "Innovative agroforestry designs: ecobuffers," in *Agroforestry: A Profitable Land Use. Proceedings, 12th North American Agroforestry Conference*, eds S. F. Ashton, S. W. Workman, W. G. Hubbard, and D. J. Moorhead (Athens, GA: University of Georgia), 46.

Silim, S. (2004). *Ensuring High Productivity of Alberta's Poplar Plantation Under a Changing Climate. A Progress Report Prepared for the Forest Management Branch Alberta Lands and Forests Division*. Indian Head, SK: Agriculture and Agri-Food Canada, 14.

Skjelvåg, A. O., Arnoldussen, A. H., Klakegg, O., and Tveito, O. E. (2012). Farm specific natural resource base data for estimating greenhouse gas emissions. *Acta Agricult. Scand. Sec. A* 62, 310–317. doi: 10.1080/09064702.2013.777092

Soolanayakanahally, R. (2010). *Latitudinal gradients in adaptive traits of Populus*. PhD Dissertation, University of British Columbia, Vancouver.

Stange, C., and Jackson, M. (2015). *Windbreak Renovation Video*. Fargo: North Dakota State University-Fargo Extension.

StatsCan (2019). *Statistics Canada. 2016 Census of Agriculture*. Available online at: https://www150.statcan.gc.ca/n1/daily-quotidien/170510/dq170510a-eng.htm (accessed August 15, 2019).

Stevenson, L. (2018). *Renovated windbreak for farmer education. Manitoba Co-operator*. Available online at: https://www.manitobacooperator.ca/news-opinion/news/local/renovate-shelterbelts-dont-remove-to-make-them-more-effective/ (accessed August 7, 2019).

Toensmeier, E. (2016). *The carbon farming solution. White River Junction*. Vermont: Chelsea Green Publishing.

Udawatta, R. P., and Jose, S. (2012). Agroforestry strategies to sequester carbon in temperate North America. *Agroforestry Syst.* 86, 225–242. doi: 10.1007/s10457-012-9561-1

United Nations [UN] (2019). *Department of Economic and Social Affairs, Population Division (2019). World Population Prospects*

2019: Highlights (ST/ESA/SER.A/423). New York, NY: United Nations.

United Nations Framework Convention on Climate Change [UNFCCC] (2019). *National Determined Contributions. NDC. Canada's 2017nationally determined contribution submission to the united nations framework Convention on Climate Change*. Rio de Janeiro: UNFCCC.

Waldron, R. M., and Hildahl, V. (1974). *Deterioration of Shelterbelts in Southwestern Saskatchewan*. Edmonton: Canadian Forestry Service, Northern Forest Research Centre.

Ward, T. (2016). *Agroforestry: A land management option for building resilient Canadian agricultural systems in a changing climate*. Unpublished Master's Thesis, University of Missouri, Columbia Missouri.

Ward, T., and de Gooijer, H. (2017). "Expanding the North American Perspective – Canada," in *Agroforestry: Enhancing Resiliency in U.S. Agricultural Landscapes Under Changing Conditions*, eds M. M. Schoeneberger, G. Bentrup, and T. Patel-Weynand (Washington, DC: USDA Forest Service Gen. Tech. Report WO-96), 101–112.

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 24 September 2020
doi: 10.3389/fsufs.2020.512292

Assessing Effects of Agronomic Nitrogen Management on Crop Nitrogen Use and Nitrogen Losses in the Western Canadian Prairies

Symon Mezbahuddin[1,2]*, David Spiess[1], David Hildebrand[1], Len Kryzanowski[1], Daniel Itenfisu[1], Tom Goddard[1], Javed Iqbal[1] and Robert Grant[2]

[1] Environmental Stewardship Branch, Alberta Agriculture and Forestry, Edmonton, AB, Canada, [2] Department of Renewable Resources, University of Alberta, Edmonton, AB, Canada

OPEN ACCESS

Edited by:
Alisa W. Coffin,
United States Department of Agriculture (ARS), United States

Reviewed by:
Laura Ercoli,
Institute of Life Sciences, Italy
Reynald L. Lemke,
Saskatoon Research and Development Centre, Canada

***Correspondence:**
Symon Mezbahuddin
symon.mezbahuddin@gov.ab.ca

Specialty section:
This article was submitted to Agroecology and Ecosystem Services, a section of the journal Frontiers in Sustainable Food Systems

Received: 26 November 2019
Accepted: 14 August 2020
Published: 24 September 2020

Citation:
Mezbahuddin S, Spiess D, Hildebrand D, Kryzanowski L, Itenfisu D, Goddard T, Iqbal J and Grant R (2020) Assessing Effects of Agronomic Nitrogen Management on Crop Nitrogen Use and Nitrogen Losses in the Western Canadian Prairies. Front. Sustain. Food Syst. 4:512292. doi: 10.3389/fsufs.2020.512292

Effective agronomic nitrogen management strategies ensure optimum productivity, reduce nitrogen losses, and enhance economic profitability and environmental quality. Farmers in western Canada make key decisions on formulation, rate, timing, and placement of fertilizer nitrogen that are suitable for soils, weather, and farming operations within which they operate. Suitability of agronomic nitrogen management options are assessed by estimates from linear interpolations and extrapolations of temporally and spatially discrete field-plot measurements of nitrogen responses. Such estimates do not account for non-linear and offsetting biogeochemical feedbacks of nitrogen cycles and cannot provide comprehensive nitrogen budgets for alternative nitrogen management options. These limitations can be overcome by using process-based agro-ecosystem models that adequately simulate basic processes of nitrogen biogeochemical cycles and are rigorously tested against site observations. *Ecosys* is a process-based ecosystem model that successfully simulated the biogeochemical feedbacks among nitrogen, carbon, and phosphorus cycles across different agro-ecosystems. This study deployed *ecosys* to generate spatially and temporally continuous estimates to assess crop nitrogen use and agronomic nitrogen losses from the crop fields across Alberta for alternative nitrogen fertilizer management scenarios. The study simulated effects of four nitrogen management scenarios: fall banded urea, fall banded ESN (Environmentally Smart Nitrogen), spring banded urea, and spring banded ESN on nitrogen recovery and losses from barley fields on mid-slope landforms. These simulations were done at township grids of $\sim$10 km $\times$ 10 km over 2011–2015 utilizing provincial soil and climate datasets. Modeled annual N_2O, N_2, and NH_3 emissions, and nitrogen losses in surface runoff and sub-surface discharge were lower by about 25, 30, 70, and 40%, respectively, with spring banding than in fall banding across Alberta. Modeled barley yields and grain nitrogen uptake were similar in spring and fall banding, indicating agro-economic and environmental sustainability advantage of spring banding in Alberta. These modeled estimates were consistent with estimates based on plot and laboratory research for

Alberta and similar prairie conditions. This study pioneered a methodology of process-based agroecosystem modeling, which is replicable and scalable to assess cumulative impacts of alternative agronomic nitrogen management options on crop production and the environment on provincial, regional, federal, continental, and global scales.

Keywords: nitrous oxide, ammonia, mineralization, grain yield, denitrification, nitrification, volatilization, Alberta

INTRODUCTION

Sustainable fertilizer use management is key for optimizing crop production. Fertilizer nitrogen (N) is a major input for crop production across western Canadian prairies (Shen et al., 2019a,b). About 75% of total Canadian fertilizer N application takes place across the three prairie provinces of Alberta, Saskatchewan, and Manitoba (Statistics Canada, 2016). Alberta contains one third of the agricultural land area in Canada and encompasses a wide range of agro-climatic and soil conditions (Statistics Canada, 2016). The agricultural zone (i.e., white zone) of Alberta encompasses about 42% of total agricultural lands of Canada, which includes about 21 million ha of farmlands (Statistics Canada, 2016). However, nitrogen use efficiency (NUE) on Alberta farmlands is low, as only about 30–50% of the total applied fertilizer N is recovered in crops (Cassman et al., 2002; Janzen et al., 2003). Nitrogen fertilizers not taken up by the crop can either be immobilized by microbes or vulnerable to losses as ammonia (NH_3), nitrate ($NO_{3}-$), nitrous oxide (N_2O), and di-nitrogen (N_2) through volatilization, leaching, and denitrification processes, respectively (Janzen et al., 2003; Qiao et al., 2015; Shrestha et al., 2018; Cui and Wang, 2019; Shen et al., 2019a,b). Losses of N can contaminate water bodies, release potent greenhouse gas (GHG) (e.g., N_2O) and create particulate aerosols (e.g., NH_3) that impact human health (Forster et al., 2007; Qiao et al., 2015; Shrestha et al., 2018; Cui and Wang, 2019; Shen et al., 2019a,b). Minimizing N losses from agro-ecosystems could thus provide opportunities for reducing the environmental footprint of crop production in western Canada (Shen et al., 2019a,b). In addition, reduced fertilizer N loss can enhance fertilizer NUE, which minimizes fertilizer requirements and saves input costs (Grant and Wu, 2008). Optimal fertilizer N management ensures high crop production for human and livestock consumption and is the foundation of value-added sustainable agricultural products. Efficient fertilizer N use would increase consumers' confidence in crop production sustainability and food security and hence would support an increased market access for the producers (Urso and Gilbertson, 2018). Management of the balance between fertilizer N losses and recovery in agriculture is also very critical since humanity has already exceeded the safety threshold of biogeochemical N cycling (Rockström et al., 2009). The need to reduce N losses and increase NUE is one of the top 10 global environmental priorities (UNEP, 2014), which is also a part of the sustainable development goals and aligns with the growing popularity of the cyclical economy awareness of industry and governments (MacArthur Foundation, 2019).

Optimization of agronomic N fertilizer application depends on agro-climatic, soil, crop, nutrient management, and economic variables (Snyder, 2017). In western Canada, farmers' agronomic nutrient planning includes making key decisions on fertilizer products, and timing, placement, and rates of application based on operational variables such as product availability, time, budget, labor, and equipment (Grant and Wu, 2008). The majority of N fertilizers used in western Canada are ammonium-based (Statistics Canada, 2016). Although the majority of N fertilizers in western Canada are applied in the spring, fall application is sometimes preferred to distribute the workload and take advantage of lower fertilizer prices in the fall (Statistics Canada, 2016). However, fall applied fertilizer N is prone to losses, especially during the subsequent spring thaw before the crop can utilize it. After application, N fertilizers undergo hydrolysis and release ammonium (NH_{4+}), which is further oxidized to nitrate (NO_{3-}) by a microbial process called nitrification (Butterbach-Bahl et al., 2013; Bhanja et al., 2019; Shen et al., 2019a,b; Li et al., 2020). Ammonium (NH_{4+}) and nitrate (NO_{3-}) are the forms of N utilized by crops. However, NO_{3-} is highly mobile through the soil and hence more prone to losses through runoff and leaching through the soil especially during spring snowmelt, which can cause surface water and groundwater pollution. NO_{3-} can also be denitrified to produce potent GHG N_2O and inert N_2 gases under saturated conditions (Butterbach-Bahl et al., 2013; Bhanja et al., 2019; Shen et al., 2019a,b; Li et al., 2020). These N losses can be minimized with a spring application that can optimize fertilizer NUE and reduce loss concerns. For instance, N_2O emissions from prairie crop fields can be reduced by up to 30% depending on weather and soil type by avoiding spring thaw following a fall N fertilizer application (Dunmola et al., 2010; Glenn et al., 2012; Li et al., 2012, 2016; Maas et al., 2013).

Fertilizer application placement also plays an important role in crop N use efficiency and agronomic N losses. Although most of the N fertilizers in western Canada are applied in some forms of in-soil banding, broadcasting is still a dominant placement method in large areas of pastures or forages and in split or in-season applications (Grant and Wu, 2008). However, surface broadcasting can be very inefficient agronomically, economically, and environmentally since it can cause up to 50% more N losses through NH_3 volatilization compared to banding (Sheppard et al., 2010). Various enhanced efficiency products such as nitrification and urease inhibitors, and coated urea, have been developed with the objective of reducing N losses and to improve crop N use efficiency (Li et al., 2020). For instance, Environmentally Smart N (ESN, Nutrien) is a polymer-coated urea that is designed to slow down the rate of N release to better match the crop demand than conventional urea, which would improve crop N use and reduce N losses (Cahill et al., 2010; Gao et al., 2015). Selecting the right combinations of fertilizer products, rate, application timing, and placement, i.e., the 4R (Right Source @ Right Rate, Right Time, and Right Place) nutrient stewardship, can thus be an economically viable and environmentally sustainable strategy in western Canadian prairies (Malhi et al., 2001; Grant et al., 2002).

Effectiveness of 4R options for optimizing crop N use efficiency and minimizing N losses varies with variations in soils, landforms, and weather. Suitability assessment of a given combination of 4R for a given weather, landform, and soil condition is a prerequisite before a farmer makes a decision on adopting it. A provincial or prairie-wide numerical inventory of the fate of N applied in the agro-ecosystems would identify dominant regional N loss pathways and direct farmers to the most beneficial N management practices across various landforms, soils, and weather conditions (Dimitrov and Wang, 2019). The fertilizer industry can also use these assessments to identify opportunities of developing and commercializing enhanced efficiency fertilizer products. These estimates can also be scaled-up and displayed spatially across the province to show locations of "hot-spots," which would support governments in targeting and designing appropriate incentive programs and policies. Currently, the effects of various 4R N management combinations on the fate of applied N in agro-ecosystems are being assessed based on site measurements, which are usually temporally and spatially discrete (e.g., Rawluk et al., 2001; Asgedom et al., 2014; Gao et al., 2015). The field measurements of different forms of N losses require linear temporal interpolations and spatial extrapolations for these assessments, which impart substantial uncertainties into these evaluations, since N transformation processes are highly non-linear and often involve offsetting mechanisms (Flesch et al., 2018). Moreover, the site measurements are limited to fewer soil types, weather conditions, and management options for logistic reasons. Often different field projects are intended to evaluate different pathways of N transformations, which makes construction of a comprehensive budget for the fate of applied N into the agro-ecosystems very difficult. Consequently, a comprehensive inventory with temporally and spatially continuous estimates of crop N uses and fertilizer N loss under various agronomic N management does not exist to date for any of the prairie provinces.

Process-based mechanistic agro-ecosystem models can provide spatially and temporally scaled-up numerical estimates of different N pools under different N management scenarios. However, such a process model has to be built upon site-independent algorithms from independent research, which can then be applied to various soil, weather, and agronomic management conditions without site-specific calibration of the model algorithms. The model should be able to reproduce a particular field condition from site-specific model inputs on soils, weather, and land, crop, and nutrient managements rather than tweaking the model codes for each scenario depending on the training dataset. The model outputs of different N pools have to be rigorously validated against site measurements under varying soils, weather, and agronomic management to evaluate model precision. *Ecosys* is such an ecosystem model, which successfully simulated soil–plant–atmosphere N continuum across different agro-ecosystems within and outside western Canada (Grant and Pattey, 1999, 2003, 2008; Grant, 2001; Grant et al., 2006, 2016; Metivier et al., 2009). Building upon those field-level validation studies, this study aimed at deploying the *ecosys* model spatially to generate provincial estimates of crop N use and agronomic N losses for Alberta crop fields under alternative N fertilizer management scenarios. In this study, we describe *ecosys* simulations to assess the effects of four selected N management scenarios: fall banded urea, fall banded ESN, spring banded urea, and spring banded ESN on yield, N uptake, and N losses across dryland barley fields of Alberta. We then corroborate the modeled results against available data and literature values to examine the adequacy of the simulated results in describing spatial distribution of the pathways of movement of applied N within Alberta agro-ecosystems. These selected scenarios serve as prototypes for establishing a scaling-up methodology, which numerically estimate the fate of fertilizer N under various N management scenarios.

METHODS

Model Description

Ecosys is a process-based, hourly time-step, terrestrial, ecosystem model where transformations, transport, and exchanges of N within the modeled ecosystem are simulated in conjunction with those of carbon (C) and phosphorus (P) in a comprehensive modeling scheme, in which they are coupled with soil water, heat, and solute transport (Grant, 2001). Nitrogen transformation processes in *ecosys* are predominantly governed by coupled reduction–oxidation reactions, which result in microbial and root energy yields, decomposition, and growth, and hence drive N mineralization–immobilization, nitrification, and denitrification. Root and mycorrhizal N uptake occur through ion exchange, radial diffusion, and convection, which affects rubisco (ribulose-1,5-bisphosphate carboxylase/oxygenase) activation that drives modeled crop productivity, growth, and yields in *ecosys*. Rubisco activation in *ecosys* can also be affected by functions of water, temperature, and oxygen stresses, and availability of other nutrients such as P. Gaseous and aqueous N losses through NH_3 gas and dissolved organic and inorganic N are also modeled in *ecosys*. Nitrogen availability in the modeled soil solution is also affected by simulated adsorption and desorption of NH_{4+} between soil solution and clay surfaces.

Nitrogen inputs to a modeled ecosystem in *ecosys* include various chemical fertilizer formulations, manure, organic amendments, atmospheric deposition, and biological N_2. Formulation, timing, placement, and rates of a fertilizer application event are explicitly defined by model inputs. Granular NH_3-based fertilizers (e.g., urea) undergo hydrolysis, which controls the rate of N release from fertilizer granules. The hydrolysis process in *ecosys* is a function of soil moisture and temperature that is mediated by microbial activity. The rate of hydrolysis is calculated from a specific rate constant multiplied by total heterotrophic microbial activity, the urea concentration relative to its Michaelis-Menten (MM) constant, and an Arrhenius function of soil temperature, and can be reduced by urease inhibition. The urease inhibition is calculated from another MM function of the aqueous concentration of total active heterotrophic activity that serves as a proxy of urease activity. For ESN, the specific rate constant is one-fourth of the rate constant for urea until 10% of the total applied N is hydrolyzed, after which the rate constants for both ESN and urea become the same. This algorithm simulates an initial lag in N release from ESN as opposed to urea, which approaches a sigmoidal N release response for ESN (Cahill et al., 2010). All of the above algorithms are parameterized from the kinetics and equilibria of complex biogeochemical and eco-physiological processes reported by independent research on ecosystem functioning within a broad scope of spatial scale. These algorithms in *ecosys* thus do not require calibration, training, or parameterizing for each unique space–time scenario. Instead, a modeled agro-ecosystem in *ecosys* is simulated from site-specific model inputs of weather, soil, and agronomic management data. A more detailed description of *ecosys* algorithms representing N transformation, transport, and exchange including all key equations, variable definitions, parameters, and references can be found in Grant and Pattey (2003), Grant et al. (2006), Metivier et al. (2009), and Grant et al. (2002).

Methodology and Model Inputs

The N transformation, transport, and exchange algorithms in *ecosys* were used to derive numerical estimates of effects of N fertilizer timing and products on crop N use and agronomic N losses in the western Canadian province of Alberta. Alberta is the fourth largest province of Canada, which extends between $49°-60°$N and $110°-114°$W occupying an area of 661,848 km² (**Figure 1**). The southern part of Alberta has a semi-arid climate (Köppen climate classification BSk) whereas central and northern Alberta experience humid continental climate (Köppen climate classification Dfb). A total of four parallel sets of model simulations were set up to simulate four selected N management scenarios: fall banded urea, fall banded ESN, spring banded urea, and spring banded ESN. Modeled outputs of N recovery in yield and N uptake of dryland/rain-fed barley; gaseous N losses as N_2O, NH_3, and N_2; and aqueous N losses through surface runoff and subsurface discharge from those barley fields were used to determine the effects of the selected N management scenarios on the fate of N in Alberta crop fields. Each of these four simulations had a total of 3,063 township scale ($\sim$10 km × 10 km) spatially explicit grid cells that spread across the agricultural areas of Alberta (**Figure 2**; **Table 1**). Each of the township grid cells was divided into four landforms: top-, mid-, and foot-slopes, and depressional areas for simulations, depending on slope classification for different landforms (MacMillan and Pettapiece, 2000). Since mid-slope landforms comprise the largest proportion (40%) of the total arable lands in Alberta, we simulated the mid-slope land form elements in this study (**Table 1**). However, the other three landforms will eventually be simulated in future phases of this modeling project to generate more comprehensive scenarios of fate of N in prairie agro-ecosystems. Model inputs of the soil properties represented the key characteristics of four major agricultural soil groups of Alberta, i.e., Brown Chernozems (Aridic Borolls), Dark Brown Chernozems (Typic Borolls), Black Chernozems (Udic Borolls), and Dark Gray and Gray Chernozems and Luvisols (Boralfs and Mollic Cryoboralfs) (**Figure 2**; **Table 1**). The soil properties were derived from the most frequently occurring soil (modal or dominant) profile in each slope position for each township (Protz et al., 1968). The modal soil profiles were selected from the lands falling within Land Suitability Rating System's rating of 2–4 to represent Alberta's best arable lands for spring seeded small grains (Bock et al., 2018; LSRS, 2019).

Each grid cell was seeded with a barley plant functional type (PFT) (**Table 1**). Barley was selected as the model crop for this study since it is the third largest crop grown in Canada, with over 90% of production located in the three Prairie Provinces of Alberta, Saskatchewan, and Manitoba (Statistics Canada, 2016). The barley PFT was built by customizing the wheat PFT in Grant et al. (2011) for eco-physiological adaptation for barley crops in Alberta. The ecological adaptation in the barley PFT were represented by adjusting the crop climate zone adaptation to the appropriate Köppen climate zones for different regions in Alberta. The physiological adaptation in the barley PFT was represented by raising the fractions of leaf proteins in rubisco and in mesophyll chlorophyll by 40%, which would simulate higher productivity and hence more rapid accumulation of grain biomasses in barley than in the wheat PFT in Grant et al. (2011). These adjustments to the barley PFT in *ecosys* with respect to the wheat PFT were made based on relative performance between *ecosys* simulated barley and wheat yields, biomass growth, and nitrogen uptake, which were rigorously tested by Grant et al. (2020) against Alberta field data. Typical soil, crop, and nutrient management practices and recommended N fertilizer rates across different regions of Alberta were used as model inputs (**Figure 2**; **Table 1**). The ratios of organic C to N, and N to P, in each soil layer were assumed as 10–1, which were typical to agricultural soils in Alberta. Sustained grain removal would create P limitation in the modeled crop. To eliminate P limitation, a phosphate fertilizer at a rate of 2.5 kg P ha^{-1} $year^{-1}$ was applied to each grid cell in each simulation along with banded N fertilizers (**Table 1**). The spin-up runs ensured that the modeled ecosystems attained mass and energy balances to represent stable site conditions. All the simulations started with spin-ups from 2001 to 2010, which then extended to simulation runs from 2011 to 2015 using gridded, real-time, daily, weather data, i.e., maximum and minimum air temperature, incoming shortwave radiation, precipitation, wind speed, and relative humidity (**Table 1**) (ACIS, 2019). Since the *ecosys* model is an hourly time-step model, the model inputs of daily weather variables were first scaled down to hourly, to be implemented as hourly vertical model boundary conditions, to drive hourly model calculations. This temporal downscaling of the weather variables from daily to hourly was done internally inside the *ecosys* weather sub-model. The daily incoming shortwave radiation was downscaled to hourly values using a sinusoidal curve for radiation based on day length (**Figure 3**). The maximum and minimum daily air temperatures were used to drive a sinusoidal curve that calculated hourly

FIGURE 1 | A map showing the geographic location of the western Canadian province of Alberta. Labels are the major cities and red lines are major highways of Alberta.

temperature, so that the minimum temperature was reached at dawn and the maximum temperature was reached at 3 h after solar noon (**Figure 3**). Relative humidity was downscaled from daily to hourly by using a similar sinusoidal curve as the temperature (**Figure 3**). The daily precipitation was equally distributed to each hour in a day, and the average daily wind speed was used for each hour of the day as model upper boundary conditions. The temporal downscaling of air temperature and radiation would enable simulation of diurnal variations in N, C, heat, and water balance in the model. However, the temporal downscaling procedure of daily weather variables to hourly values, as described above, could still miss a sudden drop or rise of temperature from 1 h to another in a day, or a large precipitation event that occurred in some hours of a day. Lack of model inputs for these episodic events may affect the model's capability of accurately simulating sudden flushes of N losses (e.g., N_2O). Model vertical boundary conditions, as described by the model inputs of air temperature and precipitation above, adequately represented drier growing seasons in the southern part of the province and long harsh winters, with relatively shorter and wetter growing seasons in the north. This is apparent in the monthly mean temperature and precipitation distribution across Alberta over the period of 5 simulation years from 2011 to 2015 (**Figure 4**).

Validation and Analyses of Modeled Outputs

Enhanced efficiency N fertilizers like ESN differ from conventional urea in their N release patterns from fertilizer granules. The granules of urea are coated with a polymer

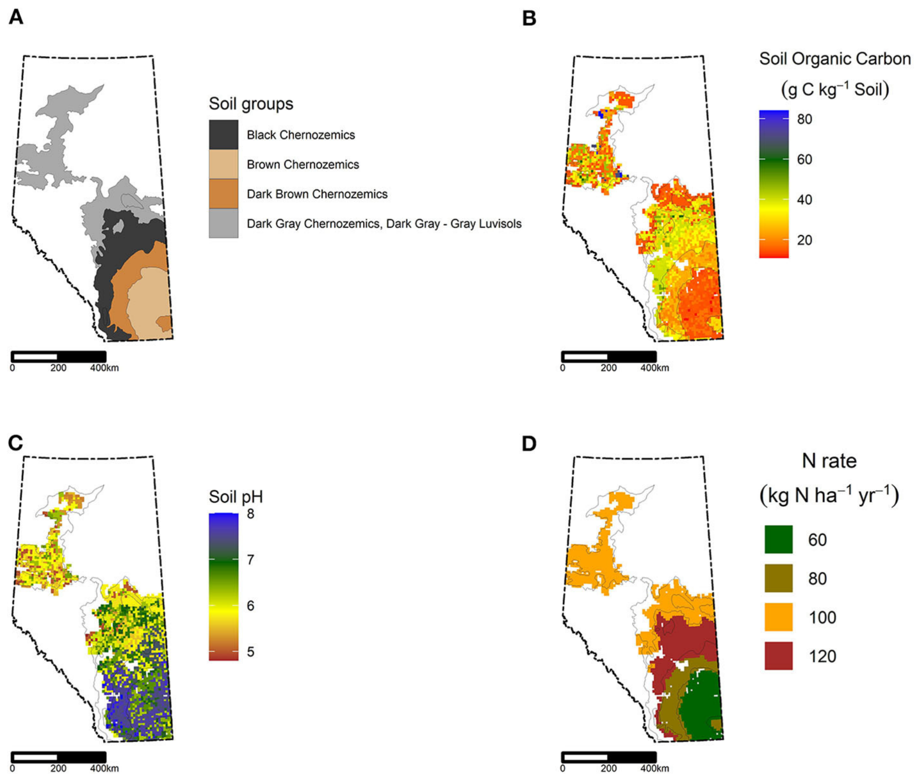


FIGURE 2 | (A) Spatial distribution of the soil groups comprising the agricultural zone of Alberta, which represents the study area for *ecosys* simulation. **(B)** Spatial distribution of top soil (0–15 cm) organic carbon contents, and **(C)** pH of the agricultural zone of Alberta that was simulated in this study. **(D)** Spatial distribution of typical fertilizer nitrogen application rates used in *ecosys* simulations. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary, and the thin black lines within the maps demarcate soil group areas.

to slow down N release rates in ESN fertilizers. This slower release is designed to better match crop N demand to enhance crop N uptake and minimize agronomic N losses. Simulated N release patterns for ESN vs. urea were compared against laboratory data to examine how well *ecosys* simulated the observed differences between ESN and urea in their N release patterns. For this purpose, daily modeled N releases were accumulated and averaged as percentages of total applied N, for all four simulations, for all years, across all soils. The N release percentages of urea vs. ESN were then plotted against thermal time expressed as degree days. The thermal time was cumulative of daily accumulated modeled hourly soil temperatures above 0°C at the depth of N fertilizer banding. The N release to thermal time relationship curves were then compared with similar curves constructed by data from a laboratory experiment.

The laboratory experiment was performed with a commercial top soil "Greensmix" of a sandy loam texture with a pH of 6.6 (Dowbenko, personal communication). Two separate sets of measurements were performed to account for N release from urea vs. ESN granules with a gradual increment and decline of temperature at the rate of about 5°C/week. Each of these sets of measurements was performed under two moisture levels: 50 and 75% of the field capacity (Dowbenko, personal communication). During these laboratory experiments, the soil samples were maintained in sample pots within a growth chamber with designated temperature controls. While preparing the experiment pots, the fertilizer samples were evenly dispersed and covered with a 0.6-cm-thick soil layer. The two stated levels of moisture contents in the pots were checked daily and maintained throughout the experiment period (3–4 months). The rate of fertilizer release for each sample pot was measured once every week. For this purpose, the soil layer at the top of the fertilizer layer was removed very carefully to prevent any damage to the fertilizer granules. The loose soil particles were washed out of the granules by using a gentle stream of deionized water. The granules were then analyzed for N concentration by using colorimetry in aqueous solutions.

Seasonal and interannual variations in weather can significantly affect yield and crop N uptake. Modeled annual barley grain yields at typical grain moisture contents (13.5%) were averaged for each township over the simulation period (2011–2015) to include effects of a range of weather conditions. The averaged modeled barley yields and annual grain N uptake were reported as rates in kg ha^{-1} year^{-1}. The rates of modeled barley yield and grain N uptake were then compared for the four scenarios to assess the effects of N fertilizer timing and products

TABLE 1 | Key model inputs to *ecosys* model to simulate effects of agronomic nitrogen management on crop nitrogen uptake and nitrogen losses from simulated barley fields across the agricultural areas of Alberta during 2011–2015.

Agronomic nitrogen management scenario	Fall banded urea	Spring banded urea	Fall banded Environmentally Smart Nitrogen (ESN)	Spring banded ESN
Nitrogen application timing	Fall (late October)	Spring (prior to seeding)	Fall (late October)	Spring (prior to seeding)
Nitrogen source	Urea		ESN	
Nitrogen application placement	In-soil banding at a depth of 7.5 cm in rows at 25 cm apart from each other, representing typical side-banding practices in Alberta			
Nitrogen application rates	60 (brown soils), 80 (dark brown soils), 100 (dark gray–gray soils), and 120 (black soils) kg N ha^{-1} $year^{-1}$ (**Figure 2**)			
Phosphorus application	2.5 kg P_2O_5-P ha^{-1} $year^{-1}$ for all grids placed with N fertilizers within the same bands			
Crop	Dryland/rainfed barley			
Tillage	No till			
Irrigation	No			
Seeding and harvest dates	Typical seeding and harvesting dates for spring seeded barley in each soil group areas (**Figure 2**)			
Rotation	Continuous field crops			
Residue management	Straw removal (15 cm stubble left on the field after each harvest)			
Spin-up years	From January 1, 2001 to December 31,2010 (hourly time-step simulation)			
Simulation years	From January 1, 2011 to December 31,2015 (hourly time-step simulation)			
Model inputs of weather data	Gridded daily weather data—maximum and minimum air temperature, incoming shortwave radiation, relative humidity, precipitation, and wind speed (ACIS, 2019)			
Implemented weather data into the model	Daily incoming shortwave radiation (MJ m^{-2} d^{-1}) downscaled to hourly radiation (W m^{-2}) Daily maximum and minimum air temperatures (°C) downscaled to hourly air temperature (°C) Daily average relative humidity (%) downscaled to hourly relative humidity (%) Daily average wind speed measured at 10 m (km h^{-1}) downscaled to hourly average wind speed (m s^{-1}) Daily precipitation (mm day^{-1}) were equally redistributed as hourly precipitation (mm h^{-1}) for each of the 24 h in a day (**Figure 3**) (section Methodology and Model Inputs)			
Grid size	~10 km × 10 km (township scale)			
Landform	Mid-slope (MacMillan and Pettapiece, 2000)			
Depth of soil column	1 m (divided into 9 vertical layers)			
Soil properties	Bulk density, soil organic carbon, sand (%), silt (%), pH, and coarse fragments (%) for each of the 9 vertical soil layers in each grid cell (AGRASID, 2019)			
Organic carbon-to-nitrogen ratio	10:1 for each vertical soil layer in each grid cell, which is typical for Alberta agricultural soils			
Organic nitrogen-to-phosphorus ratio	10:1 for each vertical soil layer in each grid cell, which is typical for Alberta agricultural soils			

on the recovery of applied N fertilizer. These comparisons were performed spatially at township scales in maps and also by comparing soil group averages in bar charts.

Modeled barley grain yields were validated against Agriculture Financial Services Corporation (AFSC) of Alberta data across agricultural areas of Alberta for the simulation period (2011–2015). In Alberta, farmers using crop insurance have to report annual crop yields to be in compliance with the AFSC for crop insurance purposes (AFSC, 2019). The AFSC compiles and publishes the reported annual yields for each crop based on an area weighted averaging for a total of 22 risk zones across the agricultural areas of Alberta (AFSC, 2019). In this study, modeled barley grain yields were averaged for all modeled townships that fell within each of the AFSC agricultural risk zones, for all four N management scenarios, over the simulation period to facilitate comparison against AFSC compiled observed barley yields, averaged over the same time period (2011–2015). The comparison between modeled and observed (AFSC data) barley yields would provide a measure of model accuracy in simulating geo-spatial variations in barley yields across Alberta over the simulation period. Model accuracy was evaluated by geo-spatial Pearson's correlation, slope, intercept, and root mean square for errors (RMSE) of linear regression of modeled vs. observed yields. This test would provide details on model accuracy and uncertainties in simulating regional and provincial scale crop yields and N uptake.

Modeled hourly outputs for different forms of agronomic N losses were accumulated annually as rates in kg N ha^{-1} $year^{-1}$ and were also averaged for each grid cell in each simulation over the simulation period. These N losses were mapped and compared to facilitate township scale spatial comparisons among the four N management scenarios. The rates of different forms

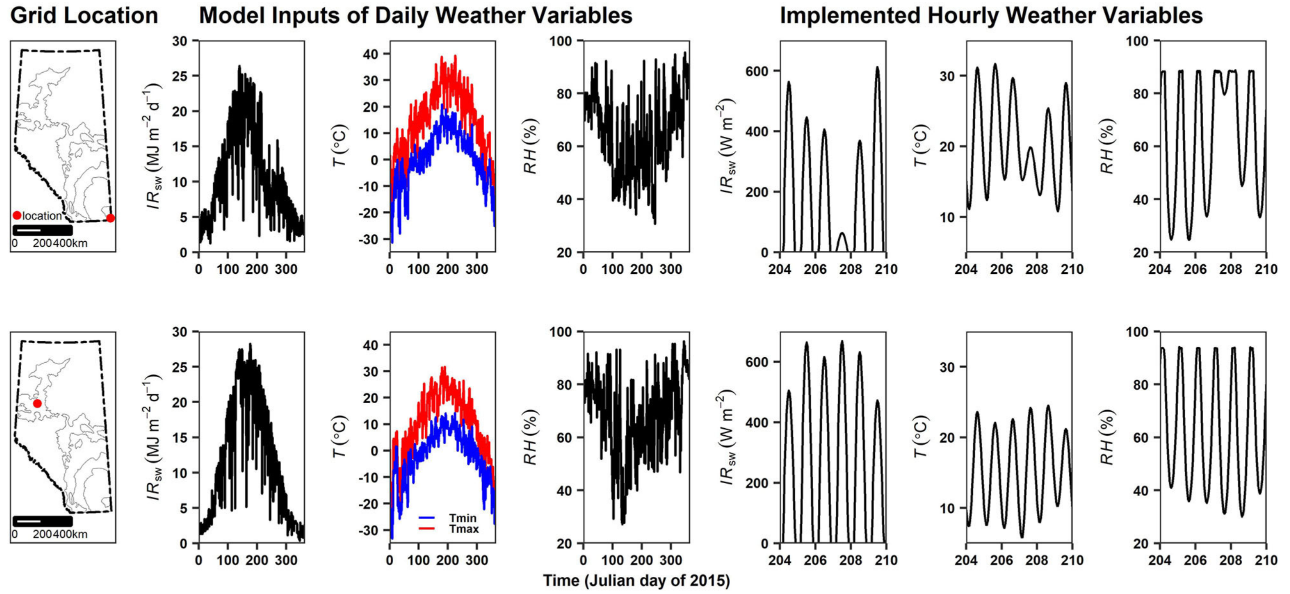


FIGURE 3 | Example representations of temporal downscaling of daily weather data to hourly weather variables to be implemented as hourly model inputs for vertical boundary conditions over two selected township grids using sinusoidal functions. IR_{sw}, incoming shortwave radiation; T, air temperature; T_{min}, daily minimum air temperature; T_{max}, daily maximum air temperature; and RH, relative humidity. The thick dashed lines on the maps represent Alberta's provincial boundary, and the thin black lines within the maps demarcate soil group areas.

of N losses were also averaged and scaled up to the soil group levels to facilitate regional comparisons among different N management scenarios. While averaging by soil groups over the simulation period (2011–2015), standard deviations were illustrated and coefficients of variations were reported to demonstrate the spatiotemporal variations of the modeled N recovery and losses due to variations in soils and weather.

Percent changes in modeled yields and in the key components of modeled N budget for a change in N fertilizer timing or product were listed to facilitate a comprehensive summary of the N management scenario analyses. Modeled trends, and magnitudes and ranges of modeled values for yields, grain N uptake, and various forms of N losses, were also compared with field observations and estimates from available published research for Alberta or similar prairie conditions.

RESULTS

Modeling Nitrogen Release From Urea vs. ESN

Modeled ESN and urea differed from each other in their average N release patterns, which was corroborated well by the observed laboratory results (**Figure 5**). The rate of modeled average N release from urea initially increased rapidly with increasing thermal time, after which it plateaued (**Figure 5**). Averaged modeled N release rate from ESN was slower than that of urea at lower thermal time after which it became very close to that of urea, approaching a sigmoidal N release pattern (**Figure 5**). Modeled N release rates for both fertilizer products for a given exposure to a thermal time varied [Coefficient of variation (CV) up to ±25%] due to variations in soils and weather across Alberta agricultural areas (spatial distribution not shown) (**Figure 5**). The observed laboratory results for urea showed a similar increase with increasing thermal time at a gradually decreasing rate as modeled (**Figure 5**). Observed N release rate for ESN followed an initial lag similar to the N release pattern as modeled (**Figure 5**).

Modeling the Effects of Agronomic Nitrogen Management on Nitrogen Recovery in Barley Grain

Effects of agronomic N management on the recovery of applied N were assessed by the variations of modeled estimates of barley grain yields and N uptake with variations in N application timing and products. Geo-spatial variations in modeled barley yields across Alberta agricultural areas during 2011–2015 corroborated well against observed (AFSC data) barley yields as indicated by a strong geo-spatial correlation between modeled and observed yields (**Figure 6**). However, a slope of 1.1 and an intercept of 538 kg ha^{-1} year^{-1} from a simple linear regression of modeled vs. observed barley yields, meant the modeled barley yields were larger than the observed yields (**Figure 6**). A smaller RMSE of 215 kg ha^{-1} year^{-1} from a simple linear regression of modeled vs. observed barley yields, however, showed lower model uncertainties in predicting geo-spatial variations in barley yields across Alberta during the simulation period (**Figure 6**). Modeled barley yields varied across the province by region, with the drier brown soils having the lowest average yields and the wetter black soils having the highest average yields (**Figures 5**, **7**). Overall, the modeled barley yields remained mostly unaffected, either by a variation in the timing of application (fall vs. spring)

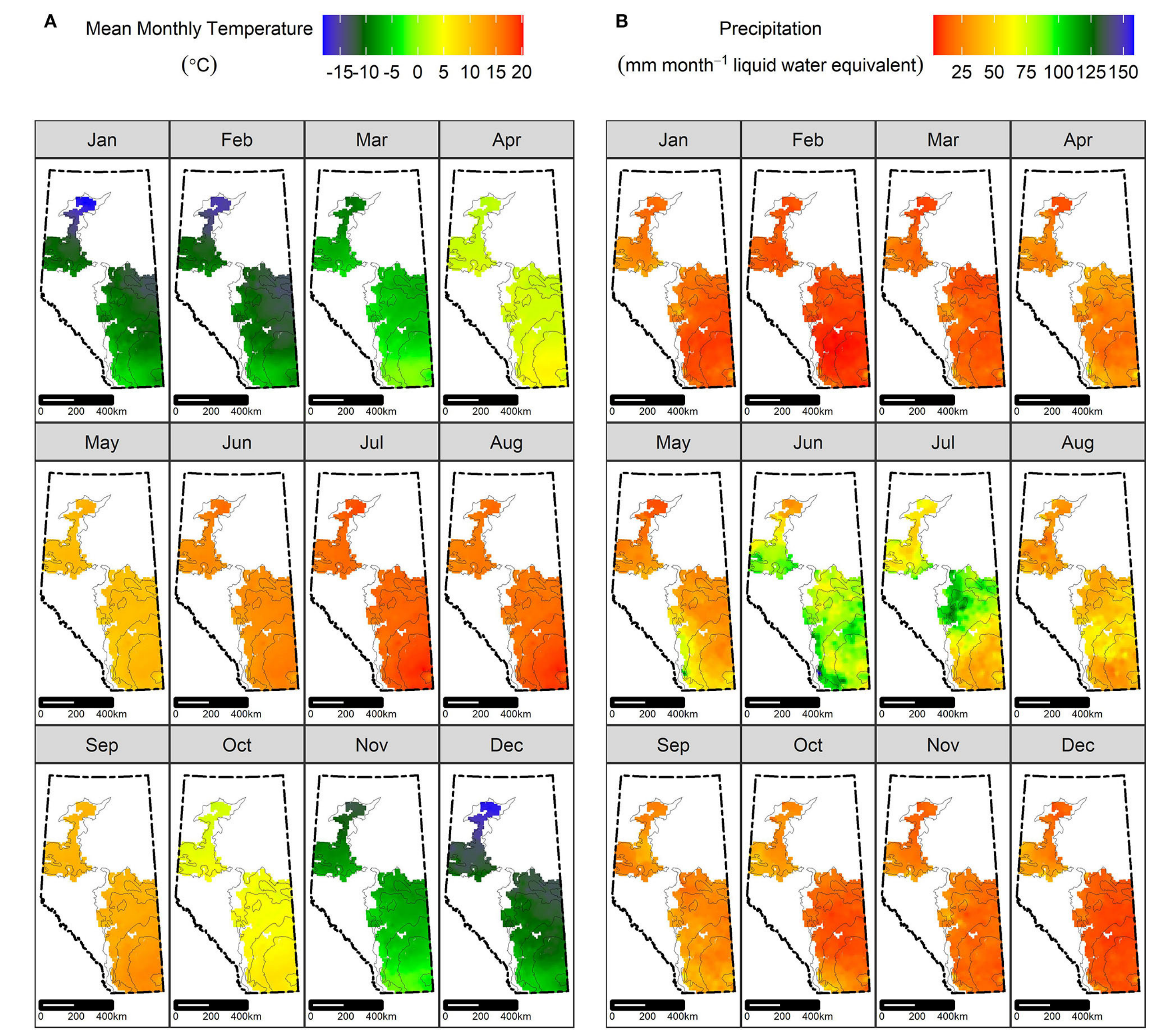


FIGURE 4 | Spatial distribution of **(A)** mean monthly temperature and **(B)** mean monthly precipitation across agricultural areas of Alberta averaged over the simulation period (January 1, 2011–December 31, 2015) (ACIS, 2019). The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary and the thin black lines within the maps demarcate soil group areas.

or by a variation in N fertilizer products (urea vs. ESN) (**Figure 7**; **Table 2**). A change in the application timing from fall to spring resulted in only about 2% overall increase in modeled barley yields across Alberta during 2011–2015 (**Table 2**). Contrary to the expectation that ESN would produce higher yields, modeled ESN application indicated no significant yield effect when compared to modeled urea application across the province (**Table 2**). However, there were localized effects of fertilizer timing and products on modeled barley yields, which was revealed at the township-scale spatial distribution of the modeled barley yields (**Figure 7**). For instance, there was about 20% reduction in modeled barley yields in some parts of the southeast dark gray–gray soil zone caused by a change from fall to spring application (**Figure 7**). These reductions were greater in ESN than in urea (**Figure 7**). In contrast, there were increases in modeled barley yields in some parts of the northwest dark gray–gray soil zone and in the southern dark brown zone, for a change from fall to spring application (**Figure 7**).

Modeled barley grain quality, as represented by grain N uptake (or content), showed a similar spatial pattern as the modeled grain yields, with brown soils having the lowest, and black soils having the highest average grain N uptake (**Figure 8**). Like

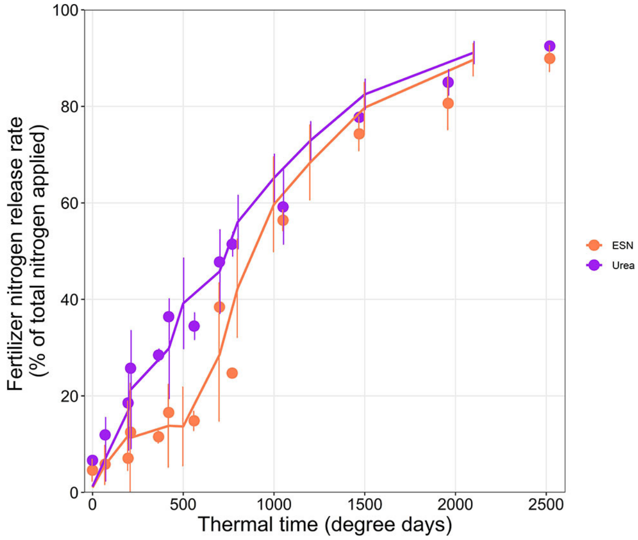


FIGURE 5 | Modeled (lines) and laboratory results (symbols) for nitrogen release with thermal time from urea and ESN applications. Modeled data (lines) were averaged for all the grid cells over the simulation period (January 1, 2011–December 31, 2015) for fall and spring banding across mid-slope landforms of Alberta under simulated barley cultivation. Error bars on modeled N release rates represent standard deviations of modeled N release rates due to variations in weather and soils. Observed laboratory results (symbols) were averaged for four treatments. Error bars on observed N release rates represent standard deviations of observed N release rates due to variations in moisture (section Validation and Analyses of Modeled Outputs).

modeled barley grain yields, grain N uptake did not show any regional or province-wide consistently discernible effects of N fertilizer timing or products (**Figure 8**; **Table 2**). Modeled grain N content was higher by about 2% in the spring application over the fall application and was down by only 1%, while ESN was applied instead of urea across the province (**Figure 8**; **Table 2**). However, the modeled barley grain N uptake also showed similar localized effects of N fertilizer timing and products, as did the modeled grain yields. Modeled grain N uptake declined by about 10–15% from fall to spring applications in parts of the southeast dark gray–gray soil zone and increased in parts of the northwest dark gray–gray and southern dark brown soil zones (**Figure 8**).

Modeling Effects of Agronomic Nitrogen Management on Nitrogen Losses

Variations in N application timing (fall vs. spring) and products (urea vs. ESN) had different effects on modeled N losses in the forms of N_2O, N_2, and NH_3 gases, and N losses in surface runoff and sub-surface discharge. Average modeled annual soil N_2O emissions varied from 0.68 (CV ± 20%) to 1.88 (CV ± 47%) kg N ha^{-1} $year^{-1}$ across the soil groups in all scenarios (**Figure 9**). Modeled average annual N_2O emission varied among soil groups. Black and dark gray–gray soils had higher modeled N_2O emissions than the brown and dark brown soils (**Figure 9**). Modeled N_2O also varied substantially within each soil group area (**Figure 9**). Modeled N_2O emissions were smaller in spring banding than in fall banding, irrespective of fertilizer products (**Figure 9**). On average, modeled N_2O emissions from spring banding was 24% less than that from fall banding throughout dryland barley fields on mid-slope landforms across Alberta during 2011–2015 (**Figure 9**; **Table 2**). Annual reductions in N_2O emissions with a change from fall to spring banding was almost double in black and dark gray–gray soils than those in brown and dark brown soils (**Figure 9**; **Table 2**). However, there was no discernible soil-group-wide or province-wide difference in modeled N_2O emissions for variations in N products from urea to ESN (**Figure 9**; **Table 2**).

Complete denitrification simulated agronomically inconsequential N losses in the form of N_2, averages of which ranged between 0.63 (CV ± 48%) and 1.20 (CV ± 56%) kg N ha^{-1} $year^{-1}$ across the soil groups in all scenarios (**Figure 10**). A change from fall to spring banding reduced N_2-N losses by about 32% across the province (**Figure 10**; **Table 2**). Variations in N products (urea vs. ESN) did not simulate any discernible change in N_2-N losses (**Figure 10**; **Table 2**).

Modeled average volatilization of NH_3 ranged from a consumption of 0.19 (CV ± 150%) kg N ha^{-1} $year^{-1}$ to an emission of 1.18 (CV ± 138%) kg N ha^{-1} $year^{-1}$ across the soil groups in all scenarios (**Figure 11**). On an average, dark gray–gray soils consumed NH_3 from the air, and the other soils emitted NH_3 (**Figure 11**). Spring banding of both fertilizers showed a reduction of NH_3 emissions by about 67% across brown, dark brown, and black soil zones (**Figure 11**; **Table 2**). However, only a 5% reduction in NH_3-N loss was simulated for a change from urea to ESN across these soils (**Figure 11**; **Table 2**). Effects of N fertilizer timing and product on NH_3-N were not relevant for dark gray–gray soils since all the modeled fertilizer management scenarios simulated average consumptions of NH_3 (**Figure 11**; **Table 2**).

Modeled average dissolved organic and inorganic N in surface runoff and sub-surface discharge ranged from 0.32 (CV ± 90%) to 1.15 (CV ± 66%) kg N ha^{-1} $year^{-1}$ across the soil groups in all fertilizer management scenarios (**Figure 12**). Nitrogen losses in surface runoff and sub-surface discharge varied regionally with black and dark gray–gray soils producing higher losses than the other two soils (**Figure 12**). However, there were large variations in modeled N losses in surface runoff and sub-surface discharge within each soil group area (**Figure 12**). Overall, N losses in surface runoff and sub-surface discharge were 37% less in spring banding than in fall banding across the province during 2011–2015 (**Figure 12**). Variations in N products from urea to ESN did not apparently affect N losses in surface runoff and sub-surface discharge (**Figure 12**; **Table 2**).

DISCUSSION

The geo-spatial variations of modeled dryland barley grain yields and N uptake across different soil groups and climates in Alberta during 2011–2015 matched reasonably well against observed variations in barley yields and N uptake reported across Alberta (**Figures 6–8**) (McKenzie et al., 2004; Anbessa and Juskiw, 2012; Perrott, 2016). However, modeled barley yields

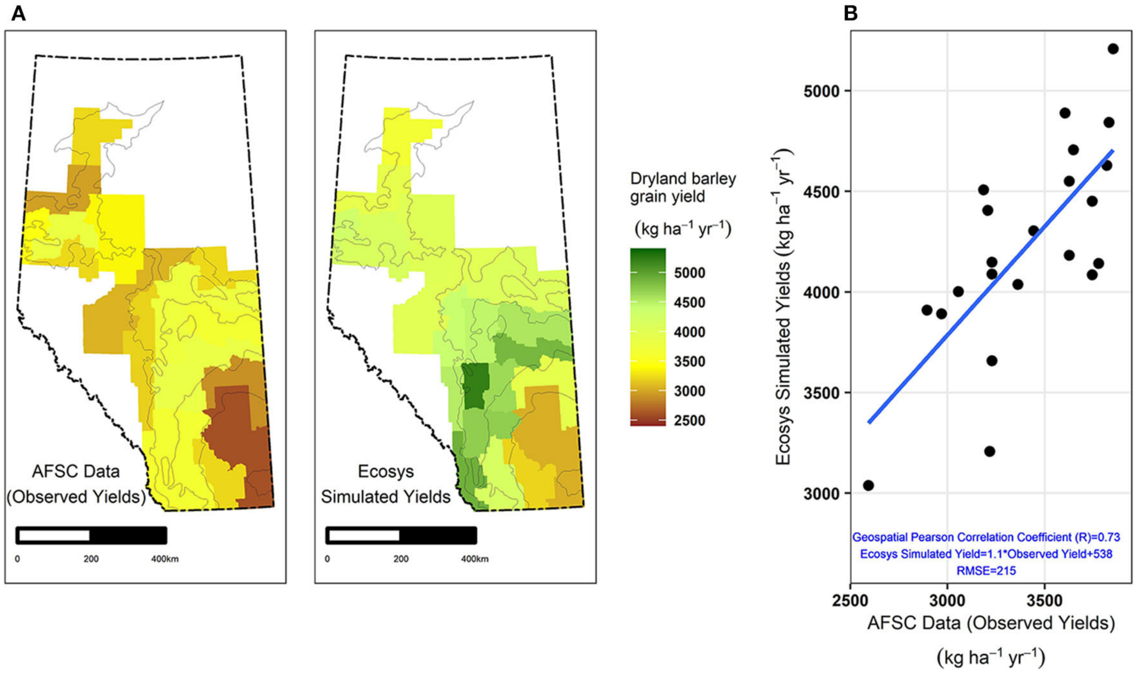


FIGURE 6 | (A) Spatial distribution of observed and modeled dryland or rainfed barley grain yields across Alberta's agricultural areas averaged over the simulation period (2011–2015). Modeled barley grain yields were simulated only for mid-slope landforms. Observed grain yields were averaged (area-weighted) for fields across all landforms as compiled and published by Alberta Financial Services Corporation (AFSC) based on farmers' reported yields (AFSC, 2019). Both the simulated and observed yields were averaged to each of the 22 AFSC designated risk areas (AFSC, 2019). The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary and the thin black lines within the maps demarcate soil group areas. **(B)** Relationship between simulated and observed yields (AFSC data) averaged over the simulation period (2011–2015) for each of the AFSC risk areas. RMSE, root mean square for errors in kg ha^{-1} year^{-1}.

were about 25% larger than the AFSC reported observed yields during this period across Alberta (**Figure 6**). We provide the following explanations for such deviation. The observed yields published by AFSC included reduction in yields due to hail events, insect and pest damages, and lodging that were not simulated in the modeled scenarios (AFSC, 2019). Moreover, the AFSC data included reported yields from landforms that were not modeled in this study (e.g., top- and foot-slope positions and depressional areas) and from fields where lower than recommended N fertilizers may have been applied, which further contributed to the modeled vs. observed grain yield divergence. Model inputs for P fertilizer (**Table 1**) would have alleviated any possible P limitation to the simulated barley yield, which could have also contributed to the larger modeled yields vs. observed AFSC data. Barley yields in top-slope positions could be suppressed by low moisture availability. Yields in foot-slope and depressional area could also be affected by excessive moisture and lodging. These phenomena were not accounted for in these simulations since these simulations included only well-drained, mid-slope landforms, which tend to be higher yielding. Although the modeled dryland barley grain yields were larger than the AFSC data, the modeled grain yields and grain N uptake were well within the ranges of long-term experimental data on barley grain yields (i.e., 4,300–6,900 kg ha^{-1} year^{-1}) and N uptake (i.e., 81–131 kg N ha^{-1} year^{-1}) across brown, dark brown, black, and gray soils of southern and central Alberta (**Figures 7**, **8**) (McKenzie et al., 2004; Anbessa and Juskiw, 2012; Perrott, 2016).

Variations in barley grain yields and N uptake across soil groups were modeled predominantly by adequate simulation of moisture and N availability to the modeled crop (**Figures 6–8**). Brown soils in southern Alberta have the lowest soil organic matter levels and received the lowest fertilizer rates and the lowest mid-growing season (July) precipitation during the simulation period (**Figures 2**, **3**). Low soil organic matter in drier soils provided less substrate for N mineralization in the model, which, along with low fertilizer inputs, caused low available N for modeled crop growth and uptake. Low moisture availability also simulated crop water stress and, hence, a further decline in modeled crop growth and N uptake. Lower N and moisture availability caused lower modeled barley grain yields and N uptake in brown soils compared to the other soils that received relatively higher moisture and N inputs and had higher N mineralization from higher organic matter (**Figures 2**, **3**, **6–8**).

Modeled spring banding produced about 5% higher barley grain yields than fall banding in the dark brown soils, with no apparent change in yields between fall and spring banding in other soils (**Figure 7**; **Table 2**). This is supported by Malhi et al. (1992), whose plot research found that spring banding of

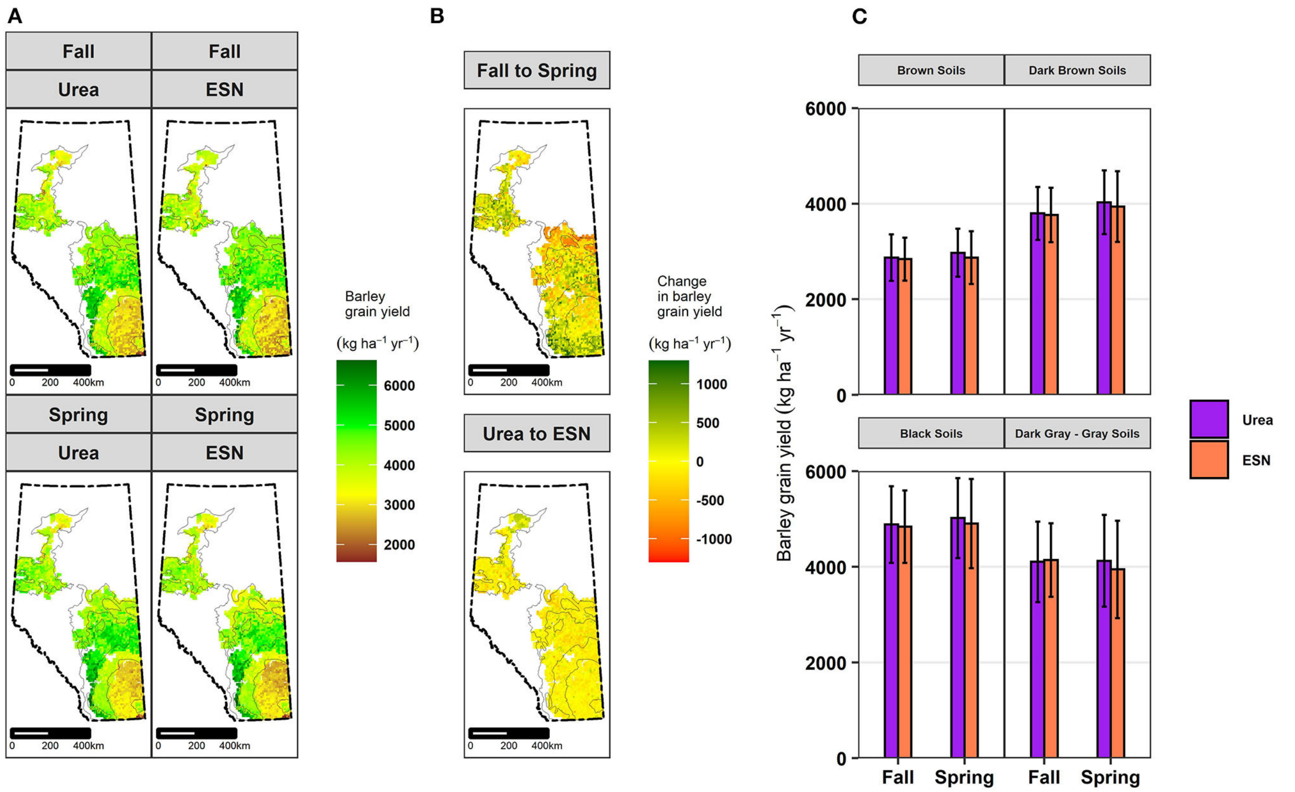


FIGURE 7 | (A) Spatial distribution of modeled dryland or rainfed barley grain yields averaged over the simulation period (2011–2015) for each township for each of the fall and spring banded urea and ESN applications across mid-slope landforms of Alberta under simulated barley cultivation. **(B)** Spatial distribution of differences in modeled dryland barley grain yields for fall vs. spring and urea vs. ESN applications averaged over the simulation period (2011–2015) for each township across mid-slope landforms of Alberta under simulated barley cultivation. A positive change represents an increase in yield for a change from fall to spring or from urea to ESN and vice versa. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary, and the thin black lines within the maps demarcate soil group areas. **(C)** Soil group-wise averages of modeled dryland barley grain yields for fall vs. spring banding and urea vs. ESN applications averaged over the simulation period (2011–2015) for all the townships in each soil group area across mid-slope landforms of Alberta under simulated barley cultivation. Error bars represent standard deviations illustrating spatiotemporal variations in modeled grain yields within each soil group area.

urea fertilizer produced about 8% higher barley grain yield than late fall banding across Alberta. Modeled ESN demonstrated an initial lag response in N release compared to urea, which was corroborated well by independent laboratory studies (**Figure 5**) (Cahill et al., 2010). Despite the slower release, modeled ESN produced similar barley yields and N uptake to those under urea application (**Figures 7**, **8**; **Table 2**). Gao et al. (2015) also found no significant change in spring wheat yields and grain N uptake between banded urea and ESN in two black chernozemic soils in Canadian prairies. Although initial slow release of ESN compared to urea was modeled and measured in laboratory studies (**Figure 5**) (Cahill et al., 2010), there was no field data available to validate the differences in modeled release pattern of N between urea and ESN. While the validation of modeled N release pattern of urea vs. ESN against the laboratory studies facilitated a comparative validation of modeled N release pattern between urea vs. ESN, it is still very important to corroborate the modeled N release pattern against field data where various factors like weather, soil temperature, moisture, land management, and crop uptake interact frequently. Validation of modeled N release against field data in future studies would further reduce the uncertainties in modeling the effects of conventional vs. controlled release N fertilizers and further improve our predictive capacity on the fate of these N management practices.

N fertilizer timing and products affected modeled barley yields and grain uptake differently based on soils and weather conditions. For instance, modeled grain yields and N uptake were lower in spring application than in fall application in some parts of southeast dark gray–gray soil group area (**Figures 7**, **8**). The reduction in grain yields and N uptake from fall to spring were higher with ESN than with urea (**Figures 7**, **8**). This was predominantly modeled from slower crop N availability governed by slower rates of urea hydrolysis as limited by lower microbial activity due to less soil organic C (**Figure 2**). Less over winter and spring precipitation over that area also increased aqueous concentration of microbial biomass, which further inhibited urease activity and hence rates of urea hydrolysis in spring banding scenarios that caused slower N release and

TABLE 2 | Effects of nitrogen fertilizer application timing (fall vs. spring) and products (urea vs. ESN) on modeled annual dryland or rain-fed barley grain yields, modeled annual grain N uptake, and different forms of modeled annual nitrogen losses averaged over the simulation period (2011–2015) for all the townships across mid-slope landforms of Alberta under simulated barley cultivation.

	Change in application timing or product	**Percent (%) change in modeled outputs for a change in application timing or product (a negative value means percent reduction and a positive value means percent increase for an associated change in nitrogen application timing or product)**			
		Brown soils	**Dark brown soils**	**Black soils**	**Dark gray–gray soils**
Barley grain yield	Fall to spring	2.3	5.4	2	−2.1
	Urea to ESN	−1.2	−0.8	−0.7	−2.7
Grain N content	Fall to spring	2.5	5.8	1.3	−2.3
	Urea to ESN	−1.1	−0.6	−0.6	−2.7
N_2O emissions from soil	Fall to spring	−26.2	−18.4	−30.1	−22.6
	Urea to ESN	2.3	1	−1.3	0.4
N_2 emissions from soil	Fall to spring	−38.8	−28.5	−36.3	−24
	Urea to ESN	−0.5	−1.3	0.4	−1
NH_3 emissions from soil	Fall to spring	−69.5	−71.4	−60.6	
	Urea to ESN	−2.2	−5.8	−8.1	
N losses in surface runoff and sub-surface discharge	Fall to spring	−39.9	−35.3	−37.4	−33.6
	Urea to ESN	5.2	4	−2.4	−0.8

reduced modeled grain yields and N uptake compared to those in fall banding scenarios (**Figures 4**, **7**, **8**). Slower N release in ESN than in urea further hindered modeled grain yields and N uptake in spring banded ESN in those areas (**Figures 5**, **7**, **8**). Asgedom et al. (2014) also found that slower release of N from banded ESN reduced spring wheat and rapeseed yields progressively over 2 years in a black chernozemic soil in the prairies. The simulations in this study, however, did not include any blend of ESN and urea, which is becoming a farm practice in some areas of Alberta to overcome the early N needs and to reduce the higher cost of ESN.

On the contrary to the southeast dark gray–gray soil group areas, modeled barley grain yields and N uptake in spring banding were higher than the fall banding in some parts of northwest dark gray–gray and southern dark brown soil group areas (**Figures 7**, **8**). These increases in grain yields and N uptake from fall to spring banding were modeled from rapid N release from spring application as facilitated by adequate microbial activity from higher soil organic C and higher urease activity due to lower aqueous microbial concentration caused by adequate over winter and spring precipitation in those areas (**Figures 2**, **3**, **7**, **8**).

Modeled N_2O-N losses were well within the range of estimates (i.e., 0.1–3.0 kg N ha^{-1} $year^{-1}$) based on linear temporal interpolations of measured N_2O fluxes in no-till fields across different soils and climates of Alberta under spring wheat and barley cultivation (**Figure 9**) (Lemke et al., 1998, 1999; Rochette et al., 2008; Soon et al., 2011). Higher N_2O emissions from black and dark gray–gray soils compared to brown and dark brown soils were simulated predominantly from larger NO_3^- accumulation from greater N inputs and N mineralization in wetter soils that enhanced the denitrification process in saturated soils (**Figure 9**). Like the modeled trend, increased N_2O-N losses from fertilized crop fields were also reported for Alberta and globally with the increase in available substrate (NO_3^-) for denitrification (Rochette et al., 2008; Shcherbak et al., 2014; Chai, 2017). Estimates from periodic field measurements of soil N_2O fluxes showed larger N_2O fluxes in black and gray soils than in brown and dark brown soils, which corroborates very well with the modeled geo-spatial distribution of N_2O emissions from dryland barley fields on mid-slope landforms of Alberta across different soils and climates (Lemke et al., 1999; Rochette et al., 2008). Although modeled N release was slower in ESN than in urea, N_2O emissions did not differ markedly in ESN simulations than in urea simulations for both application timing (**Figures 5**, **9**; **Table 2**). Annual estimates based on periodic field measurements of N_2O fluxes also showed that for recommended application rates of N fertilizer, ESN did not show any significant reduction in N_2O-N losses from spring wheat, barley, and canola fields across Alberta and other prairie provinces as compared to the conventional urea (Li et al., 2012, 2016; Gao et al., 2015). However, estimated annual N_2O-N losses were 15–25% larger in urea than in ESN when application rates were 1.5 times higher than the recommended rates, or when there was considerable seeding delay in the spring that created excessive NO_3^- accumulation from the spring banded N fertilizer (Li et al., 2012, 2016; Gao et al., 2015). Annual modeled N_2O emissions were predominantly contributed by large flushes of N_2O fluxes during spring thaw when inadequate O_2 supply forced modeled microbes to reduce NO_3^- as alternate electron acceptors. Consequently, larger soil N_2O fluxes (by up to 30%) from fall banding than from spring banding were modeled for both urea and ESN applications (**Figure 9**; **Table 2**). Estimates based on field measurements also showed that up to 30% of the total annual N_2O emissions could be contributed by the large flushes of soil N_2O fluxes during winter and spring thaw from fertilized crop fields across Canadian prairies (Dunmola

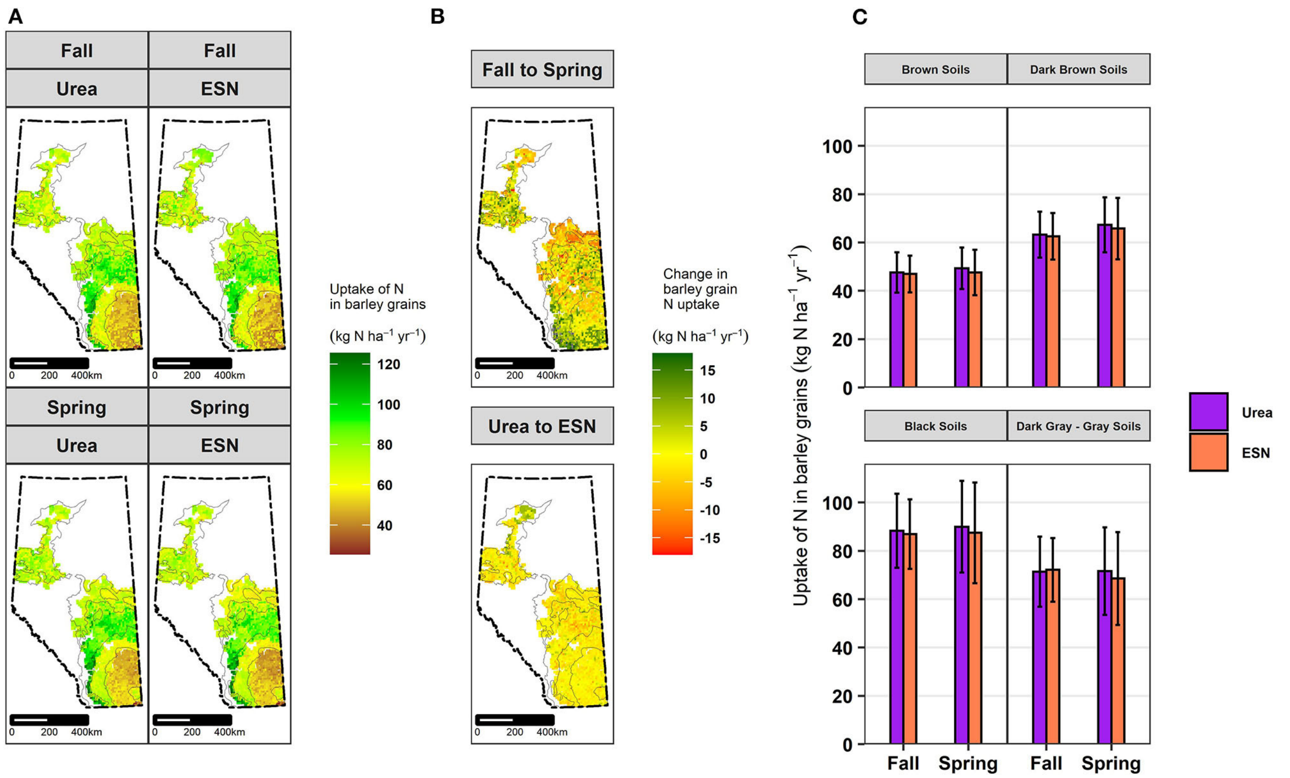


FIGURE 8 | (A) Spatial distribution of modeled annual nitrogen uptake into dryland or rainfed barley grains averaged over the simulation period (2011–2015) for each township for each of the fall and spring banded urea and ESN applications across mid-slope landforms of Alberta under simulated barley cultivation. **(B)** Spatial distribution of differences in modeled dryland barley grain nitrogen uptake for fall vs. spring and urea vs. ESN applications averaged over the simulation period (2011–2015) for each township across mid-slope landforms of Alberta under simulated barley cultivation. A positive change represents an increase in grain nitrogen uptake for a change from fall to spring or from urea to ESN and vice versa. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary, and the thin black lines within the maps demarcate soil group areas. **(C)** Soil group-wise averages of modeled dryland barley grain nitrogen uptake for fall vs. spring banding and urea vs. ESN applications averaged over the simulation period (2011–2015) for all the townships in each soil group area across mid-slope landforms of Alberta under simulated barley cultivation. Error bars represent standard deviations illustrating spatiotemporal variations in modeled grain nitrogen uptake within each soil group area.

et al., 2010; Glenn et al., 2012; Maas et al., 2013). Some estimates based on field data showed that fall banding could cause up to 50% greater N_2O-N losses than spring banded urea and ESN from wheat-barley-canola systems in dark gray soils of Alberta (Soon et al., 2011). From a plot-based periodic flux measurement study, Hao et al. (2001) estimated about 60% greater N_2O emissions with fall broadcasting of ammonium nitrate fertilizer followed by tillage than with similar spring applications. Modeled N_2O emissions varied spatially with NO_3^- availability and degree of soil saturation (**Figure 9**). Higher NO_3^- availability in wetter soils produced large modeled N_2O emissions of up to 4 kg N ha^{-1} year^{-1} in some parts of northern and central Alberta (**Figure 9**). Wetter soils were simulated in those areas during the simulation period (2011–2015) since water inputs through precipitation were greater than vertical water losses through evaporation in the modeled landscapes representing those areas. Based on periodically measured N_2O fluxes, Nyborg et al. (1997) estimated that the N_2O-N losses can be up to 3.5 kg N ha^{-1} within only an 11-day period of spring thaw, further indicating potentials of greater N_2O losses due to NO_3^- accumulation during wetter spring. Current field data-based N_2O-N loss estimates are predominantly based on linear interpolations of periodic flux measurements, which may miss the episodic flushes of N_2O and, hence, may underestimate annual N_2O losses. Flesch et al. (2018) further emphasizes the importance of higher temporal resolution measurements in estimating annual N_2O-N losses while measuring 6-hourly N_2O emissions from barley fields on gray luvisolic soils of central Alberta by using micrometeorological techniques. This study revealed that N_2O-N emissions could accumulate up to 5.3 kg N ha^{-1} from barley fields over only a month during spring thaw, which were larger than the current annual estimates for western Canadian prairies (Flesch et al., 2018). However, maintaining long-term high-resolution N_2O measurements and replicating them spatially are highly demanding of time, technology, and cost. Some of these limitations could be overcome by supplementing current estimates with the modeled annual N_2O estimates in this study that were derived by accumulating hourly calculated outputs (section Validation and Analyses of Modeled Outputs).

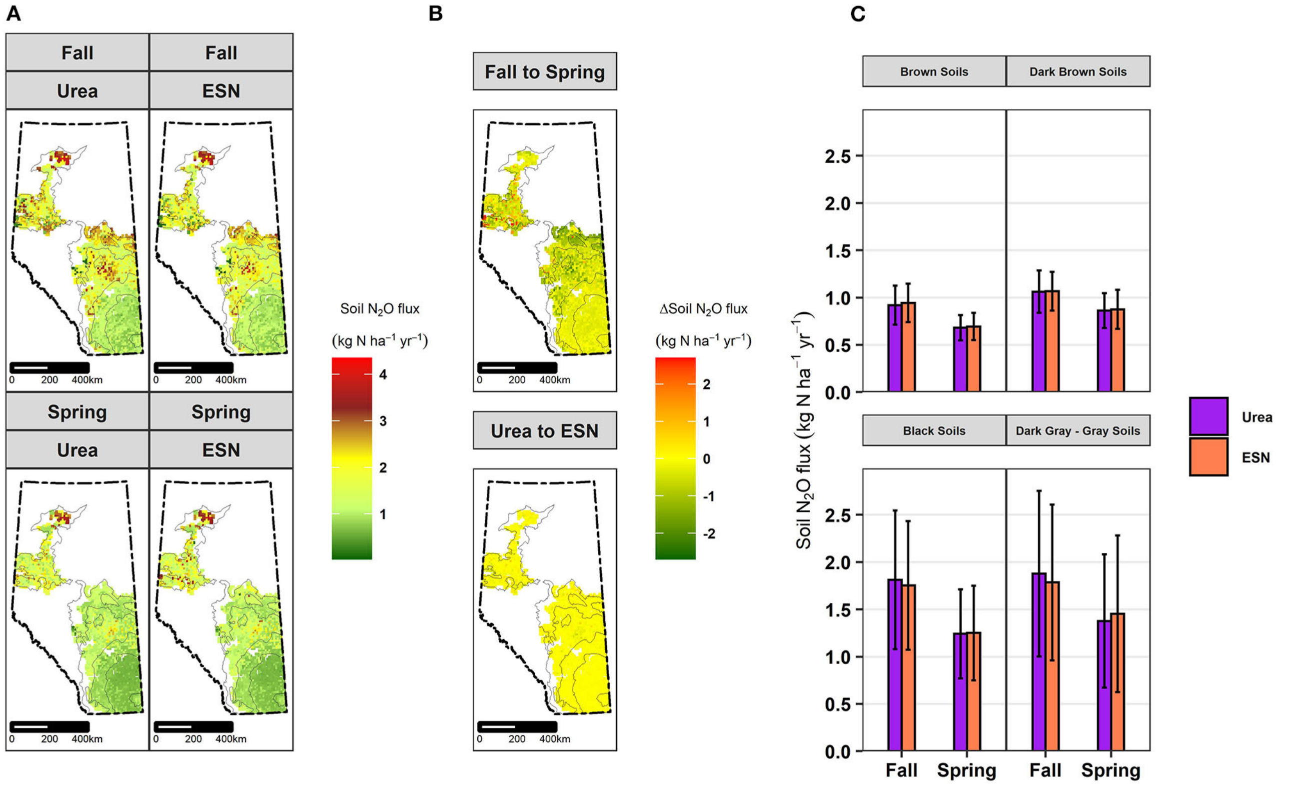


FIGURE 9 | (A) Spatial distribution of modeled annual (January 1–December 31) soil N_2O fluxes averaged over the simulation period (2011–2015) for each township for each of the fall and spring banded urea and ESN applications across mid-slope landforms of Alberta under simulated barley cultivation. Positive fluxes represent emissions and vice versa. **(B)** Spatial distribution of differences in modeled annual soil N_2O fluxes for fall vs. spring and urea vs. ESN applications averaged over the simulation period (2011–2015) for each township across mid-slope landforms of Alberta under simulated barley cultivation. A positive change represents an increase in N_2O fluxes for a change from fall to spring or from urea to ESN and vice versa. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary, and the thin black lines within the maps demarcate soil group areas. **(C)** Soil group-wise averages of modeled annual soil N_2O fluxes for fall vs. spring banding and urea vs. ESN applications averaged over the simulation period (2011–2015) for all the townships in each soil group area across mid-slope landforms of Alberta under simulated barley cultivation. Error bars represent standard deviations illustrating spatiotemporal variations in modeled annual soil N_2O fluxes within each soil group area.

Di-nitrogen (N_2) emission is usually overlooked but can be a very significant form of N loss from agro-ecosystems that receive large N inputs and are under prolonged saturation (Zistl-Schlingmann et al., 2019). Modeled N_2-N emissions were simulated predominantly from complete denitrification under saturated soil conditions during spring thaw, which eventually contributed to greater modeled N_2-N losses in fall than spring banded urea and ESN (**Figure 10**; **Table 2**). Delayed release in N from ESN than urea did not produce a lower N_2 emission (**Figures 5**, **10**; **Table 2**). Modeled N_2-N was not large enough to be agronomically and economically significant (**Figure 10**; **Table 2**).

Modeled NH_3 emissions were higher in fall vs. spring banded urea and ESN in brown, dark brown, and black soils across Alberta (**Figure 11**; **Table 2**). The higher NH_3 volatilization in fall was simulated from higher availability of NH_{4+} for volatilization in fall banding simulations resulting from over winter and early spring hydrolyses. Generally moist soils during and shortly after spring application increased NH_{4+} solubility in the model and, hence, also reduced modeled NH_3 volatilization from spring banding than in fall banding. Reduced rate of urea hydrolysis in ESN caused about 5–8% reduction in NH_3 volatilization compared to urea across dark brown and black soils in Alberta (**Figure 11**; **Table 2**) (Rawluk et al., 2001). Overall, modeled NH_3 emissions across southern and central Alberta were lower than the estimate of about 2.5 kg NH_3-N ha^{-1} $year^{-1}$ for western Canadian wheat fields in an emission factor based monthly NH_3 emission modeling (Sheppard et al., 2010). Simulated NH_3 emissions were also affected by soil pH. For instance, black and dark brown soils that had pH between 7 and 8 shifted the modeled chemical equilibrium in such a way that NH_{4+} solubility was reduced, which ultimately enhanced NH_3 volatilization (**Figures 2**, **11**) (Sommer and Ersbøll, 1996; Bouwman et al., 2002; Havlin et al., 2013; Grant et al., 2016). In contrast, most of the dark gray–gray soils had pH under 6, which enhanced modeled NH_{4+} solubility and hence caused net NH_3 consumption instead of emission (**Figures 2**, **11**) (Sommer and Ersbøll, 1996; Bouwman et al., 2002; Havlin et al., 2013; Grant et al., 2016).

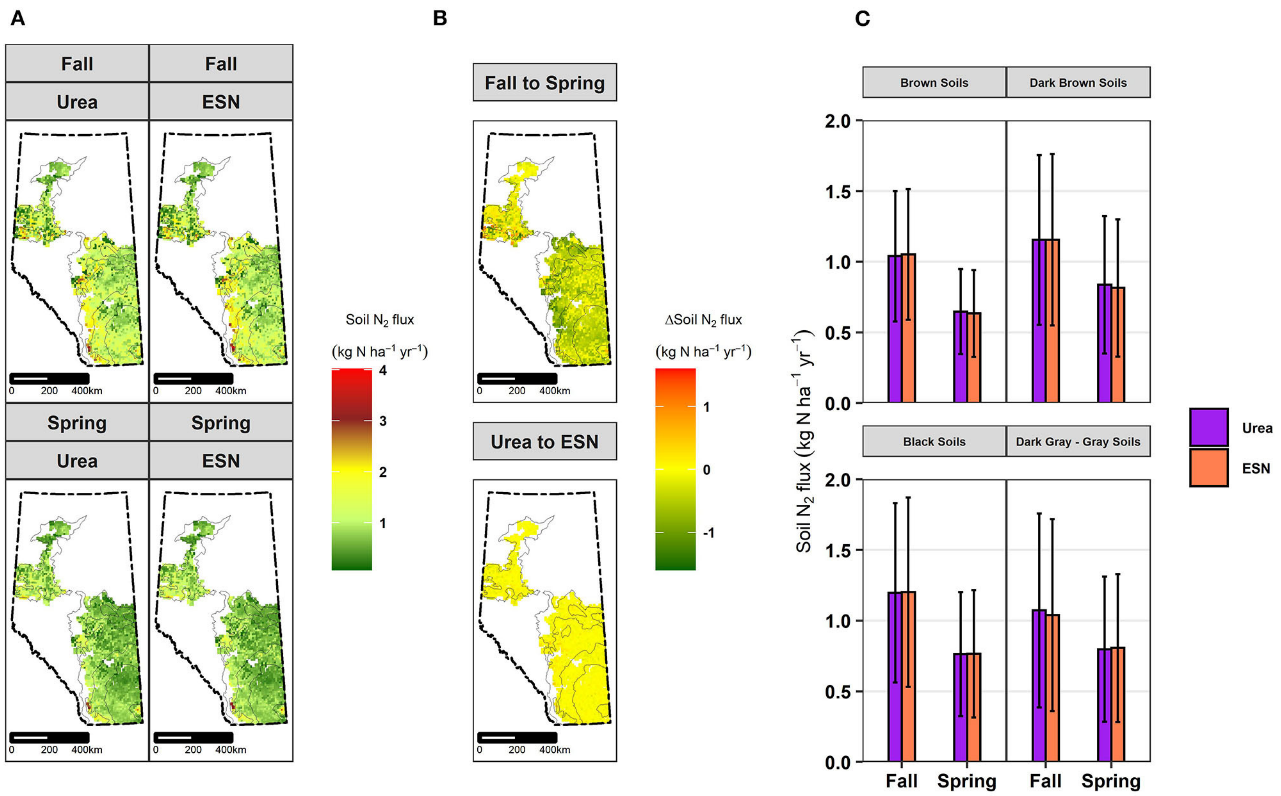


FIGURE 10 | (A) Spatial distribution of modeled annual (January 1–December 31) soil N_2 fluxes averaged over the simulation period (2011–2015) for each township for each of the fall and spring banded urea and ESN applications across mid-slope landforms of Alberta under simulated barley cultivation. Positive fluxes represent emissions and vice versa. **(B)** Spatial distribution of differences in modeled annual soil N_2 fluxes for fall vs. spring and urea vs. ESN applications averaged over the simulation period (2011–2015) for each township across mid-slope landforms of Alberta under simulated barley cultivation. A positive change represents an increase in N_2 fluxes for a change from fall to spring or from urea to ESN and vice versa. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary and the thin black lines within the maps demarcate soil group areas. **(C)** Soil group-wise averages of modeled annual soil N_2 fluxes for fall vs. spring banding and urea vs. ESN applications averaged over the simulation period (2011–2015) for all the townships in each soil group area across mid-slope landforms of Alberta under simulated barley cultivation. Error bars represent standard deviations illustrating spatiotemporal variations in modeled annual soil N_2 fluxes within each soil group area.

Nitrogen losses in surface runoff and sub-surface discharge from crop fields in western Canadian prairies are primarily snowmelt driven (Casson et al., 2008; Tiessen et al., 2010). Dissolved inorganic N from fall banded urea and ESN were transported along with the surface runoff and sub-surface discharge waters during snowmelt and spring thaw, which produced larger modeled N-runoff in fall vs. spring banding across Alberta over the simulation period (**Figure 12**; **Table 2**). Less runoff, combined with lower NO_3^- accumulation from lower N inputs and less N mineralization, caused lower modeled N-runoff fluxes in brown soils than the other soils (**Figures 2, 12**) (Casson et al., 2008). Modeled N-runoff fluxes were overall smaller compared to the estimates of 7.2–11.7 kg N-runoff losses ha^{-1} $year^{-1}$ from eastern Canadian crop fields that received higher precipitation and N inputs (De Jong et al., 2009). However, in western Canada, N loss through NO_3^- leaching was estimated to vary from 1.5 to 4.5 kg N ha^{-1} $year^{-1}$ for various N application rates in various rotations in long-term (over 30 years) research plots (Campbell et al., 1994, 2006). These leaching losses in western Canada were negligible in continuously cropped and fall seeded fields but were very large in excessively fertilized fields and fields under summer fallows, which favored NO_3^- accumulation and soil moisture buildup that are precursors of NO_3^- leaching (Campbell et al., 1994, 2006). Modeled NO_3^- losses through subsurface drainage in this study could be a proxy of NO_3^- leaching since these NO_3^- were transported out of the modeled root zone along with subsurface drainage through lateral model boundaries. Assuming modeled NO_3^- loss through sub-surface drainage as a proxy of N leaching, modeled losses of dissolved organic and inorganic N from continuously cropped fields through surface runoff and subsurface discharge in this study can be considered very conservative when compared to the aqueous N loss estimates through leaching from long-term field studies in western Canada (**Figure 12**) (Campbell et al., 1994, 2006).

Although the N loss estimates in this modeling are within the range of most field-plot based estimates, the modeled estimates can be considered conservative since the simulation

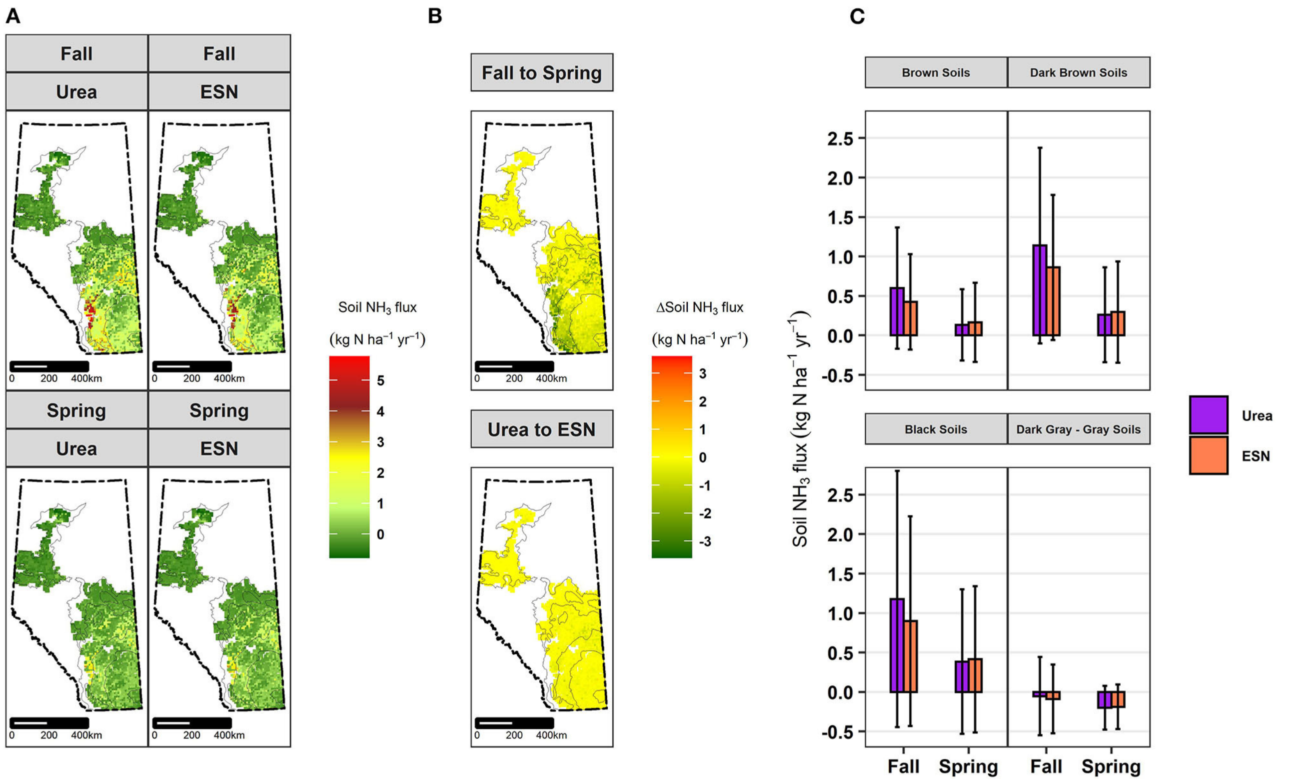


FIGURE 11 | (A) Spatial distribution of modeled annual (January 1–December 31) soil NH_3 fluxes averaged over the simulation period (2011–2015) for each township for each of the fall and spring banded urea and ESN applications across mid-slope landforms of Alberta under simulated barley cultivation. Positive fluxes represent emissions and vice versa. **(B)** Spatial distribution of differences in modeled annual soil NH_3 fluxes for fall vs. spring and urea vs. ESN applications averaged over the simulation period (2011–2015) for each township across mid-slope landforms of Alberta under simulated barley cultivation. A positive change represents an increase in NH_3 fluxes for a change from fall to spring or from urea to ESN and vice versa. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary and the thin black lines within the maps demarcate soil group areas. **(C)** Soil group-wise averages of modeled annual soil NH_3 fluxes for fall vs. spring banding and urea vs. ESN applications averaged over the simulation period (2011–2015) for all the townships in each soil group area across mid-slope landforms of Alberta under simulated barley cultivation. Error bars represent standard deviations illustrating spatiotemporal variations in modeled annual soil NH_3 fluxes within each soil group area.

did not include foot-slope or depressional areas where N losses can be 2- to 3-fold higher than those from mid-slope landforms (Izaurralde et al., 2004). Extending these simulations to remaining landforms in interconnected transects of top-, mid-, and foot-slopes, and depressional areas would thus provide more comprehensive estimates of agronomic N losses and crop N use from various N management scenarios. Reproducing these simulations for highly fertilized crop (e.g., canola) fields, irrigated lands, and for agronomic management such as pulses in rotations, and residue retention, would also include simulations of extreme N losses and, hence, would provide better approximations of the fate of N from various agronomic N management. Besides, the simulations in this study were performed over large spatial extents of each grid cell (∼10 km × 10 km) using only four selected N management scenarios, extrapolated weather data, and dominant soil properties. Given the variabilities in soils, weather, crop, land use, and management practices within each grid cell of this size, the soil, weather, and N management practices used as inputs for these simulations may not always adequately represent the conditions in a field or a farm within a grid. So, the estimates from this modeling study can only be a first approximation of crop N recovery and agronomic N losses at a field or a farm scale. However, finer resolution modeling can be performed at a field or farm scale by selecting locations of interests from the modeled landscape and providing inputs to the model for soil, weather, crop, and management practices to adequately represent a field or a farm within the simulations (**Table 1**).

CONCLUSIONS

The process-based modeling analyses in this study indicated that the spring application of fertilizer could be an optimal N fertilizer application timing for Alberta farmers in reducing N losses while not compromising agronomic and economic returns in dryland barley cultivation across mid-slope landforms. Effectiveness of the spring banding in optimizing benefits and minimizing N losses, however, would be dependent on variations in soils,

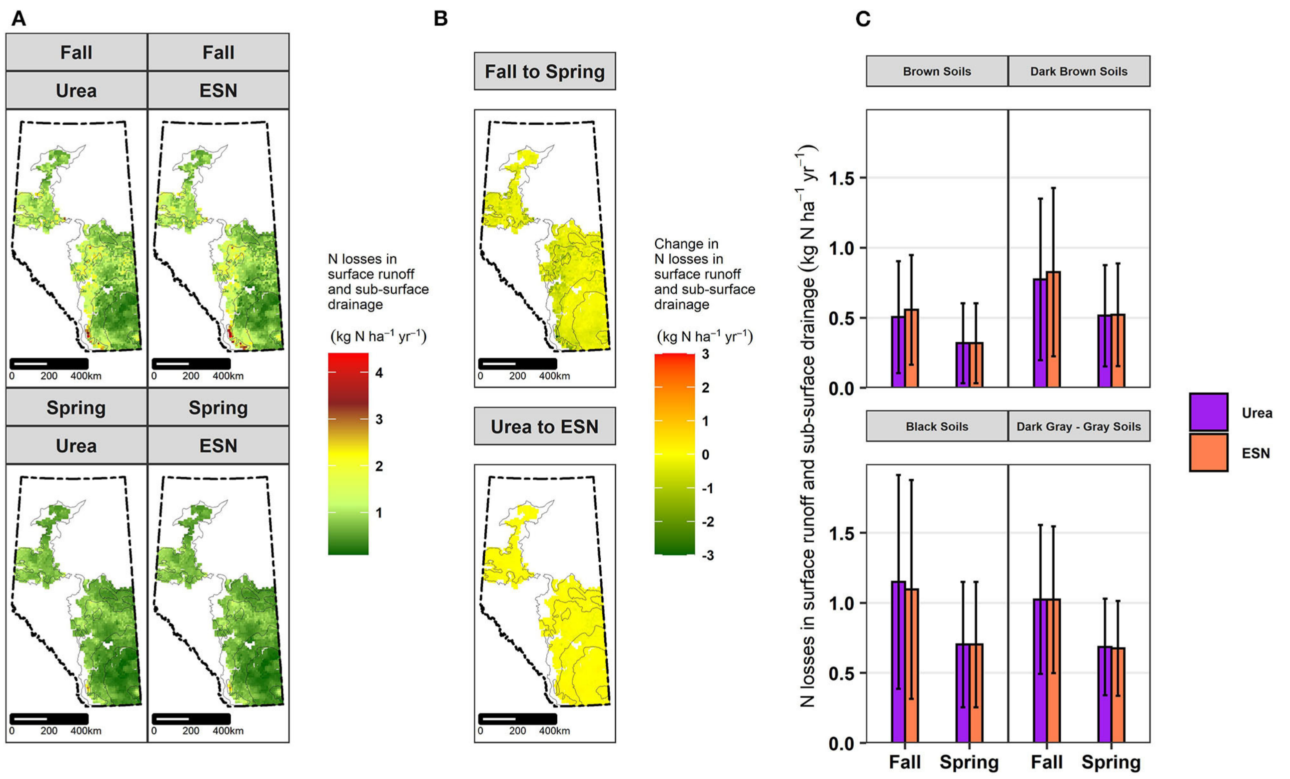


FIGURE 12 | (A) Spatial distribution of modeled annual (January 1–December 31) nitrogen losses in surface runoff and sub-surface drainage averaged over the simulation period (2011–2015) for each township for each of the fall and spring banded urea and ESN applications across mid-slope landforms of Alberta under simulated barley cultivation. **(B)** Spatial distribution of differences in modeled annual nitrogen losses in surface runoff and sub-surface drainage for fall vs. spring and urea vs. ESN applications averaged over the simulation period (2011–2015) for each township across mid-slope landforms of Alberta under simulated barley cultivation. A positive change represents an increase in nitrogen losses in surface runoff and sub-surface drainage for a change from fall to spring or from urea to ESN and vice versa. The white areas on the map were not modeled based on the selection criteria as described in section Methodology and Model Inputs. The thick dashed lines on the maps represent Alberta's provincial boundary, and the thin black lines within the maps demarcate soil group areas. **(C)** Soil group-wise averages of modeled annual nitrogen losses in surface runoff and sub-surface drainage for fall vs. spring banding and urea vs. ESN applications averaged over the simulation period (2011–2015) for all the townships in each soil group area across mid-slope landforms of Alberta under simulated barley cultivation. Error bars represent standard deviations illustrating spatiotemporal variations in modeled annual nitrogen losses in surface runoff and sub-surface drainage within each soil group area.

climate, and rates of N inputs. The modeled results, however, did not show discernible differences in barley N use or agronomic N losses from Alberta barley fields on the mid-slope landscapes due to a difference in N fertilizer products between urea and ESN.

Resilience of any agronomic N management option in terms of long-term sustainability and profitability is a key to successful farming operations. The desire to maximize production rather than optimize it may end up with N fertilizer application rates beyond economic profitability or environmental sustainability. This study opens up windows of opportunities for assessing the potential impacts of increasing N fertilizer application rates on agronomic N loss and crop yields and N uptake in Alberta. Such a study can also be used in assessing topographic influence on variable N fertilizer rates in precision farming to optimize crop productivity and minimize agricultural N losses. Application timing and placement, which are considered less efficient and more prone to losses, such as fall application and surface broadcast, are sometimes preferred for operational reasons. "What if" scenario analyses based on such modeling would provide the farmers with options of working with different enhanced efficiency N fertilizer products such as coated urea, urease inhibitors, nitrification inhibitors, or any combination of these technologies to reduce N losses and optimize production while operating within their operational limitations in terms of timing and placement.

This modeling approach can be used to identify "hot spots" or sensitive areas that are more prone to N losses. Policy makers can use this information to formulate applicable and sustainable policies and to devise incentive plans for promoting environmental stewardship in farming operations. The fertilizer industry can get valuable first-hand forecast to formulate and commercialize suitable products for profitable and sustainable agri-business. This study pioneered a methodology to assess the suitability of 4R nutrient stewardship options for sustainable crop production across a broad area of about 21 million ha. The simulations are also scalable to regional, federal, continental, and global scales by feeding the model with soil, climate, and management data appropriate to the scales (**Table 1**). Therefore, this study has important practical application, replicability, and validity in contributing to the existing knowledge pool of agricultural nutrient management science.

DATA AVAILABILITY STATEMENT

The original sources and links of all data used as model inputs and for model validation are appropriately cited in the article.

CODE AVAILABILITY

Ecosys codes can be downloaded from https://github.com/jinyun1tang/ECOSYS.

AUTHOR CONTRIBUTIONS

SM led the study and contributed to the formulation of the original idea, extracting data, designing modeling experiment, performing simulations, analyzing data, and writing the manuscript. DS, DH, LK, and DI contributed to the design and implementation of data extraction procedure for the modeling experiment and to manuscript writing. TG and JI contributed to the design of modeled output visualizations and manuscript writing. RG is the original developer of the model *ecosys* used in this study and also contributed to the modeling experiment and manuscript writing. All authors contributed to the article and approved the submitted version.

FUNDING

Funding for this project was provided by Growing Forward 2 Research and Innovation Internal Initiatives of federal–provincial partnership between Canada and Alberta. Computing support for this project was provided by Compute Canada WestGrid. Organization and analyses of model inputs and modeled outputs were performed by using R 3.6.3 packages of dplyr and tidyr. All the graphs and the maps (except **Figure 1**) were created by using R 3.6.3 packages of ggplot2 and rgdal.

ACKNOWLEDGMENTS

We thank Deb Werk, Fatima Shire, and Rachel Berardi for their inputs in data extraction for model inputs and modeled output visualization. We also thank Karen Raven for advocating and supporting the project throughout the administrative processes of funding competition.

REFERENCES

ACIS (2019). *Data From: Alberta Climate Information Service*. Available online at: https://agriculture.alberta.ca/acis/

AFSC (2019). *Data From: Yield Alberta 2015 and 2016 compiled by Agriculture Financial Services Corporation (AFSC) Alberta*. Available online at: https://afsc.ca/resources/

AGRASID (2019). *Data From: Agricultural Regions of Alberta Soil Inventory Database.* Available online at: https://www.alberta.ca/agricultural-regions-of-alberta-soil-inventory-database.aspx

Anbessa, Y., and Juskiw, P. (2012). Nitrogen fertilizer rate and cultivar interaction effects on nitrogen recovery, utilization efficiency, and agronomic performance of spring barley. *ISRN Agronomy* 2012:531647. doi: 10.5402/2012/531647

Asgedom, H., Tenuta, M., Flaten, D. N., Gao, X., and Kebreab, E. (2014). Nitrous oxide emissions from a clay soil receiving granular urea formulations and dairy manure. *Agron. J.* 106, 732–744. doi: 10.2134/agronj2013.0096

Bhanja, S. N., Wang, J., Shrestha, N. K., and Zhang, X. (2019). Microbial kinetics and thermodynamic (MKT) processes for soil organic matter decomposition and dynamic oxidation-reduction potential: model descriptions and applications to soil N_2O emissions. *Environ. Pollut.* 247, 812–823. doi: 10.1016/j.envpol.2019.01.062

Bock, M., Gasser, P. Y., Pettapiece, W. W., Brierley, A. J., Bootsma, A., Schut, P., et al. (2018). The land suitability rating system is a spatial planning tool to assess crop suitability in Canada. *Front. Environ. Sci.* 6:77. doi: 10.3389/fenvs.2018.00077

Bouwman, A. F., Boumans, L. J. M., and Batjes, N. H. (2002). Estimation of global NH_3 volatilization loss from synthetic fertilizers and animal manure applied to arable lands and grasslands. *Glob. Biogeochem. Cy.* 16:8. doi: 10.1029/2000GB001389

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., and Zechmeister-Boltenstern, S. (2013). Nitrous oxide emissions from soils: how well do we understand the processes and their controls?. *Philos. T. R. Soc. B* 368:20130122. doi: 10.1098/rstb.2013.0122

Cahill, S., Osmond, D., and Israel, D. (2010). Nitrogen release from coated urea fertilizers in different soils. *Commun. Soil Sci. Plan.* 41, 1245–1256. doi: 10.1080/00103621003721437

Campbell, C. A., Lafond, G. P., Zentner, R. P., and Jame, Y. W. (1994). Nitrate leaching in a udic haploboroll as influenced by fertilization and legumes. *J. Environ. Qual.* 23, 195–201. doi: 10.2134/jeq1994.00472425002300010030x

Campbell, C. A., Selles, F., Zentner, R. P., De Jong, R., Lemke, R., and Hamel, C. (2006). Nitrate leaching in the semiarid prairie: effect of cropping frequency, crop type, and fertilizer after 37 years. *Can. J. Soil Sci.* 86, 701–710. doi: 10.4141/S05-008

Cassman, K. G., Dobermann, A., and Walters, D. T. (2002). Agroecosystems, nitrogen-use efficiency, and nitrogen management. *AMBIO* 31, 132–141. doi: 10.1579/0044-7447-31.2.132

Casson, J. P., Olson, B. M., Little, J. L., and Nolan, S. C. (2008). *Assessment of Environmental Sustainability in Alberta's Agricultural Watersheds Project, Volume 4: Nitrogen Loss in Surface Runoff*. Edmonton, AB: Alberta Agriculture and Rural Development.

Chai, L. L. (2017). *Nitrous oxide emissions in southern alberta croplands in response to nitrogen rates, fertigation and moisture* (Master's thesis). University of Alberta, Edmonton, AB.

Cui, G., and Wang, J. (2019). Improving the DNDC biogeochemistry model to simulate soil temperature and emissions of nitrous oxide and carbon dioxide in cold regions. *Sci. Total Environ.* 687, 61–70. doi: 10.1016/j.scitotenv.2019.06.054

De Jong, R., Drury, C. F., Yang, J. Y., and Campbell, C. A. (2009). Risk of water contamination by nitrogen in Canada as estimated by the IROWC-N model. *J. Environ. Manage.* 90, 3169–3181. doi: 10.1016/j.jenvman.2009.05.034

Dimitrov, D. D., and Wang, J. (2019). Geographic inventory framework for estimating spatial pattern of methane and nitrous oxide emissions from agriculture in Alberta, Canada. *Environ. Dev.* 32:100461. doi: 10.1016/j.envdev.2019.100461

Dunmola, A. S., Tenuta, M., Moulin, A. P., Yapa, P., and Lobb, D. A. (2010). Pattern of greenhouse gas emission from a Prairie Pothole agricultural landscape in Manitoba, Canada. *Can. J. Soil Sci.* 90, 243–256. doi: 10.4141/CJSS08053

Flesch, T., Baron, V., Wilson, J., Basarab, J., Desjardins, R., Worth, D., et al. (2018). Micrometeorological measurements reveal large nitrous oxide losses during spring thaw in Alberta. *Atmosphere* 9:128. doi: 10.3390/atmos9040128

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D. W., et al. (2007). "Changes in atmospheric constituents and in radiative forcing," in *Climate Change 2007: The Physical Science Basis*, eds S. D. Solomon, M. Qin,

Z. Manning, M. Chen, K. B. Marquis, and M. T. Averyt (Cambridge; New York, NY: Cambridge University Press), 129–234.

Gao, X., Asgedom, H., Tenuta, M., and Flaten, D. N. (2015). Enhanced efficiency urea sources and placement effects on nitrous oxide emissions. *Agron. J.* 107, 265–277. doi: 10.2134/agronj14.0213

Glenn, A. J., Tenuta, M., Amiro, B. D., Maas, S. E., and Wagner-Riddle, C. (2012). Nitrous oxide emissions from an annual crop rotation on poorly drained soil on the Canadian prairies. *Agr. Forest Meteorol.* 166, 41–49. doi: 10.1016/j.agrformet.2012.06.015

Grant, C., and Wu, R. (2008). Enhanced-efficiency fertilizers for use on the Canadian prairies. *Crop Management* 7, 1–15. doi: 10.1094/CM-2008-0730-01-RV

Grant, C. A., Peterson, G. A., and Campbell, C. A. (2002). Nutrient considerations for diversified cropping systems in the northern great plains. *Agron. J.* 94, 186–198. doi: 10.2134/agronj2002.0186

Grant, R. F. (2001). "A review of the Canadian ecosystem model ecosys," in *Modeling Carbon and Nitrogen Dynamics for Soil Management*, eds. M. J. Shaffer, L. Ma, and S. Hansen (Boca Raton, FL: CRC Press), 173–264.

Grant, R. F., Dyck, M., and Puurveen, D. (2020). Nitrogen and phosphorus control carbon sequestration in agricultural ecosystems: modelling carbon, nitrogen, and phosphorus balances at the breton plots with ecosys under historical and future climates. *Can. J. Soil Sci.* doi: 10.1139/cjss-2019-0132

Grant, R. F., Kimball, B. A., Conley, M. M., White, J. W., Wall, G. W., and Ottman, M. J. (2011). Controlled warming effects on wheat growth and yield: field measurements and modeling. *Agron. J.* 103, 1742–1754. doi: 10.2134/agronj2011.0158

Grant, R. F., Neftel, A., and Calanca, P. (2016). Ecological controls on N_2O emission in surface litter and near-surface soil of a managed grassland: modelling and measurements. *Biogeosciences* 13, 3549–3571. doi: 10.5194/bg-13-3549-2016

Grant, R. F., and Pattey, E. (1999). Mathematical modeling of nitrous oxide emissions from an agricultural field during spring thaw. *Glob. Biogeochem. Cy.* 13, 679–694. doi: 10.1029/1998GB900018

Grant, R. F., and Pattey, E. (2003). Modelling variability in N_2O emissions from fertilized agricultural fields. *Soil Biol. Biochem.* 35, 225–243. doi: 10.1016/S0038-0717(02)00256-0

Grant, R. F., and Pattey, E. (2008). Temperature sensitivity of N_2O emissions from fertilized agricultural soils: mathematical modeling in ecosys. *Glob. Biogeochem. Cy.* 22:GB4019. doi: 10.1029/2008GB003273

Grant, R. F., Pattey, E., Goddard, T. W., Kryzanowski, L. M., and Puurveen, H. (2006). Modeling the effects of fertilizer application rate on nitrous oxide emissions. *Soil Sci. Soc. Am. J.* 70, 235–248. doi: 10.2136/sssaj2005.0104

Hao, X., Chang, C., Carefoot, J. M., Janzen, H. H., and Ellert, B. H. (2001). Nitrous oxide emissions from an irrigated soil as affected by fertilizer and straw management. *Nutr. Cycl. Agroecosys.* 60, 1–8. doi: 10.1023/A:1012603732435

Havlin, J. L., Tisdale, S. L., Nelson, W. L., and Beaton, J. D. (2013). *Soil Fertility and Fertilizers: An Introduction to Nutrient Management*. Hoboken, NJ: Pearson Inc.

Izaurralde, R. C., Lemke, R. L., Goddard, T. W., McConkey, B., and Zhang, Z. (2004). Nitrous oxide emissions from agricultural toposequences in Alberta and Saskatchewan. *Soil Sci. Soc. Am. J.* 68, 1285–1294. doi: 10.2136/sssaj2004.1285

Janzen, H. H., Beauchemin, K. A., Bruinsma, Y., Campbell, C. A., Desjardins, R. L., Ellert, B. H., et al. (2003). The fate of nitrogen in agroecosystems: an illustration using Canadian estimates. *Nutr. Cycl. Agroecosys.* 67, 85–102. doi: 10.1023/A:1025195826663

Lemke, R. L., Izaurralde, R. C., and Nyborg, M. (1998). Seasonal distribution of nitrous oxide emissions from soils in the parkland region. *Soil Sci. Soc. Am. J.* 62, 1320–1326. doi: 10.2136/sssaj1998.03615995006200050025x

Lemke, R. L., Izaurralde, R. C., Nyborg, M., and Solberg, E. D. (1999). Tillage and N source influence soil-emitted nitrous oxide in the Alberta Parkland region. *Can. J. Soil Sci.* 79, 15–24. doi: 10.4141/S98-013

Li, C., Hao, X., Blackshaw, R. E., Clayton, G. W., O'Donovan, J. T., and Harker, K. N. (2016). Nitrous oxide emissions in response to ESN and urea application in a no-till barley cropping system. *Commun. Soil Sci. Plan.* 47, 692–705. doi: 10.1080/00103624.2016.1146745

Li, C., Hao, X., Blackshaw, R. E., O'Donovan, J. T., Harker, K. N., and Clayton, G. W. (2012). Nitrous oxide emissions in response to ESN and urea, herbicide management and canola cultivar in a no-till cropping system. *Soil Tillage Res.* 118, 97–106. doi: 10.1016/j.still.2011.10.017

Li, Y., Shah, S. H. H., and Wang, J. (2020). Modelling of nitrification inhibitor and its effects on emissions of nitrous oxide (N_2O) in the UK. *Sci. Total Environ.* 709:136156. doi: 10.1016/j.scitotenv.2019.136156

LSRS (2019). *Data From: Land Suitability Rating System*. LSRS. Available online at: https://open.alberta.ca/publications/land-suitability-rating-system

Maas, S. E., Glenn, A. J., Tenuta, M., and Amiro, B. D. (2013). Net CO_2 and N_2O exchange during perennial forage establishment in an annual crop rotation in the red river valley, manitoba. *Can. J. Soil Sci.* 93, 639–652. doi: 10.4141/cjss2013-025

MacArthur Foundation (2019). *Financing Sustainable Development Goals*. Available online at: https://www.macfound.org/press/grantee-publications/financing-sustainable-development-goals/ (accessed November 13, 2019).

MacMillan, R. A., and Pettapiece, W. W. (2000). *Alberta Landforms, Quantitative Morphometric Descriptions and Classification of Typical Alberta Landforms*. Saskatchewan: Agriculture and Agri-Food Canada.

Malhi, S. S., Grant, C. A., Johnston, A. M., and Gill, K. S. (2001). Nitrogen fertilization management for no-till cereal production in the canadian great plains: a review. *Soil Tillage Res.* 60, 101–122. doi: 10.1016/S0167-1987(01)00176-3

Malhi, S. S., Nyborg, M., Solberg, E. D., and Heany, D. J. (1992). *Fall Compared to Spring Application of Nitrogen Fertilizers in Alberta*. Edmonton, AB: Agriculture Canada.

McKenzie, R. H., Middleton, A. B., Hall, L., DeMulder, J., and Bremer, E. (2004). Fertilizer response of barley grain in south and central Alberta. *Can. J. Soil Sci.* 84, 513–523. doi: 10.4141/S04-013

Metivier, K. A., Pattey, E., and Grant, R. F. (2009). Using the ecosys mathematical model to simulate temporal variability of nitrous oxide emissions from a fertilized agricultural soil. *Soil Biol. Biochem.* 41, 2370–2386. doi: 10.1016/j.soilbio.2009.03.007

Nyborg, M., Laidlaw, J. W., Solberg, E. D., and Malhi, S. S. (1997). Denitrification and nitrous oxide emissions from a black Chernozemic soil during spring thaw in Alberta. *Can. J. Soil Sci.* 77, 153–160. doi: 10.4141/S96-105

Perrott, L. A. (2016). *Advanced agronomic practices to maximize feed barley (Hordeum vulgare L.) yield, quality, and standability in Alberta environments* (Master's thesis). University of Alberta, Edmonton, AB.

Protz, R., Presant, E. W., and Arnold, R. W. (1968). Establishment of the modal profile and measurement of variability within a soil landform unit. *Can. J. Soil Sci.* 48, 7–19. doi: 10.4141/cjss68-002

Qiao, C., Liu, L., Hu, S., Compton, J. E., Greaver, T. L., and Li, Q. (2015). How inhibiting nitrification affects nitrogen cycle and reduces environmental impacts of anthropogenic nitrogen input. *Glob. Change Biol.* 21, 1249–1257. doi: 10.1111/gcb.12802

Rawluk, C. D. L., Grant, C. A., and Racz, G. J. (2001). Ammonia volatilization from soils fertilized with urea and varying rates of urease inhibitor NBPT. *Can. J. Soil Sci.* 81, 239–246. doi: 10.4141/S00-052

Rochette, P., Worth, D. E., Lemke, R. L., McConkey, B. G., Pennock, D. J., Wagner-Riddle, C., et al. (2008). Estimation of N_2O emissions from agricultural soils in Canada. I. Development of a country-specific methodology. *Can. J. Soil Sci.* 88, 641–654. doi: 10.4141/CJSS07025

Rockström, J., Steffen, W. L., Noone, K., Persson, Å., Chapin, F. S. III, Lambin, E. F., et al. (2009). Planetary boundaries: exploring the safe operating space for humanity. *Ecol. Soc.* 14:32. doi: 10.5751/ES-03180-140232

Shcherbak, I., Millar, N., and Robertson, G. P. (2014). Global metaanalysis of the nonlinear response of soil nitrous oxide (N_2O) emissions to fertilizer nitrogen. *Proc. Natl. Acad. Sci. U.S.A.* 111, 9199–9204. doi: 10.1073/pnas.1322434111

Shen, J., Melaku, N. D., Treu, R., and Wang, J. (2019a). Inventories of methane and nitrous oxide emissions from animal and crop farms of 69 municipalities in Alberta, Canada. *J. Clean. Prod.* 234, 895–911. doi: 10.1016/j.jclepro.2019.06.270

Shen, J., Treu, R., Wang, J., Hao, X., and Thomas, B. W. (2019b). Modeling growing season and annual cumulative nitrous oxide emissions and emission factors from organically fertilized soils planted with barley in Lethbridge, Alberta, Canada. *Agric. Syst.* 176:102654. doi: 10.1016/j.agsy.2019.102654

Sheppard, S. C., Bittman, S., and Bruulsema, T. W. (2010). Monthly ammonia emissions from fertilizers in 12 canadian ecoregions. *Can. J. Soil Sci.* 90, 113–127. doi: 10.4141/CJSS09006

Shrestha, N. K., Thomas, B. W., Du, X., Hao, X., and Wang, J. (2018). Modeling nitrous oxide emissions from rough fescue grassland soils subjected to long-term grazing of different intensities using the soil and water assessment tool (SWAT). *Environ. Sci. Pollut. Res. Int.* 25, 27362–27377. doi: 10.1007/s11356-018-2719-2

Snyder, C. S. (2017). Enhanced nitrogen fertiliser technologies support the '4R'concept to optimise crop production and minimise environmental losses. *Soil Res.* 55, 463–472. doi: 10.1071/SR16335

Sommer, S. G., and Ersbøll, A. K. (1996). Effect of air flow rate, lime amendments, and chemical soil properties on the volatilization of ammonia from fertilizers applied to sandy soils. *Biol. Fertil. Soils* 21, 53–60. doi: 10.1007/BF00335993

Soon, Y. K., Malhi, S. S., Lemke, R. L., Lupwayi, N. Z., and Grant, C. A. (2011). Effect of polymer-coated urea and tillage on the dynamics of available N and nitrous oxide emission from gray luvisols. *Nutr. Cycl. Agroecosys.* 90, 267–279. doi: 10.1007/s10705-011-9428-2

Statistics Canada (2016). *Data from: 2016 Census of Agriculture.* Available online at: https://www.statcan.gc.ca/eng/ca2016

Tiessen, K. H. D., Elliott, J. A., Yarotski, J., Lobb, D. A., Flaten, D. N., and Glozier, N. E. (2010). Conventional and conservation tillage: influence on seasonal runoff, sediment, and nutrient losses in the canadian prairies. *J. Environ. Qual.* 39, 964–980. doi: 10.2134/jeq2009.0219

UNEP (2014). *UNEP Year Book 2014: Emerging Issues in Our Global Environment*. United Nations Environment Programme: Division of Early Warning, and Assessment. Available online at: https://agriculture.alberta.ca/acis/

Urso, J. H., and Gilbertson, L. M. (2018). Atom conversion efficiency: a new sustainability metric applied to nitrogen and phosphorus use in agriculture. *ACS Sustain. Chem. Eng.* 6, 4453–4463. doi: 10.1021/acssuschemeng.7b03600

Zistl-Schlingmann, M., Feng, J., Kiese, R., Stephan, R., Zuazo, P., Willibald, G., et al. (2019). Dinitrogen emissions: an overlooked key component of the N balance of montane grasslands. *Biogeochemistry* 143, 15–30. doi: 10.1007/s10533-019-00547-8

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 29 September 2020
doi: 10.3389/fsufs.2020.497943

Iowa Farm Environmental Leaders' Perspectives on the U.S. Farm Bill Conservation Programs

Gabriel Medina[1*†], *Catherine Isley*[2†] and *J. Arbuckle*[3†]

[1] Department of Agronomy, University of Brasilia, Brasília, Brazil, [2] Division of Applied Social Sciences, University of Missouri, Columbia, MO, United States, [3] Department of Sociology, Iowa State University, Ames, IA, United States

OPEN ACCESS

Edited by:
Alisa W. Coffin, United States Department of Agriculture, United States

Reviewed by:
Christopher Edward Latimer, The Nature Conservancy, United States
Alycia Rae Bean, United States Department of Agriculture, United States

***Correspondence:**
Gabriel Medina
gabriel.medina@unb.br

†ORCID:
Gabriel Medina
orcid.org/0000-0002-5815-6812
Catherine Isley
orcid.org/0000-0003-3431-1395
J. Arbuckle
orcid.org/0000-0001-9419-4624

Specialty section:
This article was submitted to Land, Livelihoods and Food Security, a section of the journal Frontiers in Sustainable Food Systems

***Received:** 13 September 2019*
***Accepted:** 03 September 2020*
***Published:** 29 September 2020*

Citation:
Medina G, Isley C and Arbuckle J (2020) Iowa Farm Environmental Leaders' Perspectives on the U.S. Farm Bill Conservation Programs. Front. Sustain. Food Syst. 4:497943. doi: 10.3389/fsufs.2020.497943

As stakeholders prepare to lobby future Farm Bills, this study reveals farmers' perspectives on federal conservation programs. In-depth interviews were held with ten farm environmental leaders, farmers who have extensive experience with conservation practices and federal conservation programs. Results reveal that conservation programs have played a limited but important role in incentivizing the adoption of and offsetting costs for establishing conservation practices. Programs' strengths, weaknesses, and potential improvements were also explored; results reveal that most farmers believe existing conservation programs could be improved with relatively minor tweaks and adaptations, such as more flexibility in working land program requirements and adjustments to land retirement program payment rates. To some extent, farm environmental leaders also align themselves with the perspectives of environmental NGOs, advocating for transformative approaches, such as expanding mandatory conservation compliance to all cropland, including non-Highly Erodible Lands cropland.

Keywords: conservation stewardship program (CSP), environmental quality incentive program (EQIP), conservation reserve program (CRP), conservation compliance (CC), sustainable farming

INTRODUCTION

Farmers perceive a tradeoff between short-term profit and long-term environmental sustainability (Arbuckle, 2016; Roesch-McNally et al., 2017). To some extent, U.S. farmers can rely on federal agricultural policy, the Farm Bill, to mitigate the short-term costs of adopting conservation-related practices (Reimer and Prokopy, 2014).

The United States Department of Agriculture (USDA) Farm Bill conservation title includes a number of conservation programs. This research focused on several of the largest and most commonly used: the Conservation Stewardship Program (CSP), the Environmental Quality Incentives Program (EQIP), the Conservation Reserve Program (CRP), and Conservation Compliance (ERS, 2020). The CSP is a working lands program that provides annual and cost-share payments to reward existing conservation practices and promotes further improvements by incentivizing incorporation of new conservation practices over time through 5-year contracts (USDA, 2016). Similarly, EQIP is another working lands program that provides conservation practice cost-share, but with an emphasis on livestock production and through shorter-term contracts for specific practices and conservation planning and technical assistance (USDA, 2018). The CRP is a program that establishes 10 and 15-year contracts with farmers and landowners to remove environmentally sensitive lands from agricultural production and install resource-conserving practices (Lambert et al., 2007; USDA, 2019). Conservation Compliance is a program that ties USDA program benefits such as subsidized crop insurance premiums to environmental performance on sensitive lands such as highly erodible lands (HEL) and wetlands, stipulating that benefits can be lost if, for example, wetlands are converted to crop production or agricultural commodities are produced on HEL without an approved conservation plan or exemption (Arbuckle, 2013). Individual states often have additional conservation programs, but this research has focused on the aforementioned Farm Bill programs.

Farm Bill conservation programs are implemented on a voluntary basis and promote specific practices targeting soil health and water and nutrient management (Lambert et al., 2007; Reimer and Prokopy, 2014). Promoted practices include both structural measures, such as buffer strips, and management measures, such as cover crops (Ulrich-Schad et al., 2017; Stuart et al., 2018).

After decades of investment in conservation–related practices (Mcfadden and Hoppe, 2017), progress can be seen, especially when it comes to the reduction of soil erosion rates (USDA, 2015). But many challenges remain, particularly surrounding soil health and water and nutrient management (Rundquist and Cox, 2016). An understanding of how farmers view conservation programs is fundamental for overcoming these challenges.

Recent studies have provided a comprehensive understanding of the adoption of conservation-related practices by farmers. Adoption is often voluntary, but it can also be catalyzed by conservation programs (Medina et al., 2015; Nebel et al., 2017). Factors explaining adoption range from perceived relative advantage to the cost and risk of trying a new practice (Reimer et al., 2012; Arbuckle and Roesch-McNally, 2015; Prokopy et al., 2019; Ranjan et al., 2019).

Studies have also assessed the reach of Farm Bill conservation programs. They have found great variability in program participation across states (Reimer et al., 2013) and types of farming operations (Lambert et al., 2007). Barriers to participation include farmers' lack of knowledge on existing conservation programs and their requirements (Reimer and Prokopy, 2014; Prokopy et al., 2019; Ranjan et al., 2019).

As stakeholders prepare to lobby future Farm Bills, advocacy groups are proposing changes to Farm Bill programs (Medina et al., 2020). While farmer and commodity groups support adaptions to increase flexibility in conservation programs (Farm Bureau, 2017), environmental NGOs advocate for more transformative and often mandatory approaches (EWG, 2017).

Nonetheless, limited effort has been made to understand farmer perspectives on federal conservation programs. Therefore, this study aims to identify farmer perspectives on Farm Bill conservation programs, including the CSP, EQIP, CRP, and Conservation Compliance programs. Specifically, this study aims to understand farmer perspectives on:

- The role these programs play in supporting adoption of conservation practices;
- These programs' strengths, weaknesses, and potential improvements.

THEORETICAL FRAMEWORK

We employ a conceptual framework adapted from Hall (1993) and Atwell et al. (2011) to examine farmers' perspectives on the major U.S. federal conservation programs. Policy changes can be divided into three subtypes according to magnitude (Hall, 1993). First and second order changes are likely to display features of incrementalism and development of new policy instruments, but the changed policy is still within the same paradigm (Hall, 1993). Third order change is associated with a change in paradigm, which is preceded by significant shifts in the locus of authority over policy and experimentation with new forms of policy (Hall, 1993).

Policy changes include the fine-tuning of existing policy instruments (tweak), adaptation of the existing instruments (adapt), and overall transformation of the policy (transform) (Atwell et al., 2011). Ranging from incremental tweaks and adaptations to transformative proposals, farmers and stakeholders can be identified along a continuum of paradigmatic orientation (Arbuckle, 2009).

Policy resilience is supported by a capacity to absorb new ideas and still maintain its essential configuration (Atwell et al., 2011). However, the process of internalizing new ideas may result in changes in the locus of authority over policy from one stakeholder to another and a broadening of the policy network (Hall, 1993). Transition theory suggests that, while at times, coherent phases of societal organization can be identified, at other times chaotic transitional characteristics may dominate, leading eventually to a new set of structured coherences (Cloke and Goodwin, 1992).

Interviewed farmers primarily suggest tweaks and adaptations to current conservation programs. These suggested changes are listed by program in our Results section. Several farmers also shared more transformational ideas, which are discussed after the program-by-program results.

MATERIALS AND METHODS

The study was conducted with farmers in the state of Iowa, which is a major U.S. agricultural producer and has high conservation program payments per capita (Reimer, 2013). Specifically, interviews were held with farmers who had received the Iowa Farm Environmental Leadership (IFEL) Award[1] for incorporating conservation practices into their farming operations. These farmers had extensive experience implementing conservation practices and participating in Farm Bill conservation programs.

This study analyzed data from in-depth personal interviews with a sample of ten farmers, conducted between October and December 2017. The research focused on farmers who had received the IFEL award in 2012 or 2013, the first 2 years

[1]The Iowa Farm Environmental Leader Award is a joint effort of the Governor, Lt. Governor, Iowa Department of Agriculture and Land Stewardship, and Iowa Department of Natural Resources to recognize the exemplary voluntary efforts of Iowa's farmers as environmental leaders committed to healthy soils and improved water quality.

of the program. In 2014, a previous study selected a sample of 20 farmers from the 131 farmers who received the award in the first 2 years of the IFEL (Rosman, 2015). The sample selection process was designed to recruit study participants who were widely recognized as conservation-oriented opinion leaders, even relative to other award winners. Participants were selected based on an Internet search engine query of their names and locations as stated in the public listing of the award recipients. The eligibility criteria for participants was set to be three or more links to media articles featuring their soil and water conservation achievements on the first three pages of search results. Thus, the selected participants were farmers who had been recognized for their stewardship through both the award and multiple instances of recognition in the farm and mainstream media.

Because the 2017 study reported in this paper focuses on farmer perspectives on program participation, we selected a subsample of ten farmers from the 2014 sample whom we knew had substantial experience with conservation programs. Given our research questions and the population of interest (high-level conservation farmers), we feel confident that the sample, both in terms of size and constitution, is appropriate for the scope of the paper. While a sample of ten is on the small side, it can be more than adequate, especially if the research participants are a relatively homogeneous group, as in this case (Guest et al., 2006; Mason, 2010). The sample is biased and not representative of all farmers because it is made up of farmers who are exemplars in terms of their soil and water conservation behaviors and program participation over a long time period. However, we purposely selected them as key informants precisely because they, as exemplars, have unique perspectives that we believe make them ideal participants with whom to engage in semi-structured discussions that evaluate current programs and provide insights into how to shape future conservation policies and programs.

Participants' farm operations had rotations of corn (*Zea mays*) and soybeans (*Glycine max*) that are typical of Iowa agriculture, and several farmers also raised some livestock. Farms size varied ranging from 320 acres to 5,000 acres and averaging 1,962 acres (**Table 1**), compared to the 2017 USDA Census of Agriculture Iowa average of 355 acres (USDA, 2017). Thus, participants were primarily large-scale family farms as defined by the USDA Economic Research Service (Hoppe and MacDonald, 2013). Large-scale family farms participate in conservation programs at a disproportionately high rate compared to small-scale farms (Lambert et al., 2007).

Interviews were held on-site at each farm. Each interview followed the same semi-structured protocol covering the specific objectives outlined in this paper's introduction (see **Supplementary Material**). After introducing the research objective, we systematically asked all interviewed farmers the same questions. The main topics addressed were: 1. the federal conservation programs interviewed farmers had experience with, 2. their perspectives on those programs based on their own experience (each mentioned program was explored based on its perceived strengths and weaknesses), 3. the actual practices adopted in the farm operation either supported by Farm Bill programs, third-party investments or farmers' out-of-pocket money, and 4. a final question asking their opinion about the potential expansion of conservation compliance to all cropland, including non-Highly Erodible Land (HEL) cropland.

TABLE 1 | Profile of interviewed farmers.

Farmer code	Farmed area (in acres)*	Farm business enterprises
1	2,000	60% corn, 40% soybeans
2	3,600	60% corn, 40% soybeans, 4,800 head hog finishing
3	1,000	50% corn, 50% soybeans
4	800	50% corn, 50% soybeans
5	1,200	3 years corn/1 year soybean rotation, farrow-to-finish operation
6	1,000	2-year corn/soybean rotation
7	1,500	50% corn, 50% soybeans
8	5,000	Variable rotations of corn and soybeans, runs a precision agriculture equipment dealership
9	3,200	Variable rotations of corn and soybeans
10	320	Half row-crop, half non-tillable (pasture, CRP), 40 head of cattle

**Includes both owned and rented land.*

Each of these main questions were followed by clarification questions, and on average, interviews lasted for 1.5 h. In some cases, when suggested by farmers, interviews were followed by a visit to farm fields, barns, etc.

All interviews were recorded and transcribed. We analyzed the interview data employing a hierarchical coding procedure to identify themes under each set of questions (Corbin and Strauss, 1990). Preliminary analysis of the interview data consisted of code development based on the interview protocol questions, and coding of transcribed responses to questions about the programs (e.g., CSP strength, EQIP weakness). Transcripts were further coded using the "tweak, adapt, transform" framework to identify emergent themes associated with farmers' assessments of how programs and policies might be improved. The first author was the primary coder and the third author reviewed the coded transcripts to enhance reliability. Finally, we include direct quotes to improve the validity and transparency of the analysis and provide readers with nuanced details of context and meanings within the interview data (Prokopy, 2011). Results are presented in comparative tables, and predominant outcomes are illustrated by quotes. Each farmer has authorized this information to be shared, provided that his or her identity would not be revealed.

RESULTS

Federal Conservation Programs' Role in the Adoption of Conservation Practices

Federal conservation programs have covered costs associated with some practices adopted by study participants, though other practices have been paid for with out-of-pocket or third-party investments (**Figure 1**). Overall, CSP had provided farmers an incentive to enhance their conservation practices,

Farm	CSP promoted practices	Year	Status
1	No-till ahead of soybeans and strip-till ahead of corn	1993	Integrated
2	Conservation tillage, terraces and nutrient management	2000	Expansion
3	No-till, buffer strip along creek and set-aside land with wildflowers (no longer)	1980	Integrated
	Nutrient management	1997	Adjusting
6	Strip-till (100% of farm)	2008	Integrated
	Nutrient management	1980	Integrated
7	No-till (on about 20% of acres)	-	Expansion
	Integrated pest and nutrient management	-	Integrated
9	Terraces	1990	Integrated
10	Rotational grazing and nutrient management	-	Integrated

Farm	EQIP promoted practices	Year	Status
1	Cover crops (used on 80% of the farm)	2014	Expansion
2	Cover crops (used for three years)	2014	No longer
3	Cover crops	2015	Trial
	Windbreaks	-	Integrated
6	Side-dresser purchase	1999	Integrated
8	Strip-till (on 1/3 of the farm)	2008	Integrated
	Cover crops on two rented farms	2008	Trial
9	Nutrient management	-	Integrated
10	Cover crops (100 acres)	2014	Expansion
	Pond and waterways	-	Integrated

Farm	CRP promoted practices	Year	Status
1	Waterways and set aside three hilltops	-	Integrated
2	Waterways (1/4 is CRP)	2000	Integrated
	Filter strip along an open ditch	2009	No longer
3	Waterways and wetlands	-	Integrated
4	Buffer strips along streams (60 acres)	1999	Integrated
5	Wetlands	-	Integrated
6	Buffer strips and pollinator habitat	2009	Integrated
	Ponds	1962	Integrated
7	Buffer strips	-	Integrated
8	Buffer strips	2008	Integrated
9	Buffer strips and stream restoration	1990	Integrated
10	Buffer strips and set aside hillsides	-	Integrated

Farm	Out-of-pocket investment	Year	Status
1	Nutrient management	2008	Integrated
	Set aside hilltops	-	Integrated
	Buffer along stream	-	No longer
2	Nutrient management (soil sampling, manure spreading)	2006	Integrated
	No-till (40 acres)	2017	Trial
3	Terraces	1994	Adjusting
4	No-till for soybeans, strip-till for corn and nutrient management	1996	Integrated
	Cover crops and waterways	2015	Trial
5	Conservation tillage and nutrient management (spring-applied nitrogen)	2001	Integrated
	Cover crops and strip till	2015	No longer
7	Waterways	1982	Integrated
8	Nutrient management and waterways	2008	Integrated
	Precision agriculture with grid soil sampling and conservation tillage	-	Integrated
9	Prairie strip	1990	Integrated
10	Extended crop rotation (corn, soybeans, oats, alfalfa)	-	Integrated

Farm	Other sources	Year	Status
2	Bioreactor – 100% paid through non-federal organization	2008	Trial
4	Pollinator habitat (one acre) – cost-share from pheasants forever/DNR	-	Integrated
7	Cover crops – cost-share from ADM's sustainability program	-	Integrated
8	Bioreactor – 100% paid through non-federal organization	2008	Trial
9	Cover crops and bioreactor – 100% paid through non-federal organization	2008	Trial

Farm	Conservation Compliance	Year	Status
1	No-till	-	Integrated
2	Conservation tillage	-	Integrated
3	No-till, terraces	-	Integrated

FIGURE 1 | Conservation-related practices adoption process by farm environmental leaders in Iowa. Source: Field research. Interviewed farmers define reduced tillage practices as follows: conservation tillage–tillage with some residue left covering soil; strip-tillage–disturbs only the portion of the soil that is to contain the seed row; no-till–no soil disturbance. Conventional tillage is defined as tillage with no residue left covering soil. In the case of conservation compliance, only farmers with HEL were reported.

often through the implementation of conservation tillage and nutrient management plans. EQIP had also aided in conservation adoption, often incentivizing farmers to try cover crops. Many of the practices that improve wildlife habitat and water quality, such as pollinator habitat and buffer strips, were established with support from CRP. Conservation Compliance had had less of an impact among the interviewed farmers, but it had helped catalyze the adoption of conservation tillage in a few cases.

For interviewed farm environmental leaders, federal conservation programs had played an important role in incentivizing conservation efforts. Many interviewed farmers cited these programs as a reason they adopted the practices they have: "I bought my side-dresser 18 years ago because the EQIP program, state EQIP, they paid for it.…the payment each year was like four thousand dollars.…And I took them dollars and paid [for the side-dresser] and we still have the side-dresser, we're still using it" (Farmer 6). Farmer 8 shared a similar sentiment: "The government money has been a big help. Because when profit margins are low, are we able to show a return on the year we do it (implement conservation practices)? Not necessarily" (Farmer 8).

Some practices, such as no-till and strip-till, had been a part of participants' farming operations for a relatively long time (**Figure 1**). Other conservation practices, particularly cover crops, were newer and in the process of being adopted by farmers. Many farmers discussed how high implementation costs had been a barrier to cover crop use: "I've tried cover crops out here. … I've got all this other work to do and then for me to go and spend fifty dollars an acre to put a cover crop out there that's going to possibly save thirty pounds of nitrogen? The economics isn't there for that" (Farmer 5). "It can work. But you know, it's, there again, a pretty expensive thing to do. Until you see the benefit of increased soil activity and what not, it's hard to get a return off of that or see a return" (Farmer 4).

Several farmers directly emphasized the importance of government programs in overcoming this cost barrier: "It's expensive, and once again the government will pay you for one time to use cover crops. Well, the way to put it on is with an airplane. That gets pretty expensive. Again, how do you pay for this without government subsidy?" (Farmer 6). "I've had cost-share for cover crops. … We really can't afford to do a cover crop without cost-share. And even with cost share, it doesn't come close to covering the cost" (Farmer 9). "Recently there's the cover crop push, so Badger Creek Watershed was slated for extra [EQIP] funding. So I decided it was a good opportunity to get on board with that" (Farmer 10). These quotes illustrate how participants valued government conservation programs to incentivize adoption of new practices, especially cover crops.

Several interviewed farmers, however, had also implemented multiple practices using their own money with no government support. **Figure 1** differentiates practices adopted due to Farm Bill conservation programs incentives (on the left side) from practices adopted through out-of-pocket investments and third-party investments (on the right side). Many participants expressed that they feel a responsibility to be a good steward regardless of the availability of public funds: "I had looked at that [the CSP program], I was real enthused about that, and just never did get into it. I guess I thought it's kind of a philosophy, what's right I should do on my own. I shouldn't take government money for doing something that's right" (Farmer 4).

Other participant farmers cited inflexible requirements as reasons for not using Farm Bill conservation programs: "We were limited on what we could put for nitrogen. When corn and bean prices went up, we kind of got up on our nitrogen rate, shooting for higher yields. … So that's kind of why I got out of that program. It just wasn't working. … I felt like it [the CSP program] could be holding us back on our yield potential and profitability a little bit" (Farmer 2). Farmer 5 shared a similar sentiment regarding conservation program requirements: "I went up to my NRCS [office] … and I said I put Agrotain on late-season application nitrogen, do I qualify for this $25 an acre? … They said, well, Agrotain isn't approved. The only thing that is approved is N-Serve. … Agrotain does the same thing for liquid nitrogen, but they didn't recognize it, so I didn't quality for it [the EQIP program]. Frustration!" (Farmer 5).

Programs' Strengths, Weaknesses, and Potential Improvements

Interviewed farmers provided insight into the perceived strengths and weaknesses of conservation programs, and many farmers also discussed recommendations on how programs' shortcomings can be addressed. In nearly all cases, farmers described each program's positives in very general terms (saying they have had generally good experiences with certain programs, overall they like certain programs, etc.) while they described the negatives more specifically. Therefore, this section primarily focuses on programs' weaknesses and how study participants believed each program could be improved. Some farmers were poorly aware of administrative details of individual programs, but all were able to identify which programs they have used and were able to give insight into their experiences. Many of their resulting recommendations involved specific tweaks and adaptations to existing programs, but other recommendations involved the transformation of current approaches and mentalities.

CSP

Though many farmers reported that their overall impression of CSP was positive, all interviewed farmers who had experience with the program had at least one critique (**Table 2**). The biggest complaints surrounded the program's inflexible requirements and decreased payment rates.

One farmer recalled a particular experience with CSP's rigid requirements:

> "I remember going into NRCS…and I complained. I said I want to be a part of this program [CSP] but it's real difficult to get enough points to get above Tier 1 because it's so heavily no-till oriented. And the comment I got from the NRCS guy was, "Well, everybody should no-till." That's very frustrating to me... It's a lot different how I farm here than in southern Iowa. And that's a frustration I have with government programs because they always want to treat us as farming the same way." (Farmer 5)

Another interviewed farmer shared the same sentiment:

> "The government requires you to do things almost to the letter. … I don't really care for tissue samples—they have more sophisticated tools now to measure nitrogen in the soil. Well, no, we've got to have tissue samples [to meet CSP requirements]. … If you agree to do this practice, you pretty much have to do it the way they state. And that's why a lot of farmers don't like the CSP. It's too confining." (Farmer 7)

Overall, four of the ten interviewed farmers cited inflexible requirements as a weakness of the CSP program (**Table 2**).

The same number of farmers commented on CSP's reduced payments: "They aren't paying as much now as they did… they cut it [the CSP payment rate] down quite a bit. I think the max is around $28 an acre now. Which is still fine, but it [higher payments] just helps offset some of those costs" (Farmer 7). "It (the CSP program) has been a help, an incentive. In years like this I'm glad the check is in the mail. … I remember at the sign-up thinking, "Is this worth it?." As I recall, the funding dropped from my initial contract to the second contract. The amount of funding went down, and the enhancements became a little more challenging" (Farmer 10).

Farmer 10 also brings up another theme: that CSP "enhancements" (conservation steps taken to maintain or

TABLE 2 | Farm environmental leaders' perspectives on conservation programs.

Program	Farmer	Program status	Improvement	Perspectives
CSP	1	Enrolled	Budget	Promising program, but congress decreased its budget, so farmers didn't get the benefit they should have
	2	No longer enrolled	Paperwork, requirements	The program wasn't flexible enough, especially regarding how the amount of nitrogen in manure was calculated. The paperwork and documentation were also a hassle, which has discouraged me from re-enrolling
	3	No longer enrolled	Requirements	We had a positive experience, but to stay in the program you have to do more. Some of the additional enhancements made no sense
	5	Enrolled	Requirements	The program is not flexible enough. It's very no-till oriented, but there are other practices that are a better fit for my farm and the environmental issues it faces
	6	No longer enrolled	Payments	CSP should continually pay farmers for conservation practices, not just incentivize farmers to get started on a practice
	7	Enrolled	Requirements, payments	Overall, it has been a good experience. Requirements can be restrictive, though. You have to do every practice just the way they say
	8	Never enrolled	Requirements	It seems like a beneficial program. Many of the enhancements make sense for landowners, but not necessarily renters, so I haven't enrolled
	9	No longer enrolled	Payments, requirements	In some instances, the payment rates are not high enough to make participation worthwhile. Also, many of the enhancements are operation-wide changes; if you're renting, it can be hard to get all your landlords on board
	10	Enrolled	Requirements	Positive experience overall. Provides a good incentive to help further improve conservation. Payments are helpful, though they aren't as high as they used to be. The program doesn't allow for spontaneity
EQIP	1	Enrolled	Paperwork, requirements	Cover crop cost-share was a frustrating experience because of excessive paperwork, strict seed mix requirements, and slow payments
	3	Never enrolled	Budget	Have tried to sign up twice, but it was full both times. Budget isn't big enough
	5	Never enrolled	Requirements, county differences	Some requirements are needlessly strict. I used a nitrogen stabilizer but didn't qualify for the program because it wasn't their approved brand. Also, EQIP is run by the county, which results in unfair county-by-county differences
	6	No longer enrolled	Requirements	In some cases, the government pays for things that don't need to be done. For example, cover crops work on some acres, but in other cases they aren't the best option
	7	No longer enrolled	Payments	The program's budget and cost-share rates are weaknesses. You get paid well the first year, but payments are lower after that. It's hard to get re-enrolled at all
	9	No longer enrolled	Payments	The program doesn't pay enough. I use ADM's cost-share program now, and it pays significantly more
CRP	1	Enrolled	Requirements, payments	Some requirements are nitpicky and illogical. Also, CRP payments didn't keep up with the rising rental rates
	2	Enrolled	Requirements	It's a good program, but it's a lot of work to control the thistles and volunteer trees that grow on CRP ground. I've taken some land out of the program for this reason, but I do plan to re-enroll the environmentally sensitive areas
	3	Enrolled	Requirements	Some requirements are non-sensical and prevent good outcomes. For example, we had a wetland in CRP for 15 years, but we have to farm it for 3 years before we can enroll it again
	4	Enrolled	Payments, requirements	Good program. Payments were previously too low, but now they're too high. They lag behind rental rates and should be adjusted on an annual basis. Mid contract management requirements could be made more flexible, too
	5	Enrolled	Payments	The program is good in theory, but its implementation can be frustrating. Rates should be based on current economics and determined yearly
	6	Enrolled	Targeting	People shouldn't be able to enroll entire fields of flat land. Eligibility should be based on slope and proximity to bodies of water
	8	Enrolled	Payments	CRP payments are high compared to cash rents, so CRP competes with renters. CRP payments should be more in line with cash rents
	9	Enrolled	Targeting	CRP should be targeted to sensitive areas. Right now, the higher the CSR, the higher the payment. We should be paying more to get the vulnerable land
	10	Enrolled	—	Fairly happy with the program. Being able to graze CRP ground is a positive

Source: Field research. Farmers who had no comments or had never tried to enroll with specific programs are not listed.

improve CSP status) do not meet some farmers' needs and goals, especially as they move further along in the program. For example, farmer 3 stated: "In order now to stay in it [CSP] you have to do more. We were already at the top. Some of the things that you had to do [to progress in the program] really didn't make sense. So we're not in that anymore." Interviewed farmers also felt that enhancements may not be a good fit for operations that include rented land (**Table 2**).

EQIP

As with CSP, payment rates and strict requirements were the most frequently reported weaknesses of EQIP. One interviewed farmer cited payment rates as a reason he had switched to private cost-share programs:

> "Yeah, I use cost-share on that [cover crops]. It was NRCS, and then I'm with ADM. … The first year you do it (EQIP), they'll cost share $25 an acre. And then after you've done that it goes down to $15 an acre, and you have to get in there pretty quickly or else the money is gone. The sustainability program I'm with at ADM, they'll pay $25 an acre, plus I still get the ten cents for the beans too [ten cents per bushel sustainability premium]." (Farmer 7)

Another farmer, also involved with Archer Daniels Midland's sustainability program, shared similar thoughts: "If I went through state [EQIP]—I've raised cover crops before—I'd get a fifteen dollar an acre subsidy. If I go through ADM, I still get the twenty-five dollars" (Farmer 9).

Other farmers critiqued the program's red tape and strict requirements:

> "Very, very frustrating. I signed all the papers, thought we had signed all the papers. I think we went back either three or four times to sign papers. … Even that [the seed mix requirement] was somewhat frustrating in that you've got to go up and get approval for a given mix with a given amount per acre. And we thought we had everything all set up, and oh no, you can't do that. You've got to have either a different species in there or you've got to have more per acre. Well what's the difference if you get a good cover crop?" (Farmer 1)

Two other interviewed farmers shared similar opinions, as evidenced in **Table 2**.

County-by-county differences in EQIP implementation was another weakness cited by one farmer: "The EQIP program is run by the county. … our NRCS guy only wants to do large projects. … He wants to spend the money on one or two farmers in the county. Other counties do a better job of that [spreading money around]. So there's another frustration, how it's managed. It's different from one county to another" (Farmer 5).

CRP

Many interviewed farmers viewed CRP as a good program in the sense that it results in positive environmental benefits; however, eight of the ten farmers believed the program had at least one weakness (**Table 2**). The CRP payment rate was the most frequently discussed weakness, mentioned by four farmers. Specifically, farmers expressed that the rate is not aligned well-enough with cash rents: "Right now cash rent is running $240 to $260 in this area, and CRP is paying like $320 or $350… the CRP will compete with us [renters] for acres" (Farmer 8). Multiple farmers suggested more frequent rate adjustments to solve this problem: "I think you can have a 10-year contract, but maybe how much you get should be based as a year-to-year thing" (Farmer 4); "To me it should be a yearly rate based on the economics" (Farmer 5).

Several farmers also cited certain program requirements as weaknesses, suggesting that some rules may actually be preventing environmentally beneficial outcomes: "We had a square patch that was in the early CRP. … It came out this year, and he had to put it back in, and he had to kill the seeding on the hill—spray it and kill it—and re-seed it to the weed mix, the pollinator mix. [It was] wonderfully established, couldn't have been better as far as stabilizing the soil and all that kind of stuff" (Farmer 1). Another farmer had a similar experience with a wetland enrolled in CRP:

"You put it in for 15 years, but you take your payments over 10 years. And then they said even though you didn't get a check for that 5 years, you have to treat it like it wasn't farmed. And so you have to start all over again if you wanted to put it back into a CRP program, because it has to have been farmed 3 of the 5 years previous. … (in order to get it into a different program) we actually had to start farming it." (Farmer 3)

Other interviewed farmers suggested CRP could be improved by being better targeted to environmentally sensitive land: "I think from an environmental standpoint, we should be targeting CRP. We shouldn't be putting really good land into CRP, I don't think. We should be paying more to get the poor land in. Right now you get paid more if you put good land in" (Farmer 9). "There's now quarter sections of good, black, flat ground that should be in production that are getting $300 an acre in CRP. … Why did they allow that?... It (the CRP rate) is based off of the corn suitability rating… it should be based off of slope factors and closeness to open water" (Farmer 6).

Transformations

Most participants suggested improvements that involved relatively small changes to existing conservation programs, as illustrated above. These changes fall under the categories of "tweak" or "adapt." However, most interviewed farmers also proposed more transformational changes that did not necessarily fall under the umbrella of individual conservation programs.

Several farmers expressed that landlords' mentalities toward conservation must change, especially when it comes to their unwillingness to compensate farmers for adopted practices: "They want the same cash rent [for land with conservation]. … I [the landlord] want X dollars an acre. You (the farmer) take care of everything. If you want, put it in CRP. Whatever you want, that's just fine with me as long as I get my cash rent" (Farmer 1). "We have something like twenty-five landlords, and it's a lot more difficult to put conservation practices on rented land because we can't afford to pay for it. … The land I own has a lot more conservation on it than land I don't own" (Farmer 9).

And some study participants, although opposed to government interference, doubted the effectiveness of the current voluntary programs and acknowledge that mandatory approaches may be on the horizon: "People (farmers) don't care (about conservation). And I think what's going to bring it around, and I don't want mandates, is when the people in the city and the country don't have fresh water. Look at the lawsuit from Des Moines… When it gets mandated then we're going to have to wake up" (Farmer 6).

Along the lines of mandated conservation, farmers were asked their opinion on the expansion of conservation compliance, which currently applies only to farmland classified as highly erodible land (HEL), to all cropland. Surprisingly—given the complaints about red tape and strict requirements for CSP, EQIP, and CRP—eight of the ten farmers expressed what we believe to be a transformative view, in that they would support the program's expansion beyond HEL (**Table 3**). Several farmers cited the need for accountability as rationale for this viewpoint. Farmer 8 explained, "If the government's kicking in money (for subsidized crop insurance), it's not wrong to ask them (those who receive it) to be accountable. … For Iowa I think it's [expanding conservation compliance] well-justified" (Farmer 8). Farmer 7 shared a similar opinion: "I don't think it [expanding conservation compliance] would be a bad thing. I think the taxpayers are asked to spend a lot of money on this cost-share. … If you're going to take the taxpayers' money to buy down your crop insurance, you should give something back in return" (Farmer 7).

Other farmers believed expanding conservation compliance would help improve tillage practices: "It just blows my mind. You can take one little piece of ground and call it HEL and you've got to follow all these certain restrictions, but yet the neighbor across the road, they're tilling it until it's black. … If they come out with something like that [expanded conservation compliance], I wouldn't be opposed to it because we've got to cut back on tillage I think" (Farmer 2). "I think it [expanding conservation compliance] would be a terrific idea. … [It would] maybe be an incentive for farmers that are plowing up right now… They don't need to do that. So give them an incentive to wait until spring to do whatever field work they are going to do" (Farmer 3).

TABLE 3 | Farm environmental leaders' perspectives on the expansion of conservation compliance (favorable views in light green).

Farmer	Status	Perspective
1	HEL	We have some land that's HEL. It's long-term no-till at this point, which keeps the NRCS happy. Expanding conservation compliance would be a good thing, but I don't think it'll happen
2	HEL	We have some HEL ground. We're just required to leave a certain amount of residue on it. If conservation compliance was expanded, I wouldn't be opposed to it; we need to reduce tillage
3	HEL	We have one farm that's HEL. We have terraces on it and do no-till. I think expanding conservation compliance would be a terrific idea. Many farmers plow up everything, including soybean ground. Maybe this would incentivize them to reduce their tillage
4	No HEL	I am definitely in favor of crop insurance and conservation compliance being coupled together on HEL. I'm not opposed to this being expanded to non-HEL, but I also don't like being told how to run my farm
5	No HEL	I strongly oppose expanding conservation compliance. Regulations don't work. They don't level the playing field. I can do a better job managing my farm than someone sitting in Washington D.C. The program has implementation issues, too; people who have done the right thing have been found out of compliance simply due to weather events
6	No HEL	Expanding conservation compliance would force farmers to do conservation practices to get their subsidies. Wouldn't you as a taxpayer want to see a benefit from the subsidies you pay for?
7	No HEL	I don't think expanding conservation compliance would be a bad thing. Taxpayers spend a lot of money on crop insurance cost-share. In order to get subsidized you should give something back
8	No HEL	If the government is kicking in money by subsidizing your crop insurance, it's not wrong to ask you to be accountable. Expanding conservation compliance would be fine in Iowa, but I can't speak for other states
9	No HEL	I've been a big promoter of expanding conservation compliance, and I think every farm should have a conservation plan
10	—	—

Source: Field research.

CONCLUSION

Results revealed that many of the farm environmental leaders we interviewed had had generally positive experiences with federal conservation programs, but many were also quick to point out programs' weaknesses and suggest potential improvements. Conservation programs had covered part of the costs for establishing practices, but nine of the ten interviewed farmers had implemented at least one practice with out-of-pocket or third-party investments. Some believed that conservation is their own responsibility as a good steward, and therefore were not as involved in conservation programs. Others cited red tape, external interference in their farming operations, or low payment rates as reasons for not utilizing federal dollars. Nonetheless, for most participants conservation programs had played an important role in incentivizing adoption and offsetting costs of conservation practices. Building on previous studies (Reimer et al., 2012; Arbuckle and Roesch-McNally, 2015; Nebel et al., 2017), these findings emphasize the relevance of conservation policies in helping reduce risk for farmers trying new practices.

According to interviewed farmers, all three of the federal conservation programs discussed could benefit from more flexible requirements and higher payment rates. Program-specific tweaks and adaptions were also suggested: CSP could be improved with some adjustments to enhancements, EQIP with more uniform implementation, and CRP with more targeted implementation. Changes such as these could potentially increase farmers' participation in federal conservation programs and catalyze conservation nationwide.

While many of participants' suggestions involved tweaks and adaptions to existing programs, others pointed toward major policy transformations (Hall, 1993; Atwell et al., 2011). A full 80% of the interviewed Iowa farm environmental leaders favored more transformative actions such as expanding conservation compliance to all cropland, providing incentives to landowners to help overcome the challenges faced by renters, and even regulatory measures. In contrast to participant farmers, more traditional and powerful organizations, such as the Farm Bureau, tend to support adaptations but not transformative approaches to existing Farm Bill conservation programs (Medina et al., 2020).

Interviewed farmers discussed the challenges of implementing conservation on rented acres, the possibility of mandatory conservation regulations, and showed support for the expansion of conservation compliance to all cropland. Challenges such as how to promote conservation practices to a growing number of tenant farmers and landlords have also been reported elsewhere (Varble et al., 2016). As these issues are not addressed by current programs, they may require a more transformative approach in future policy revisions.

As stakeholders prepare to lobby future Farm Bills, it is critical to consider farmers' perspectives on federal conservation programs (Roesch-McNally et al., 2017). To some extent, the farm environmental leaders interviewed aligned themselves with the perspectives of environmental NGOs, advocating for transformative approaches, such as the extension of Conservation Compliance to all farms receiving crop insurance subsidies. But in many issues interviewed farmers also share views with commodity and farmer groups, preferring less red tape and more flexibility within federal conservation programs (Medina et al., 2020).

This study focused only on farmers who had been formally recognized as environmental leaders. Thus, they were exemplars of farmers who have strong conservation ethics and operationalize them through the establishment much greater levels of soil and water conservation practices than is typical. In this sense, the results presented here do not represent perspectives of farmers in general across U.S. agricultural systems. Future research efforts should focus on less conservation-oriented farmers to evaluate their perspectives on the strengths and weaknesses of conservation programs. That said, these exemplary farmers who had substantial knowledge of and experience with major U.S. soil and water conservation programs provided important insights into the strengths and weaknesses of programs and how they fit or did not fit within their long-term economic and environmental sustainability strategies. Their perspectives helped point to potential improvements that could be made to current programs, suggested alternative conservation programs/measures, and importantly, potential transformations in conservation policy (e.g., extension of conservation compliance to all cropland) that could lead to major increases in soil and water conservation actions across the U.S. agricultural landscape.

DATA AVAILABILITY STATEMENT

The datasets generated for this study will not be made publicly available in order to protect farmers' identity.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Iowa State University. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

GM and JA: carried out fieldwork in Iowa. CI supported data processing. All authors listed have made a substantial, direct and intellectual contribution to the written work, and approved it for publication.

FUNDING

This research was also supported in part by the Iowa Agriculture and Home Economics Experiment Station, Ames, Iowa.

ACKNOWLEDGMENTS

The authors wish to thank the Fulbright Scholar Program for their support for the first author's visit to Iowa State University. We also wish to thank Hanna Bates, whose MS thesis research (Rosman, 2015) laid the groundwork for this research project.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.497943/full#supplementary-material

REFERENCES

Arbuckle, J. (2013). Farmer support for extending conservation compliance beyond soil erosion: evidence from Iowa. *J. Soil Water Conserv.* 68, 99–109. doi: 10.2489/jswc.68.2.99

Arbuckle, J. (2016). *Iowa Farm and Rural Life Poll: 2015 Summary Report*. Extension report PM 3016, Iowa State University.

Arbuckle, J., and Roesch-McNally, G. (2015). Cover crop adoption in Iowa: the role of perceived practice characteristics. *J. Soil Water Conserv.* 70, 418–429. doi: 10.2489/jswc.70.6.418

Arbuckle, J. G. J. (2009). "Cattle and trees don't mix!?!: competing agri-environmental paradigms and silvopasture agroforestry in the missouri ozarks," in *Farming With Grass*, ed A. J. Franzluebbers (Ankeny, IA: Soil and Water Conservation Society), 116–133.

Atwell, R. C., Schulte, L. A., and Westphal, L. M. (2011). Tweak, adapt, or transform: policy scenarios in response to emerging bioenergy markets in the U.S. corn belt. *Ecol. Soc.* 16:10. doi: 10.5751/ES-03854-160110

Cloke, P. J., and Goodwin, M. (1992). Conceptualising countryside change: from post-fordism to rural structured coherence. *Trans. Instit. Br. Geogr.* NS17, 321–336. doi: 10.2307/622883

Corbin, J., and Strauss, A. (1990). Grounded theory research: procedures, canons, and evaluative criteria. *Qual. Sociol.* 13, 3–21. doi: 10.1007/BF00988593

ERS (2020). *USDA ERS-Conservation Programs*. Available online at: https://www.ers.usda.gov/topics/natural-resources-environment/conservation-programs/ (accessed August 21, 2020).

EWG (2017). *Less Farm Pollution, More Clean Water: An Agenda for Conservation in the 2018 Farm Bill*. Ames, IA: EWG. p. 7.

Farm Bureau (2017). *2017 Policy Book*. Jefferson City, MO; Missouri Farm Bureau. p. 136.

Guest, G., Bunce, A., and Johnson, L. (2006). How many interviews are enough? *Field Methods* 18, 59–82. doi: 10.1177/1525822X05279903

Hall, P. (1993). Policy paradigms, social learning, and the state: the case of economic policymaking in Britain. *Comp. Polit.* 25, 275–296. doi: 10.2307/422246

Hoppe, R. A., and MacDonald, J. M. (2013). Updating the ERS farm typology. *SSRN Electron. J.* 110. doi: 10.2139/ssrn.2267293

Lambert, D. M., Sullivan, P., Claassen, R., and Foreman, L. (2007). Profiles of US farm households adopting conservation-compatible practices. *Land Use Policy* 24, 72–88. doi: 10.1016/j.landusepol.2005.12.002

Mason, M. (2010). Sample size and saturation in PhD studies using qualitative interviews. *Forum Qual. Soc. Res.* 11, 1–19. doi: 10.17169/fqs-11.3.1428

Mcfadden, J. R., and Hoppe, R. A. (2017). *The Evolving Distribution of Payments From Commodity, Conservation, and Federal Crop Insurance Programs*. Washington, DC: USDA.

Medina, G., Isley, C., and Arbuckle, J. (2020). Promoting sustainable agriculture: Iowa stakeholders' perspectives on the US Farm Bill conservation programs. *Environ. Dev. Sustainability*. doi: 10.1007/s10668-019-00572-5

Medina, G., Potter, C., and Pokorny, B. (2015). Farm business pathways under agri-environmental policies: lessons for policy design. *Estud. Soc. Agric.* 23, 5–30. Available online at: https://revistaesa.com/ojs/index.php/esa/article/view/493

Nebel, S., Brick, J., Lantiz, V., and Trenholm, R. (2017). Which factors contribute to environmental behaviour of landowners in Southwestern Ontario, Canada? *Environ. Manage*. 60, 454–463. doi: 10.1007/s00267-017-0849-9

Prokopy, L. S. (2011). Agricultural human dimensions research: the role of qualitative research methods. *J. Soil Water Conserv.* 66, 9A−12A. doi: 10.2489/jswc.66.1.9A

Prokopy, L. S., Floress, K., Arbuckle, J. G., Church, S. P., Eanes, F. R., Gao, Y., et al. (2019). Adoption of agricultural conservation practices in the United States: evidence from 35 years of quantitative literature. *J. Soil Water Conserv.* 74, 520–534. doi: 10.2489/jswc.74.5.520

Ranjan, P., Church, S. P., Floress, K., and Prokopy, L. S. (2019). Synthesizing conservation motivations and barriers: what have we learned from qualitative studies of farmers' behaviors in the United States? *Soc. Nat. Resour.* 32, 1171–1199. doi: 10.1080/08941920.2019.1648710

Reimer, A. (2013). *Trends in U.S. Agricultural Conservation Programs*. Harrisburg, PA: National Agricultural & Rural Development Policy Center.

Reimer, A., Gramig, B., and Prokopy, L. (2013). Farmers and conservation programs: explaining differences in environmental quality incentives program applications between states. *J. Soil Water Conserv.* 68, 110–119. doi: 10.2489/jswc.68.2.110

Reimer, A., and Prokopy, L. (2014). Farmer participation in U.S. farm bill conservation programs. *Environ. Manage.* 53, 318–332. doi: 10.1007/s00267-013-0184-8

Reimer, A. P., Weinkauf, D. K., and Prokopy, L. S. (2012). The influence of perceptions of practice characteristics: an examination of agricultural best management practice adoption in two indiana watersheds. *J. Rural Stud*. 28, 118–128. doi: 10.1016/j.jrurstud.2011.09.005

Roesch-McNally, G., Arbuckle, J. G., and Tyndall, J. C. (2017). Soil as social-ecological feedback: examining the "ethic" of soil stewardship among corn belt farmers. *Rural Sociol.* 83, 145–173. doi: 10.1111/ruso.12167

Rosman, H. (2015). *The Farmer and Farm Networks: An Examination of Conservation Adoption in the U.S. Corn Belt.* Iowa State University. Available online at: https://lib.dr.iastate.edu/cgi/viewcontent.cgi?article=5613andcontext=etd (accessed August 01, 2019).

Rundquist, S., and Cox, C. (2016). *Fooling Ourselves*. Washington, DC: Environmental Working Group.

Stuart, D., Denny, R. C. H., Houser, M., Reimer, A. P., and Marquart-Pyatt, S. (2018). Farmer selection of sources of information for nitrogen management in the US midwest: implications for environmental programs. *Land Use Policy* 70, 289–297. doi: 10.1016/j.landusepol.2017.10.047

Ulrich-Schad, J. D., de Jalón, S. G., Babin, N., Pape, A., and Prokopy, L. S. (2017). Measuring and understanding agricultural producers' adoption of nutrient best management practices. *J. Soil Water Conserv*. 72, 506–518. doi: 10.2489/jswc.72.5.506

USDA (2015). *2012 National Resources Inventory*. Available online at: http://www.nrcs.usda.gov/technical/nri/12summary (accessed August 01, 2019).

USDA (2016). *Conservation Stewardship Program Fact Sheet*. Available online at: https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/programs/financial/csp/?cid=nrcseprd1296837 (accessed January 08, 2019).

USDA (2017). *2017 Census of Agriculture*. USDA National Agricultural Statistics Service. Available online at: www.nass.usda.gov/AgCensus (accessed August 01, 2019).

USDA (2018). *Environmental Quality Incentives Program Fact Sheet*. Available online at: https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/programs/financial/csp/?cid=nrcseprd1296837 (accessed August 01, 2019).

USDA (2019). *Conservation Reserve Program Fact Sheet*. Available online at: https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/programs/financial/csp/?cid=nrcseprd1296837 (accessed January 08, 2019).

Varble, S., Secchi, S., and Druschke, C. G. (2016). An examination of growing trends in land tenure and conservation practice adoption: results from a farmer survey in Iowa. *Environ. Manage.* 57, 318–330. doi: 10.1007/s00267-015-0619-5

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 05 November 2020
doi: 10.3389/fenvs.2020.556452

Spatio-Temporal Patterns of Crops and Agrochemicals in Canada Over 35 Years

Egina Malaj[1], *Levi Freistadt*[2] *and Christy A. Morrissey*[2,3]*

[1] Toxicology Centre, University of Saskatchewan, Saskatoon, SK, Canada, [2] Department of Biology, University of Saskatchewan, Saskatoon, SK, Canada, [3] School of Environment and Sustainability, University of Saskatchewan, Saskatoon, SK, Canada

In an effort to feed a growing world population, agriculture has rapidly intensified over the last six decades, relying heavily on agrochemicals (fertilizers, insecticides, fungicides, and herbicides) to increase and maintain desired crop yields. Despite environmental concerns in Canada's agricultural regions, long-term patterns of changing crops and the associated trends in the proportion of cropland treated with agrochemicals are poorly documented. Using the Canadian Census of Agriculture, we compiled historical data over 35 years (eight census periods: 1981–2016) on agrochemical applications, measured as the proportion of cropland treated with pesticides and fertilizers and the associated crop classes, to identify and interpret spatial and temporal trends in Canada's agricultural practices across 260 census units. Due to differences in agricultural practices, soil, and climatic conditions across the country, the Pacific (British Columbia), Prairie (Alberta, Saskatchewan, Manitoba), Central (Ontario, Quebec), and Atlantic (Nova Scotia, New Brunswick, Newfoundland/Labrador, Prince Edward Island) regions were analyzed separately. Most of the agrochemicals in Canada were applied in the Prairie and Central regions, which combined comprise 97% of the total cropland. Fertilizers were the dominant agrochemicals across Canada applied on 48% (Pacific) to 78% (Prairie) of the total cropland area, followed by herbicides, which were applied on 30% (Pacific) to 81% (Prairie) of the total cropland area in 2016. Notably, we observed significant changes between 1996 and 2016 in area treated with fungicides and insecticides, which increased by 412% and 50% in the Prairie region and by 291% and 149% in the Central region, respectively. The proportion and distribution of crops shifted in favor of more oilseeds and soybeans in the most intensive Prairie and Central regions, whereas cereals decreased over the same time period. Our analysis of past and current trends of agrochemicals and cropping patterns within Canada indicates a rapid and systemic increase in chemical use, and policies that promote a shift toward lower chemical reliance through sustainable agricultural practices are urgently needed.

Keywords: Canadian Census of Agriculture, cropping system, agricultural intensification, sustainable agriculture, land cover change, fertilizers, pesticides

OPEN ACCESS

Edited by:
Mark A. Drummond,
United States Geological Survey (USGS), United States

Reviewed by:
Elzbieta Antczak,
University of Łódź, Poland
Angelina Sanderson Bellamy,
Cardiff University, United Kingdom

Correspondence:
Christy A. Morrissey
Christy.morrissey@usask.ca

Specialty section:
This article was submitted to Land Use Dynamics, a section of the journal Frontiers in Environmental Science

Received: *29 April 2020*
Accepted: *12 October 2020*
Published: *05 November 2020*

Citation:
Malaj E, Freistadt L and Morrissey CA (2020) Spatio-Temporal Patterns of Crops and Agrochemicals in Canada Over 35 Years. Front. Environ. Sci. 8:556452. doi: 10.3389/fenvs.2020.556452

INTRODUCTION

Modern agriculture across the world has seen rapid shifts in technological enhancements, and increased inputs of fertilizers and pesticides that have doubled food production over the past 60 years (Godfray et al., 2010). Society has benefited from these gains through increased food security and reductions in food costs, largely made possible by higher production efficiencies and economies of scale. However, with these benefits, there is ongoing concern about the negative environmental, agronomic, and economic consequences of intensification of agrochemical use and shifting cropping practices (Pastor et al., 2019).

Unintended environmental impacts of high agrochemical use have led to biodiversity losses, reduced water quality, and increased greenhouse gas emissions. For example, large-scale studies have demonstrated that pesticides were the primary cause for the decline of grassland and farmland birds (Mineau and Whiteside, 2013; Stanton et al., 2018), terrestrial insects (Sánchez-Bayo and Wyckhuys, 2019), and aquatic invertebrates (Beketov et al., 2013). Overuse of fertilizers in recent decades has resulted in elevated nitrogen and phosphorus levels in the environment (Lassaletta et al., 2014), which in turn have polluted surface water (Goyette et al., 2016), ground water (Burow et al., 2010), and coastal zones (Howarth, 2008). In addition, nitrogen-based fertilizers are implicated as a primary source of greenhouse gas emissions in the form of nitrous oxides (Park et al., 2012). Therefore, considering the ongoing environmental concerns from the continued expansion and intensification of agricultural activities, current agricultural practices require environmentally sustainable solutions, in order to maintain desired agronomic and ecosystem services (Godfray et al., 2010; Foley et al., 2011).

Globally, large-scale agrochemical trends in pesticide and fertilizer use have been comprehensively explored (Tilman et al., 2002; Lassaletta et al., 2014) with the focus on countries where agriculture is a major industry (e.g., United States; Douglas and Tooker, 2015; Meehan and Gratton, 2016, or China; Yu et al., 2019). Similarly, recent studies in the United States have projected land cover datasets farther back in time to assess historical land use trends (Arora and Wolter, 2018), to identify socio-political and environmental issues of land use changes within an agroecological framework (Spangler et al., 2020), or to identify major drivers of shifting crop diversity (Goslee, 2020). Although, Canada is one of the world's largest producers and exporters of major crops such as cereals and oilseeds (Food and Agriculture Organization [FAO], 2018), and agriculture is an important aspect of the country's economy (Sarkar et al., 2018), systematic analysis of agrochemical use and associated cropping patterns across the country are limited. Only recently, Malaj et al. (2020) documented the spatial distribution of herbicide, fungicide, and insecticide use, and predicted fate to wetlands in one region, the Canadian Prairies. However, due to different cultural farming practices, crop types, farm sizes, and climatic conditions (Gagnon et al., 2014; Clearwater et al., 2016), agrochemical use is predicted to vary widely across the country; therefore, studies limited in their geographic scope do not adequately contextualize land use dynamics and diversity of agricultural practices at the national level (Goslee, 2020; Spangler et al., 2020). To date, there are only a few long-term, pan-Canadian data aggregations available including research on the stochastic risk from pesticides (Gagnon et al., 2014), general overviews on agricultural land use (Daneshfar and Huffman, 2016), and status reports for agri-environmental indicators of soil, water, and air quality (Clearwater et al., 2016). Detailed assessments of the proportion of cropland treated with agrochemicals, and the associated changes in specific crops are lacking across Canada. Therefore, data-driven, large-scale analysis of historical agrochemical trends (Ryberg and Gilliom, 2015) and cropping practices (Goslee, 2020; Spangler et al., 2020) in Canada are essential for understanding regional dynamics, which can help prioritize targeted, sustainable environmental and agronomic initiatives.

Here, we used a 35 year (1981–2016) spatial and temporal dataset from the Canadian Census of Agriculture (Statistics Canada, 2016) to evaluate the region-specific changes in agricultural practices. We hypothesized that increased reliance on agrochemicals over time would be associated with increased area of input intensive crops. To quantify the magnitude of the regional changes, our objectives were to: (i) assess the proportional increase in cropland treated with fertilizers and pesticides (insecticides, fungicides, and herbicides) and (ii) evaluate associated changes in the proportion of cropland planted with major input intensive crops (oilseeds, grains, and fruits/vegetables) across four agricultural regions of Canada.

MATERIALS AND METHODS

Data Formatting

All data were extracted from the Canadian Census of Agriculture database (via ODESI digital portal; Ontario Council of University Libraries [OCUL], 2020), which includes survey information collected every 5 years covering aspects of land use, agricultural production, and socioeconomics of farming across Canada (Statistics Canada, 2016). The data were available for the period 1981–2016 in the Census Consolidated Subdivision (CCS), which is the smallest survey unit available. However, the CCS units are not consistent across time, as they are often merged, split, or dropped between census years (e.g., 2,202 CCS units in 1981, 1,780 CCS units in 2001, and only 1,574 CCS in 2016). Furthermore, due to confidentiality restrictions, data in the CCS unit, where an individual or agricultural operation could be identified, are suppressed (Robertson, 1993). On average, 24% of data were suppressed every census year due to confidentiality restrictions. Statistics Canada also aggregates the data into larger Census Divisions (CD), which includes several CCS units (e.g., in 2016 there were between 1 and 19 CCS units within one CD in the Atlantic provinces, 1–12 in British Columbia, 1–18 in the Central provinces, and 1–22 in the Prairies). The CD unit is less variable over time, and the supressed data from the CCS level are included at the CD level provided that they did not breach confidentiality restrictions. CD units with missing data for total farm area and cropland were omitted, as well as units where agrochemicals were not reported. Therefore, we used the aggregated data at the CD level, which contained between 254 (in 1981) and 277 (in 2016) CD units.

This study was conducted across the Canadian agricultural landscape, which was divided into four regions: (i) the Pacific province of British Columbia (Pacific), (ii) Prairie provinces of Alberta, Saskatchewan, Manitoba (Prairie), (iii) Central provinces of Ontario and Quebec (Central), and (iv) the Atlantic provinces of New Brunswick, Nova Scotia, Prince Edward Island and Newfoundland and Labrador (Atlantic). Data were restricted to: (i) area treated with fertilizers, (ii) area treated with pesticides (insecticides, herbicides, fungicides), (iii) area planted with specific crops, (iv) area in cropland, (v) total farm area, and (vi) number of farms. The “tidyverse” R package (Wickham, 2017b) was used for data cleaning, formatting, and compilation into a master dataset, using the open source statistical program R, version 3.6.1 (R Core Team, 2019).

Calculations of Crop Type and Area Treated

Similar to Meehan et al. (2011), cropland was defined as all land in field crops, fruit and vegetable crops, nuts, and berry crops, and resulted in a total of 68 unique crops as surveyed by the Canadian Census of Agriculture (Statistics Canada, 2016). These crops were assembled into nine crop groups (e.g., spring wheat, durum wheat, and winter wheat were grouped as wheat), and based on common growing conditions, pesticide use, and farming practices, these nine crop groups were further assembled into three general crop classes: (i) cereals and pulses, (ii) oilseeds and soybeans, and (iii) fruits and vegetables (**Supplementary Table 1**). Data availability (census years by CD) varied for the different agrochemicals. Area treated with fertilizers was reported for the period 1991–2016 within each CD unit. Area treated with herbicides in the CD unit was available for the period 1981–2016, whereas area treated with fungicide and insecticide was only available for the 1996–2016 period. Before 1996, fungicide and insecticide data were jointly reported as “chemicals used to control insects and diseases” and could not be analyzed separately. Also, questionnaires distributed to farmers from Statistics Canada explicitly ask to not record seed treatments as pesticide use; therefore, only insecticides and fungicides that are sprayed or dusted are included, in the survey. This exclusion is known to underestimate actual use since insecticide and fungicide seed treatments have rapidly increased in popularity over the time period analyzed (Malaj et al., 2020).

The proportion of land treated with (i) cereals and pulses, (ii) oilseeds and soybeans, and (iii) fruits and vegetables in the CD unit was calculated as the area in each class divided by the total area in cropland. The proportion of area treated with fertilizers, insecticides, fungicides, and herbicides applied in the CD unit was calculated as the area treated with each of these agrochemicals divided by the total area in cropland. The average farm size was calculated as the area in cropland divided by the number of farm operators in a CD unit.

The 2016 Census of Agriculture boundary file for the CD units was used for spatial mapping. Mapping categories were optimized to illustrate the spatial distribution of each agrochemical group based on their specific distributions by census year. Quantile distributions were used as distributions were generally skewed to the left (Brewer, 2006). The “ggplot2” package (Wickham, 2017a) in the statistical program R, version 3.6.1 (R Core Team, 2019) was used for generating maps.

Statistical Analyses

To assess whether the proportion of land area to which agrochemicals were applied varied over census years and across different regions of Canada, we performed the analysis at the CD unit level in order to retain the hierarchical structure of the data. We used generalized linear mixed models (GLMMs) with beta distribution and logit link function to account for non-normal and continuous-based, proportional data (0–1). These models are recommended due to improved statistical inference and less biased estimates than the alternative of raw data transformation (i.e., logit transformation and use of Gaussian distributions; Bolker et al., 2009; Brooks et al., 2017; Harrison et al., 2018; Douma and Weedon, 2019). Four GLMMs were constructed to assess responses in the proportion of cropland treated with agrochemicals (fertilizers, herbicides, fungicides, insecticides) with the following structure: (i) census year, region (Pacific, Prairie, Central, and Atlantic) and their interactions as fixed effects, and (ii) CD unit as a random intercept term. Since beta distributions in GLMMs only accept values between 0 and 1 (Douma and Weedon, 2019), proportions > 1 from cropland treated more than once for herbicides (two CD units) and for fertilizers (11 CD units) were removed. Contrasts of model-retained fixed effects were calculated using Type II Wald chi-squared likelihood-ratio tests, and for significant effects, the comparison between different levels (i.e., year as fixed effect across different regions) was evaluated with multiple pairwise comparisons (Tukey’s HSD).

Similarly, we fitted GLMMs to assess whether the proportion of major crop classes changed over the census years and across different regions of Canada. For statistical purposes, changes in three major crop classes were investigated, namely: (i) cereals and pulses, (ii) oilseeds and soybeans, and (iii) fruits and vegetables (**Supplementary Table 1**). A GLMM was fitted for each class. The proportional data for each crop class were similar to the proportion of agrochemicals applied—that is, data were non-normally distributed, and continuous-based, proportions (0–1), in addition to being zero- and one-inflated. True zeros and ones occurred when a crop group was absent, or when it was the only land use in the CD unit, respectively. Therefore, zero-inflated, beta distribution GLMMs with a logit link function were fitted for each crop group (Bolker et al., 2009; Brooks et al., 2017; Harrison et al., 2018), as they better represented the distribution of this data type (Douma and Weedon, 2019). The GLMM for cereals had more ones (1.3% of the data), although it did not represent a distinct mode; therefore, we slightly shrank the one values to 0.999 to avoid fitting one-inflated models. Fruits and vegetables in the Prairies, and oilseeds and soybeans in the Pacific regions were removed from the modeling, as both these crop types were largely absent in these regions ($>$90% are zeros), and inclusion of such data heavily biases models toward zero-inflation. All

GLMMs assessing responses in the proportion of crop class had the following structure: (i) census year, region (Pacific, Prairie, Central, Atlantic), and their interactions as fixed effects, (ii) CD unit as a random intercept term, and (iii) year and CD unit as random slope. The random slope was added as it improved the remaining overdispersion of the model for each crop class (Brooks et al., 2017). Statistical testing of fixed effects was done using Type II Wald chi-squared likelihood-ratio tests, and for significant effects, the differences between different levels (i.e., year as fixed effect across different regions) were evaluated with multiple pairwise comparisons (Tukey's HSD).

Odds ratios (OR) were used to evaluate the performance of pair wise comparisons for changes in agrochemicals and crop classes over time. An OR < 1 indicates lower odds of occurrence in the earlier rather than the later census year, and an OR > 1 indicates higher odds of occurrence in the earlier rather than the later census year. Effects were considered statistically significant for p-values smaller than 0.05.

All statistical analyses were performed in the statistical program R, version 3.6.1 (R Core Team, 2019). The "glmmTMB" package was used for regression analyses (Brooks et al., 2017), residuals and assumptions of each model were checked using "DHARMa" package (Hartig, 2020), likelihood-ratio testing was done with the "car" package (Fox and Weisberg, 2019), and "emmeans" package was used for generating *post hoc* estimates (Lenth, 2019).

RESULTS

Agrochemical Use

Total cropland area treated with agrochemicals in Canada ranged from 22 to 28 million ha (1991–2016) for fertilizers, 15 to 29 million ha (1981–2016) for herbicides, 2.9 to 5.2 million ha (1996–2016) for insecticides, and 1.8 to 9.3 million ha (1996–2016) for fungicides (**Table 1**). The change in crop area treated with agrochemicals over time was region specific, and this change was more prominent in the large and agriculturally intensive Prairie and Central regions (**Table 1** and **Figures 1**, **2**). There were statistically significant differences between years, regions, and their interaction in areas treated with agrochemicals for all agrochemical groups (**Supplementary Table 2A**). Pairwise comparisons between census years show steady and significant increases over time in area treated with insecticides, fungicides, and herbicides across Canada, and increases in area treated with fertilizer for the Prairies (**Supplementary Table 3**).

There was a significant increase in the mean proportion of cropland treated with fertilizers in the Prairies between 1991 and 2016 (OD = 0.59, $p < 0.001$), and a significant decrease in all other regions (e.g., OD = 2.43, $p < 0.001$ in Pacific region; see **Table 2** for full model results). Areas treated with fertilizers in the Prairies have increased gradually through time and space (**Figures 1A**, **2A**), from 17.7 million ha (64% of the total cropland) in 1991 to 24.4 million ha (78% of the total cropland) in 2016 (**Table 1**). For other regions, the proportion of cropland treated with fertilizers, either decreased (Pacific and Atlantic regions), or remained the same (Central region; **Table 1**).

TABLE 1 | Summary of agrochemical applications in Canada between the earliest and latest year.

Agrochemical group	Year	Pacific			Prairie			Central			Atlantic		
		Area treated (ha)	Cropland treated (%)	Change in cropland treated (%)	Area treated (ha)	Cropland treated (%)	Change in cropland treated (%)	Area treated (ha)	Cropland treated (%)	Change in cropland treated (%)	Area treated (ha)	Cropland treated (%)	Change in cropland treated (%)
Fertilizers	1991	330,944	59.44		17,693,157	64.31		3,270,241	64.75		267,960	68.91	
	2016	276,887	47.68	−19.8	24,365,219	77.94	21.2	3,555,594	64.64	−0.2	238,006	59.7	−13.4
Insecticides	1996	26,346	4.66		2,366,686	8.26		456,454	8.64		85,212	20.05	
	2016	42,846	7.38	58.4	3,875,342	12.4	50.1	1,181,274	21.48	148.6	84,865	21.29	6.2
Fungicides	1996	25,031	4.42		1,465,567	5.12		247,885	4.69		79,200	18.64	
	2016	32,041	5.52	24.9	8,192,885	26.21	411.9	1,009,488	18.35	291.3	101,624	25.49	36.7
Herbicides	1981	121,157	21.32		12,631,725	51.34		2,324,505	43.14		143,353	35.28	
	2016	176,359	30.37	42.4	25,312,453	80.97	57.7	3,583,057	65.14	51.0	186,137	46.69	32.3

The earliest year was 1991 for fertilizers, 1981 for herbicides, and 1996 for both insecticides and fungicides. The latest year for all agrochemicals was 2016. Area treated (ha), percent of the total cropland treated, and percent change in cropland treated between the earliest and latest years are given for each agrochemical group and region. The agricultural regions are ordered from west to east: British Columbia (Pacific), Alberta, Saskatchewan and Manitoba (Prairie), Ontario and Quebec (Central), and Nova Scotia, New Brunswick, Newfoundland/Labrador, Prince Edward Island (Atlantic).

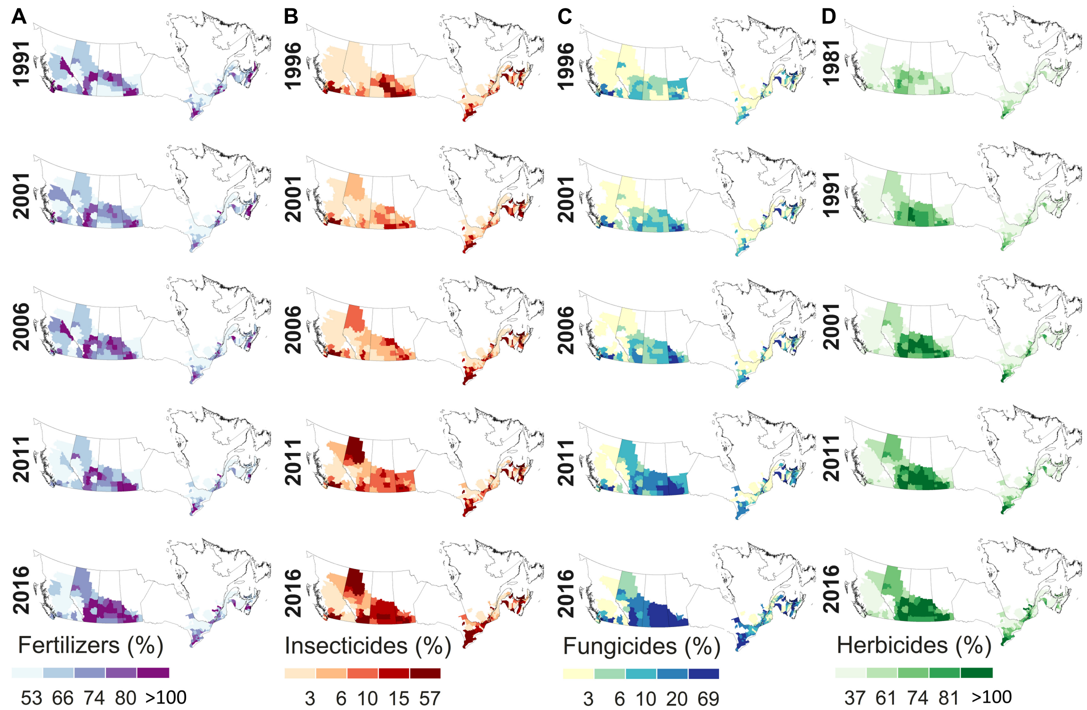


FIGURE 1 | Geographic distribution of agrochemicals calculated as percent of cropland treated with fertilizers **(A)**, insecticides **(B)**, fungicides **(C)**, and herbicides **(D)** for each census of agriculture year across Canada. Note that the date range from the Census of Agriculture varies by agrochemical group: fertilizers (1991–2016), insecticides and fungicides (1996–2016), and herbicides (1981–2016). Two census division units for herbicides and 11 census division units for fertilizers reported > 100% area cropped, suggesting frequent, repeated applications in a growing season.

Between 1996 and 2016, the increase in proportion of cropland treated with insecticides was statistically significant for the Prairie (OD = 0.57, $p < 0.001$) and Central (OD = 0.48, $p < 0.001$) regions, but it was not significant for the Atlantic and Pacific regions (**Table 2** and **Figure 2B**). Furthermore, the increase was more prominent for the last two census years (2011 and 2016; **Figure 2B**), and it is spatially evident for areas in southern Ontario (Central region), Saskatchewan and Manitoba (Prairie region), and Peace River area (Pacific region; **Figure 1B**). The areas treated with insecticides increased from 2.4 million ha (8.3% of the total cropland) in 1996 to 3.9 million ha (12.4% of the total cropland) in 2016 in the Prairies and from 0.5 to 1.2 million ha (8.6–21.5% of the total cropland) in the Central region (**Table 1**).

Between 1996 and 2016, the increase in proportion of cropland treated with fungicides was statistically significant for the Prairie (OD = 0.13, $p < 0.001$), Central (OD = 0.3, $p < 0.001$), and Atlantic (OD = 0.46, $p < 0.001$) regions, and there was no significant change for the Pacific region (**Table 2** and **Figure 2C**). A sharp increase in fungicide treatment was observed in 2011 and 2016 in both Prairie and Central regions (**Figures 1C**, **2C**), where areas treated with fungicides (>20% of the total cropland) were predominately located in eastern Saskatchewan and Manitoba (Prairie region), as well as in southern Ontario (Central region; **Figure 1C**). The area in the Prairies treated with fungicides increased from 1.5 million ha (5.1% of the total cropland) in 1996 to 8.2 million ha (26.2% of the total cropland) in 2016 (**Table 1**). For the Central region, fungicide-treated area increased from 0.3 to 1 million ha (4.7–18.4% of the total cropland; **Table 1**).

Consistent data for herbicides were available for a longer time period (1981–2016) and showed that the proportion of cropland treated with herbicides increased significantly for the Prairie, Central, and Atlantic regions (**Table 2**, **Supplementary Table 3**, and **Figures 1D**, **2D**). Herbicide-treated areas significantly increased across the Prairies and southern Ontario, such that in 1981, herbicide-treated areas were less than 30% of the total farmed area, but rose to 100% of the total farmed area in 2016 (**Figure 1D** and **Supplementary Figure 1**). Similar to all other agrochemicals, herbicide-treated areas increased most dramatically in the Prairie and Central regions (**Table 1**). Between 1996 and 2016, herbicide-treated area for the Prairies grew from 12.6 to 25.3 million ha (51–81% of the total cropland), and for the Central region, this area increased from 2.32 to 3.6 million ha (43–65% of the total cropped area). The increase in cropland treated with herbicides over three decades was gradual, and it peaked with the two most recent census years for all regions (2011 and 2016; **Figures 1D**, **2D** and **Supplementary Figure 1**).

Farms and Farm Size

Between 1981 and 2016, the number of agricultural farms reporting to the Census of Agriculture decreased from 318,361 to 192,8787 farms, while cropland in these farmed areas increased from 31 to 38 million ha. Consistent with other reports (Statistics Canada, 2017), these numbers indicate that farms

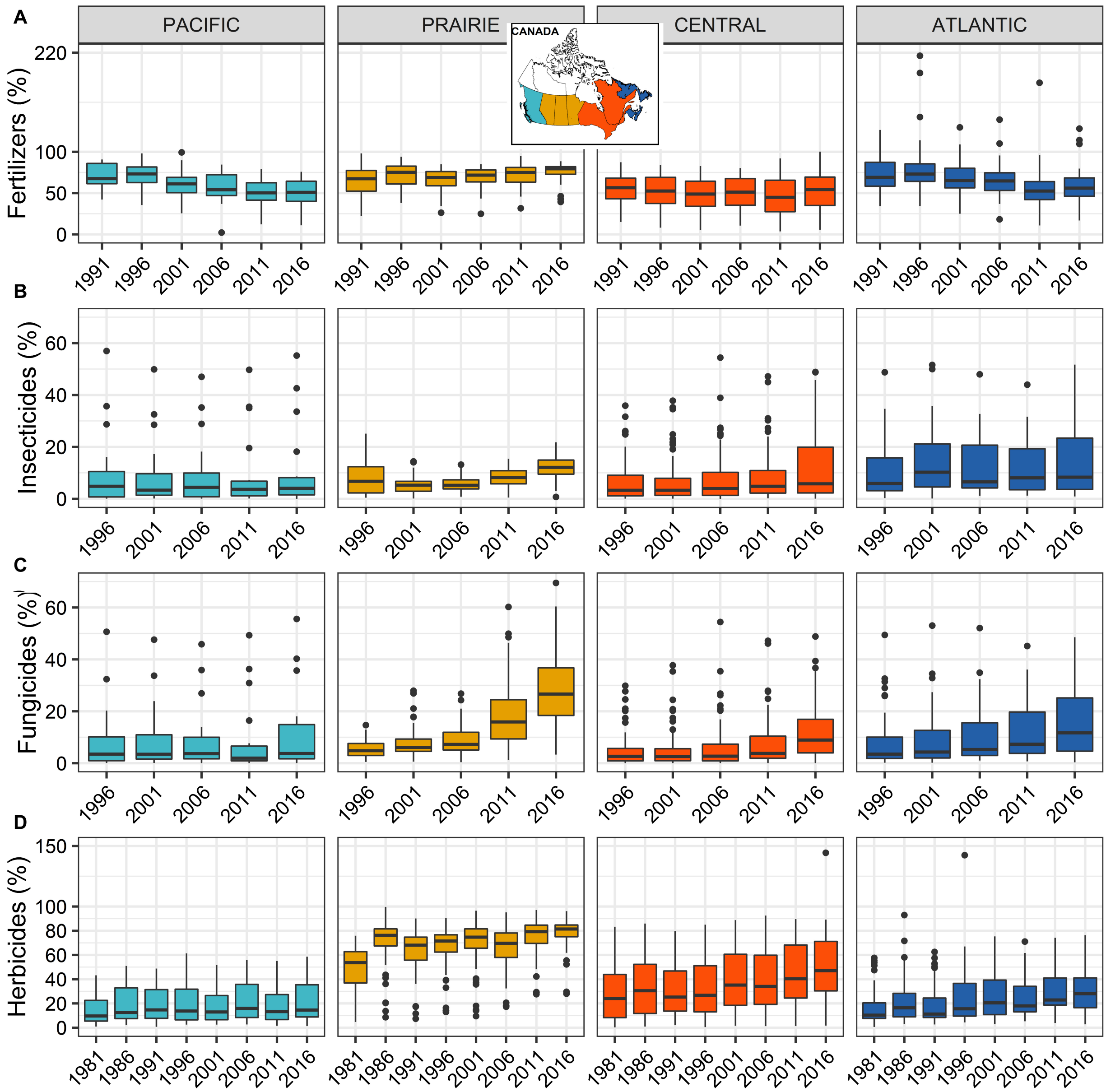


FIGURE 2 | Percent of the cropland treated with agrochemicals for each Census of Agriculture year and agricultural region in Canada. Agrochemicals include fertilizers **(A)**, insecticides **(B)**, fungicides **(C)**, and herbicides **(D)**, and the agricultural regions are ordered from west to east: British Columbia (Pacific; turquoise), Alberta, Saskatchewan and Manitoba (Prairie; orange), Ontario and Quebec (Central; red), and Nova Scotia, New Brunswick, Newfoundland/Labrador, Prince Edward Island (Atlantic; blue). Note that the date range of available data from the Census of Agriculture varies by agrochemical group: fertilizers (1991–2016), insecticides and fungicides (1996–2016), and herbicides (1981–2016). Two census division units for herbicides and 11 census division units for fertilizers reported > 100% area cropped, suggesting frequent, repeated applications in a growing season.

have become larger operations over time and that more of the farm area has been converted into cropland. Regionally, between 1981 and 2016, average farm size increased the most in the Prairies (359–602 ha), followed by Pacific (93–133 ha), Central (81–116 ha), and Atlantic (86–112 ha) regions. Most of the cropland, both historically and currently, is located in the Prairie region (24.6 million ha in 1981 to 31.2 million ha in 2016), followed by Central provinces of Ontario and Quebec (5.4 million ha in 1981 to 5.5 million ha in 2016), Pacific region (0.57 million ha in 1981 to 0.58 million ha in 2016), and the Atlantic region (0.41 million ha in 1981 to 0.4 million ha in 2016).

Cropping Patterns

We found significant changes in the proportion of crop types and their distribution over 35 years across Canada. These differences were apparent among regions (χ^2 = 312, df = 3 for cereals and pulses, χ^2 = 123, df = 2 for oilseeds and soybeans, and χ^2 = 70.6,

TABLE 2 | Pairwise contrasts between the earliest and latest Census of Agriculture year of area treated (estimated marginal means) with fertilizers, insecticides, fungicides and herbicides across Canada's four agricultural regions.

Agrochemical group	Contrast	Region	OR	SE	df	*t*-Ratio	*p*-Value
Fertilizers	1991–2016	Pacific	2.43	0.27	1,555	8.05	**<0.001**
		Prairie	0.59	0.04	1,555	−7.37	**<0.001**
		Central	1.26	0.06	1,555	5.34	**<0.001**
		Atlantic	2.26	0.20	1,555	9.07	**<0.001**
Insecticides	1996–2016	Pacific	0.98	0.16	1,176	−0.12	1.000
		Prairie	0.57	0.05	1,176	−6.12	**<0.001**
		Central	0.48	0.03	1,176	−10.80	**<0.001**
		Atlantic	0.75	0.08	1,176	−2.70	0.055
Fungicides	1996–2016	Pacific	0.74	0.12	1,132	−1.92	0.307
		Prairie	0.13	0.01	1,132	−25.35	**<0.001**
		Central	0.30	0.02	1,132	−18.01	**<0.001**
		Atlantic	0.46	0.05	1,132	−6.90	**<0.001**
Herbicides	1981–2016	Pacific	0.80	0.13	1,988	−1.40	0.858
		Prairie	0.24	0.02	1,988	−14.97	**<0.001**
		Central	0.29	0.02	1,988	−18.68	**<0.001**
		Atlantic	0.47	0.06	1,988	−6.19	**<0.001**

The earliest year was 1991 for fertilizers, 1981 for herbicides, and 1996 for both insecticides and fungicides. The latest year for all agrochemicals was 2016. The agricultural regions are ordered from west to east: British Columbia (Pacific), Alberta, Saskatchewan and Manitoba (Prairie), Ontario and Quebec (Central), and Nova Scotia, New Brunswick, Newfoundland/Labrador, Prince Edward Island (Atlantic). For each comparison, odds ratio (OR), standard error (SE), degrees of freedom (df), t-ratio, and p-values (in bold when significant at $p < 0.05$) are presented. An OR<1 indicates lower odds of occurrence in the earlier census year, and an OR>1 indicates higher odds of occurrence in the earlier census year. For full pairwise comparisons between years, see ***Supplementary Table 3****.*

df = 2 for fruits and vegetables; $p < 0.001$ for all groups), and among census years ($\chi^2 = 185$, df = 7 for cereals and pulses; $\chi^2 = 1329$, df = 7 for oilseeds and soybeans; $\chi^2 = 156$, df = 7 for fruits and vegetables; $p < 0.001$ for all groups), as well as their interactions (**Supplementary Table 2B**). There were notable regional patterns in the distribution of major crops with: (i) canola, wheat, pulses, oats, and barley mostly grown in the Prairie region; (ii) soybeans and corn almost exclusively grown in the Central region; (iii) fruits and vegetables grown in all provinces, but with the greatest area in production in the Central and Pacific regions; and (iv) potatoes predominantly grown in the Atlantic and the Prairie regions (**Figure 3** and **Supplementary Figure 2**).

Pairwise comparisons by year clearly showed that oilseeds and soybeans significantly increased between 1981 and 2016, whereas cereal production decreased over the same timeframe (**Table 3** and **Supplementary Table 4**). This change was particularly relevant for the Prairie region, where between 1981 and 2016, cereals drastically decreased (OD = 3.03, $p < 0.001$) in favor of oilseed production (OD = 0.08, $p < 0.001$; see also **Supplementary Figure 2** for spatial changes in this region). Specifically, area in wheat decreased from 60% (12 million ha) to

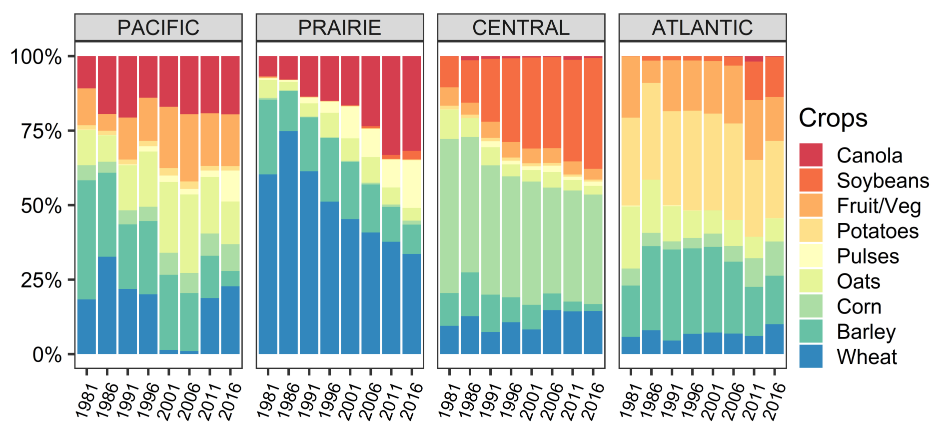


FIGURE 3 | Proportion of land planted with nine major agricultural crops for eight census years (1981–2016) for each of the four agricultural regions in Canada. The agricultural regions are ordered from west to east: British Columbia (Pacific), Alberta, Saskatchewan and Manitoba (Prairie), Ontario and Quebec (Central), and Nova Scotia, New Brunswick, Newfoundland/Labrador, Prince Edward Island (Atlantic).

TABLE 3 | Estimated marginal means of pairwise comparisons between the earliest (1981) and latest (2016) years for the area in each crop class, for four regions in Canada.

Crop class	Region	OR	SE	df	*t*-Ratio	*p*-Value
Cereals and pulses	Pacific	1.31	0.21	2,010	1.68	0.703
	Prairie	3.03	0.27	2,010	12.55	**<0.001**
	Central	1.06	0.06	2,010	0.95	0.981
	Atlantic	1.07	0.14	2,010	0.51	1.000
Oilseeds and soybeans	Prairie	0.08	0.01	1,819	−13.82	**<0.001**
	Central	0.01	0.002	1,819	−29.44	**<0.001**
	Atlantic	0.02	0.01	1,819	−9.51	**<0.001**
Fruits and vegetables	Pacific	0.79	0.26	1,570	−0.70	0.997
	Central	1.51	0.22	1,570	2.87	0.079
	Atlantic	3.33	0.85	1,570	4.72	**<0.001**

The agricultural regions are ordered from west to east: British Columbia (Pacific), Alberta, Saskatchewan and Manitoba (Prairie), Ontario and Quebec (Central), and Nova Scotia, New Brunswick, Newfoundland/Labrador, Prince Edward Island (Atlantic). For each comparison, odds ratio (OR), standard error (SE), degrees of freedom (df), t-ratio, and p-values (in bold when significant at p < 0.05) are presented. Fruits and vegetables for the Prairies and oilseeds and soybeans for the Pacific region were omitted from respective models due to insufficient occurrence of these crops over time. For full pairwise comparisons between years, see ***Supplementary Table 4****. For the spatial distribution of each crop group, see* ***Supplementary Figure 2****.*

34% (8.6 million ha) of the total cropland area, and area in barley decreased from 25% (5 million ha) to 9.9% (2.5 million ha) of the total Prairie cropland area between 1981 and 2016 (**Figure 3**). The reduction in cereals was replaced by increased planting of the oilseed, canola, which represented 6.8% (1.4 million ha) of the cropland in 1981 rising to 31.8% (8.2 million ha) in 2016 (**Figure 3**). Similarly, area in pulse crops substantially increased in the Prairie region from 0.6% (0.1 million ha) of the cropland in 1981 to 16% (4.1 million ha) in 2016 (**Figure 3**).

The change in cereal plantings was not statistically significant (OD = 1.06, p = 0.981), whereas oilseeds and soybeans areas significantly increased (OD = 0.01, $p < 0.001$) over 35 years in Central region (**Table 3**). Major increases in area in soybeans (10–37% of cropland) replaced decreasing areas of corn (52 to 36.8% of cropland), and barley (11 to 2.3% of cropland) between 1981 and 2016 (**Figure 3**). In the Pacific region, changes in area in cereals (largely barley) or fruits and vegetables were not statistically different (**Table 3**); however, the production of canola (11–20% of cropland), and fruits and vegetables (12–17% of cropland) slightly increased (**Figure 3**). In the Atlantic region, area in cereal crops did not significantly change (**Table 3**), likely due to fluctuations between different crops categorized in the same crop class, e.g., area increases for corn (5.7–11.5% of cropland) and wheat (5.7–10% of cropland), but decreases were observed for oats (20.8 to 7.8% of cropland), which were all classified as cereals. Changes between 1981 and 2016 in soybean area (0.08–13.5% of cropland) and canola area (0.01–0.23% of cropland; **Figure 3**) were responsible for the significant increase of this crop class in the Atlantic region (OD = 0.02, $p < 0.001$; **Table 3**).

DISCUSSION

Shifts in Agrochemical Use and Cropping Patterns

This is the first comprehensive pan-Canadian analysis showing that the proportion of cropland treated with agrochemicals consistently and dramatically increased across agricultural regions over recent decades, and these increases appear to be correlated with shifts in specific farming practices over time. Rising trends in farm size and cropland area, while total farm area remained constant, have been observed more broadly across North America in the last two decades (Meehan and Gratton, 2016; Spangler et al., 2020). Areas with high agrochemical applications were related to the dominant crop production areas in Canada, and they were concentrated in specific regions such as the Prairies and southern Ontario. For example, area in oilseeds in the northern and eastern part of the Prairies coincides with the region with a high proportion of land area being treated with agrochemicals for the same time period. Pesticides are heavily applied in the canola crop in the Prairies (Malaj et al., 2020). Similarly, in the Central part of Canada, soybeans and corn are the dominant crops, and they are likely driving significant increases in areas treated with agrochemicals in southern Ontario (Farm and Food Care Ontario, 2015). The number of operating farms in Canada has also decreased, while farms have become larger over time. This is consistent with other western countries (Prestele et al., 2018; Spangler et al., 2020) that have utilized economies of scale to increase production. Homogenization of agricultural commodities and practices across North America comes as a need to consolidate input investments, share information, and remain competitive in unpredictable agricultural markets (Spangler et al., 2020). At the same time, increased farm size has been related to increased pesticide use per area (i.e., insecticides), as farmers routinely spray on large fields and avoid pest management decisions that rely on field monitoring (Larsen and Noack, 2017).

Fertilizers

A high percentage of cropland in Canada in 2016 was treated with fertilizers (e.g., 65% of the cropland in Central provinces and 78% of cropland in the Prairies)—similar to a 2011 survey conducted across 20,000 Canadian farms, which reported that 69% of the agricultural areas were treated with fertilizers (Hoppe et al., 2016). The percentage of area treated with fertilizers increased for the Prairies, but not for other regions. However, considering that 82% of the cropland in Canada is located in this region, the increase is expected to be significant at a national level. In fact, mass of fertilizers applied in Canada has doubled between 1980 and 2011, notably for large farms in the Prairies (Dorff and Beaulieu, 2014). This highlights the need to report both area treated and mass of fertilizers at a national level in order to accurately quantify changes in fertilizer use. The amount of fertilizers applied in the past, cost of fertilizers and crop seed are deciding factors known to influence use (Hoppe et al., 2016). Key requirements, such as soil conditions and crop nutritional needs, are seldomly considered as only 20% of the farms in Canada carry out soil condition testing (Hoppe et al., 2016), thus increasing the likelihood of fertilizer overuse. Excess agricultural fertilizer use has been linked to greenhouse gas pollution (Park et al., 2012), eutrophication of downstream water sources, and harmful algal blooms (Schindler, 2012), with famous examples in the Lake Winnipeg (Bunting et al., 2016) and the Lake Eerie (Michalak et al., 2013) basins of Canada.

Pesticides

In recent decades, the proportion of cropland area treated with fungicides, herbicides, and insecticides has increased in all agricultural regions of Canada. The Prairie and southern Ontario regions have notably large areas treated with fungicides, insecticides, and herbicides, which appears to overlap with areas seeded with canola, cereals, and soybeans. In the Pacific and the Atlantic regions, areas planted with fruits and vegetables were almost exclusively related to areas treated with more insecticides and fungicides. Long-term, high insecticide use has been systematically related to areas producing fruits and vegetables, as they are typically more vulnerable to insect infestations than field crops (Larsen et al., 2015; Meehan and Gratton, 2016). In this pan Canadian study, we observed trends toward large areas treated with pesticides, which supports findings from local studies reporting increased agricultural pesticide sales. For example, between 2003 and 2013, Alberta reported a 48% increase (Alberta Environment and Parks, 2015) and Ontario reported a 29% increase in the amount of pesticide used (Farm and Food Care Ontario, 2015). Higher pesticide mass applications have been attributed to canola and wheat crops in the Prairie provinces (Alberta Environment and Parks, 2015; Malaj et al., 2020), soybeans and corn in Ontario (Farm and Food Care Ontario, 2015), and fruits and vegetables in southern British Columbia and Atlantic provinces (Beaulieu et al., 2005). These regional assessments reporting pesticide use by mass further validate our general findings using only a coarse metric of area treated.

Herbicide applications have undoubtedly increased in recent decades in Canada, likely due to the development and widespread adoption of herbicide-tolerant varieties of genetically modified (GM) crops, such as Roundup Ready (glyphosate resistant) and Liberty Link (glufosinate resistant) (Stringam et al., 2003; Beckie et al., 2006). Prior to GM crops, several herbicides were used to control different weed species, whereas recently, glyphosate and glufosinate herbicides have been dominantly used for a full spectrum weed control (Mamy et al., 2008). Herbicide use data reported regionally by mass show similar trends—a heavy reliance on glyphosate, e.g., 73% of the herbicide use in the Prairies (Malaj et al., 2020) and 64% of the herbicide use in Ontario (Farm and Food Care Ontario, 2015). Sales for these products also show dramatic rise, e.g., glyphosate and glufosinate increased by 230 and 995%, respectively, between 1998 and 2013 in Alberta (Alberta Environment and Parks, 2015).

The causes of increased insecticide and fungicide applications, particularly in the Prairie region, may have been related to reduced tillage (Daneshfar and Huffman, 2016; Gagnon et al., 2016; Aboukhaddour et al., 2020), or potentially as a result of farmer's preference for prophylactic control measures due to the stochastic nature of insects and diseases that threaten agricultural crops (Douglas and Tooker, 2015). Sales for fungicides steadily increased in Alberta from 2003 to 2013 by 152%, whereas insecticides fluctuated considerably from year to year as a result of changes in local pest pressures (Alberta Environment and Parks, 2015).

Fungicide and insecticide applications are likely underestimated in this analysis, due to the shift from spray treatments to seed treatments, which are generally poorly reported (Douglas and Tooker, 2015) and not captured in census surveys (Hitaj et al., 2020; Malaj et al., 2020). Considering that seeds for crops like canola, wheat, soybeans, and corn are frequently treated with insecticides and fungicides (Douglas and Tooker, 2015; Sekulic and Rempel, 2016; Malaj et al., 2020), reporting data on mass of the products used to treat seeds would be crucial in accurately determining pesticide use across Canada.

Consequences of Overreliance on Agrochemicals

Increasing trends of agrochemical applications identified in this study are of significant environmental concern. Fertilizers, in the form of nitrogen (N) and phosphorus (P), are responsible for impaired water quality through eutrophication of lakes and rivers (Michalak et al., 2013; Lassaletta et al., 2014; Bunting et al., 2016), groundwater contamination (Burow et al., 2010), and greenhouse gas production (Park et al., 2012). Similarly, frequent pesticide applications have been related to the occurrence of a wide range of pesticides in lakes (Struger et al., 2017; Metcalfe et al., 2018), rivers (Challis et al., 2018), wetlands (Main et al., 2014), and air (Yao et al., 2006). Several pesticide groups (e.g., the neonicotinoid insecticides) are known to adversly affect non-target species in terrestrial and aquatic ecosystems (Goulson, 2014; Morrissey et al., 2015). Meanwhile, many of the herbicides, fungicides, and insecticides that are in use in Canada are under regulatory review for their human and environmental safety, as there is ongoing concern about the persistence and toxicity of several current use pesticides (e.g., neonicotinoids) (Pest Management Regulatory Agency [PMRA], 2016, 2018a,b).

The increased dependency on agrochemicals in Canada comes despite efforts to reduce use of pesticides and fertilizers through integrated pest management (IPM) practices (Pimentel and Peshin, 2014), agricultural biotechnology advancements (Klümper and Qaim, 2014), and precision adjustments in fertilizer applications (Sun et al., 2016). Current increasing trends of agrochemical applications will likely continue as landscapes become more simplified due to agricultural intensification (Godfray et al., 2010; Meehan et al., 2011), agricultural expansion through land clearing (Tilman et al., 2011), and stochastic weather events as a result of climate change (Gregory et al., 2009; Olfert et al., 2016). In the light of mounting environmental concerns, current and future management practices and policies to limit agrochemical use are urgently needed to address these challenges and improve the agro-environmental performance of the sector in Canada. In addition to further promoting IPM approaches (Sun et al., 2016), regenerative agriculture (LaCanne and Lundgren, 2018) or nature-based solutions (Seddon et al., 2020) hold significant promise for systems-level solutions that not only reduce pesticide applications, but can also be cost-effective for farmers, enhance biodiversity, improve ecosystem health, combat climate change, reduce pest resistance, and enhance the resilience of croplands.

Conclusion and Future Directions

This study reveals rapid, widescale, and sustained increases in agrochemical applications across much of Canada's four agricultural regions over three decades. Dramatic shifts in agrochemical treatments and related cropping patterns were most notable for the Prairie and Central regions, but they were associated with different crop types that are unique to each region. This data-driven analysis provides clear evidence for industry and government to develop and operationalize agricultural policies and incentive programs that target agrochemicals, crops, and locations that are most in need of immediate agrochemical use reductions. Furthermore, the spatial mapping of areas where pesticides are applied provides a foundation for ongoing initiatives toward developing national and provincial pesticide monitoring schemes. While this synthesis produced useful findings and generally matches local reports, we advocate for improved publicly-available national data on agrochemical sales and use by mass at finer scales, rather than area treated, in order to closely track changes and facilitate predictive modeling into causal factors influencing agrochemical use. System-level shifts and solutions are urgently needed to change the trajectory for agricultural pesticide and fertilizer use in Canada to move toward more sustainable production practices that protect the environment, while also maintaining production yields.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. These data can be found here: Canadian Census of Agriculture https://www.statcan.gc.ca/eng/ca2016. The original datasets and R code produced from this study are available at: Federated Research Data Repository (FRDR) https://doi.org/10.20383/101.0272.

AUTHOR CONTRIBUTIONS

EM and CM conceived the study and wrote the manuscript. EM analyzed the data and produced the figures. LF collected and annotated the data and provided edits on the manuscript. All authors contributed to the article and approved the submitted version.

FUNDING

This work was supported by postdoctoral funding for EM through Global Water Futures (Prairie Water) and NSERC Discovery Grant to CM. Publication funds were provided from the University of Saskatchewan.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2020.556452/full#supplementary-material

REFERENCES

Aboukhaddour, R., Fetch, T., McCallum, B. D., Harding, M. W., Beres, B. L., and Graf, R. J. (2020). Wheat diseases on the prairies: a canadian story. *Plant Pathol.* 69, 418–432. doi: 10.1111/ppa.13147

Alberta Environment and Parks (2015). *Overview of 2013 Pesticide Sales in Alberta*. Edmonton: Land Policy Branch.

Arora, G., and Wolter, P. T. (2018). Tracking land cover change along the western edge of the US Corn Belt from 1984 through 2016 using satellite sensor data: observed trends and contributing factors. *J. Land Use Sci.* 13, 59–80. doi: 10.1080/1747423x.2018.1466001

Beaulieu, M. S., MacDonald, T., Benbrook, C., and Mineau, P. (2005). *Pesticide Use and Pest Management Practices of Canadian Apple Growers*. Agriculture and Rural Working Paper Series. Catalogue no. 21-601-M-No. 08. Ottawa, ON: Statistics Canada.

Beckie, H., Harker, K., Hall, L., Warwick, S., Légère, A., Sikkema, P., et al. (2006). A decade of herbicide-resistant crops in Canada. *Can. J. Plant Sci.* 86, 1243–1264. doi: 10.4141/P05-193

Beketov, M. A., Kefford, B. J., Schäfer, R. B., and Liess, M. (2013). Pesticides reduce regional biodiversity of stream invertebrates. *Proc. Natl. Acad. Sci. U.S.A.* 110, 11039–11043. doi: 10.1073/pnas.1305618110

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., et al. (2009). Generalized linear mixed models: a practical guide for ecology and evolution. *Trends Ecol. Evol.* 24, 127–135. doi: 10.1016/j.tree.2008.10.008

Brewer, C. A. (2006). Basic mapping principles for visualizing cancer data using geographic information systems (GIS). *Am. J. Prev. Med.* 30, S25–S36. doi: 10.1016/j.amepre.2005.09.007

Brooks, M. E., Kristensen, K., van Benthem, K. J., Magnusson, A., Berg, C. W., Nielsen, A., et al. (2017). glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *R. J.* 9, 378–400. doi: 10.32614/rj-2017-066

Bunting, L., Leavitt, P., Simpson, G., Wissel, B., Laird, K., Cumming, B., et al. (2016). Increased variability and sudden ecosystem state change in Lake Winnipeg, Canada, caused by 20th century agriculture. *Limnol. Oceanogr.* 61, 2090–2107. doi: 10.1002/lno.10355

Burow, K. R., Nolan, B. T., Rupert, M. G., and Dubrovsky, N. M. (2010). Nitrate in groundwater of the United States, 1991- 2003. *Environ. Sci. Technol.* 44, 4988–4997. doi: 10.1021/es100546y

Challis, J. K., Cuscito, L. D., Joudan, S., Luong, K. H., Knapp, C. W., Hanson, M. L., et al. (2018). Inputs, source apportionment, and transboundary transport of pesticides and other polar organic contaminants along the lower Red River, Manitoba, Canada. *Sci. Total Environ.* 635, 803–816. doi: 10.1016/j.scitotenv.2018.04.128

Clearwater, R. L., Martin, T., and Hoppe, T. (2016). *Environmental Sustainability of Canadian Agriculture: Agri-Environmental Indicator Report Series*, Report Series - Report #4. Ottawa, ON: Agriculture and Agri-food Canada.

Daneshfar, B., and Huffman, T. (2016). "Agricultural land use," in *Environmental Sustainability of Canadian Agriculture: Agri-Environmental Indicator Report Series, Report Series - Report #4*, eds R. L. Clearwater, T. Martin, and T. Hoppe (Ottawa: Agriculture and Agri-food Canada), 28–37.

Dorff, E., and Beaulieu, M. (2014). *Feeding the Soil Puts Food on Your Plate*. Working Paper Series, Catalogue no. 96-325-X - No. 004. Ottawa: Canadian Agriculture at a Glance-Statistics Canada.

Douglas, M. R., and Tooker, J. F. (2015). Large-scale deployment of seed treatments has driven rapid increase in use of neonicotinoid insecticides and preemptive pest management in US field crops. *Environ. Sci. Technol.* 49, 5088–5097. doi: 10.1021/es506141g

Douma, J. C., and Weedon, J. T. (2019). Analysing continuous proportions in ecology and evolution: a practical introduction to beta and Dirichlet regression. *Methods Ecol. Evol.* 10, 1412–1430. doi: 10.1111/2041-210x.13234

Farm and Food Care Ontario (2015). *Survey of Pesticide Use in Ontario, 2013/2014: Estimates of Pesticides Used on Field Crops and Fruit and Vegetable Crops*. Available online at: https://www.farmfoodcareon.org/ (accessed August 30, 2020).

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., et al. (2011). Solutions for a cultivated planet. *Nature* 478:337. doi: 10.1038/nature10452

Food and Agriculture Organization [FAO] (2018). *Inputs Data*. Rome: Food and Agriculture Organization of the United Nations.

Fox, J., and Weisberg, S. (2019). *An R Companion to Applied Regression*, 3rd Edn, Thousand Oaks CA: Sage.

Gagnon, P., Sheedy, C., Farenhorst, A., Cessna, A. J., Newlands, N., and McQueen, D. A. R. (2016). "Pesticides," in *Environmental Sustainability of Canadian Agriculture: Agri-Environmental Indicator Report Series, Report Series - Report #4*, eds R. L. Clearwater, T. Martin, and T. Hoppe (Ottawa, ON: Agriculture and Agri-food Canada), 153–165.

Gagnon, P., Sheedy, C., Farenhorst, A., McQueen, D. A., Cessna, A. J., and Newlands, N. K. (2014). A coupled stochastic/deterministic model to estimate the evolution of the risk of water contamination by pesticides across Canada. *Integr. Environ. Assess. Manag.* 10, 429–436. doi: 10.1002/ieam.1533

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., et al. (2010). Food security: the challenge of feeding 9 billion people. *Science* 327, 812–818. doi: 10.1126/science.1185383

Goslee, S. C. (2020). Drivers of agricultural diversity in the contiguous United States. *Front. Sustain. Food Syst.* 4:75. doi: 10.3389/fsufs.2020.00075

Goulson, D. (2014). Ecology: pesticides linked to bird declines. *Nature* 511:295. doi: 10.1038/nature13642

Goyette, J. O., Bennett, E. M., Howarth, R. W., and Maranger, R. (2016). Changes in anthropogenic nitrogen and phosphorus inputs to the St. Lawrence sub-basin over 110 years and impacts on riverine export. *Glob. Biogeochem. Cycles* 30, 1000–1014. doi: 10.1002/2016gb005384

Gregory, P. J., Johnson, S. N., Newton, A. C., and Ingram, J. S. (2009). Integrating pests and pathogens into the climate change/food security debate. *J. Exp. Bot.* 60, 2827–2838. doi: 10.1093/jxb/erp080

Harrison, X. A., Donaldson, L., Correa-Cano, M. E., Evans, J., Fisher, D. N., Goodwin, C. E., et al. (2018). A brief introduction to mixed effects modelling and multi-model inference in ecology. *PeerJ* 6:e4794. doi: 10.7717/peerj.4794

Hartig, F. (2020). *DHARMa: Residual Diagnostics for Hierarchical (Multi-Level / Mixed) Regression Models. R Package Version 0.3.2.0*. Available online at: https://CRAN.R-project.org/package=DHARMa (accessed June 6, 2020).

Hitaj, C., Smith, D. J., Code, A., Wechsler, S., Esker, P. D., and Douglas, M. R. (2020). Sowing uncertainty: what we do and don't know about the planting of pesticide-treated seed. *Bioscience* 70, 390–403. doi: 10.1093/biosci/biaa019

Hoppe, T., Haak, D., and Hewitt, J. (2016). "Farm environmental management," in *Environmental Sustainability of Canadian Agriculture: Agri-Environmental Indicator Report Series, Report Series - Report #4*, eds R. L. Clearwater, T. Martin, and T. Hoppe (Ottawa, ON: Agriculture and Agri-food Canada), 38–52.

Howarth, R. W. (2008). Coastal nitrogen pollution: a review of sources and trends globally and regionally. *Harmful Algae* 8, 14–20. doi: 10.1016/j.hal.2008.08.015

Klümper, W., and Qaim, M. (2014). A meta-analysis of the impacts of genetically modified crops. *PLoS One* 9:e111629. doi: 10.1371/journal.pone.0111629

LaCanne, C. E., and Lundgren, J. G. (2018). Regenerative agriculture: merging farming and natural resource conservation profitably. *PeerJ* 6:e4428. doi: 10.7717/peerj.4428

Larsen, A. E., Gaines, S. D., and Deschênes, O. (2015). Spatiotemporal variation in the relationship between landscape simplification and insecticide use. *Ecol. Appl.* 25, 1976–1983. doi: 10.1890/14-1283.1

Larsen, A. E., and Noack, F. (2017). Identifying the landscape drivers of agricultural insecticide use leveraging evidence from 100,000 fields. *Proc. Natl. Acad. Sci. U.S.A.* 114, 5473–5478. doi: 10.1073/pnas.1620674114

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., and Garnier, J. (2014). 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. *Environ. Res. Lett.* 9:105011. doi: 10.1088/1748-9326/9/10/105011

Lenth, R. (2019). *emmeans: Estimated Marginal Means, aka Least-Squares Means. R Package Version 1.4.3.01*. Available online at: https://CRAN.R-project.org/package=emmeans (accessed June 6, 2020).

Main, A. R., Headley, J. V., Peru, K. M., Michel, N. L., Cessna, A. J., and Morrissey, C. A. (2014). Widespread use and frequent detection of neonicotinoid insecticides in wetlands of Canada's prairie pothole region. *PLoS One* 9:e92821. doi: 10.1371/journal.pone.092821

Malaj, E., Liber, K., and Morrissey, C. (2020). Spatial distribution of agricultural pesticide use and predicted wetland exposure in the Canadian Prairie Pothole Region. *Sci. Total Environ.* 718:134765. doi: 10.1016/j.scitotenv.2019.134765

Mamy, L., Gabrielle, B., and Barriuso, E. (2008). Measurement and modelling of glyphosate fate compared with that of herbicides replaced as a result of the introduction of glyphosate-resistant oilseed rape. *Pest Manag. Sci.* 64, 262–275. doi: 10.1002/ps.1519

Meehan, T. D., and Gratton, C. (2016). A landscape view of agricultural insecticide use across the conterminous US from 1997 through 2012. *PLoS One* 11:e0166724. doi: 10.1371/journal.pone.0166724

Meehan, T. D., Werling, B. P., Landis, D. A., and Gratton, C. (2011). Agricultural landscape simplification and insecticide use in the Midwestern United States. *Proc. Natl. Acad. Sci. U.S.A.* 108, 11500–11505. doi: 10.1073/pnas.1100751108

Metcalfe, C. D., Helm, P., Paterson, G., Kaltenecker, G., Murray, C., Nowierski, M., et al. (2018). Pesticides related to land use in watersheds of the Great Lakes basin. *Sci. Total Environ.* 648, 681–692. doi: 10.1016/j.scitotenv.2018.08.169

Michalak, A. M., Anderson, E. J., Beletsky, D., Boland, S., Bosch, N. S., Bridgeman, T. B., et al. (2013). Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proc. Natl. Acad. Sci. U.S.A.* 110, 6448–6452. doi: 10.1073/pnas.1216006110

Mineau, P., and Whiteside, M. (2013). Pesticide acute toxicity is a better correlate of US grassland bird declines than agricultural intensification. *PLoS One* 8:e57457. doi: 10.1371/journal.pone.057457

Morrissey, C. A., Mineau, P., Devries, J. H., Sanchez-Bayo, F., Liess, M., Cavallaro, M. C., et al. (2015). Neonicotinoid contamination of global surface waters and associated risk to aquatic invertebrates: a review. *Environ. Int.* 74, 291–303. doi: 10.1016/j.envint.2014.10.024

Olfert, O., Weiss, R., and Elliott, R. (2016). Bioclimatic approach to assessing the potential impact of climate change on wheat midge (Diptera: Cecidomyiidae) in North America. *Can. Entomol.* 148, 52–67. doi: 10.4039/tce.2015.40

Ontario Council of University Libraries [OCUL] (2020). *ODESI: Ontario Data Documentation, Extraction Service and Infrastructure*. Available online at: https://odesi.ca (accessed May 27, 2020).

Park, S., Croteau, P., Boering, K., Etheridge, D., Ferretti, D., Fraser, P., et al. (2012). Trends and seasonal cycles in the isotopic composition of nitrous oxide since 1940. *Nat. Geosci.* 5, 261–265. doi: 10.1038/ngeo1421

Pastor, A., Palazzo, A., Havlik, P., Biemans, H., Wada, Y., Obersteiner, M., et al. (2019). The global nexus of food-trade-water sustaining environmental flows by 2050. *Nat. Sustain.* 2, 499–507. doi: 10.1038/s41893-019-0287-1

Pest Management Regulatory Agency [PMRA] (2016). *Proposed Re-Evaluation Decision for Imidacloprid*. Ottawa, ON: Health Canada.

Pest Management Regulatory Agency [PMRA] (2018a). *Special Review of Clothianidin Risk to Aquatic Invertebrates: Proposed Decision for Consultation*. Ottawa, ON: Health Canada.

Pest Management Regulatory Agency [PMRA] (2018b). *Special Review of Thiamethoxam Risk to Aquatic Invertebrates: Proposed Decision for Consultation*. Ottawa, ON: Health Canada.

Pimentel, D., and Peshin, R. (2014). *Integrated Pest Management: Pesticide Problems*. Berlin: Springer Science & Business Media.

Prestele, R., Hirsch, A. L., Davin, E. L., Seneviratne, S. I., and Verburg, P. H. (2018). A spatially explicit representation of conservation agriculture for application in global change studies. *Glob. Chang. Biol.* 24, 4038–4053. doi: 10.1111/gcb.14307

R Core Team (2019). *R: A Language and Environment for Statistical Computing. R Version 3.6.1*. Vienna: R Foundation for Statistical Computing.

Robertson, D. (1993). "Cell suppression at statistics Canada," in *Proceedings of the Annual Research Conference*, Suitland, MD.

Ryberg, K. R., and Gilliom, R. J. (2015). Trends in pesticide concentrations and use for major rivers of the United States. *Sci. Total Environ.* 538, 431–444. doi: 10.1016/j.scitotenv.2015.06.095

Sánchez-Bayo, F., and Wyckhuys, K. A. (2019). Worldwide decline of the entomofauna: a review of its drivers. *Biol. Conserv.* 232, 8–27. doi: 10.1016/j.biocon.2019.01.020

Sarkar, S. F., Poon, J. S., Lepage, E., Bilecki, L., and Girard, B. (2018). Enabling a sustainable and prosperous future through science and innovation in the bioeconomy at agriculture and Agri-Food Canada. *New Biotechnol.* 40, 70–75. doi: 10.1016/j.nbt.2017.04.001

Schindler, D. W. (2012). The dilemma of controlling cultural eutrophication of lakes. *Proc. R. Soc. B* 279, 4322–4333. doi: 10.1098/rspb.2012.1032

Seddon, N., Chausson, A., Berry, P., Girardin, C. A., Smith, A., and Turner, B. (2020). Understanding the value and limits of nature-based solutions to climate change and other global challenges. *Philos. Trans. R. Soc. B* 375:20190120. doi: 10.1098/rstb.2019.0120

Sekulic, G., and Rempel, C. B. (2016). Evaluating the role of seed treatments in canola/oilseed rape production: integrated pest management, pollinator health, and biodiversity. *Plants* 5:32. doi: 10.3390/plants5030032

Spangler, K., Burchfield, E. K., and Schumacher, B. (2020). Past and current dynamics of U.S. agricultural land use and policy. *Front. Sustain. Food Syst.* 4:98. doi: 10.3389/fsufs.2020.00098

Stanton, R., Morrissey, C., and Clark, R. (2018). Analysis of trends and agricultural drivers of farmland bird declines in North America: a review. *Agric. Ecosyst. Environ.* 254, 244–254. doi: 10.1016/j.agee.2017.11.028

Statistics Canada (2016). *Census of Agriculture*. Ottawa, ON: Statistics Canada.

Statistics Canada (2017). *A Portrait of a 21st Century Agricultural Operation. Farm and Farm Operator Data*. Available online at: https://www150.statcan.gc.ca/n1/pub/95-640-x/2016001/article/14811-eng.htm (accessed September 28, 2020).

Stringam, G., Ripley, V., Love, H., and Mitchell, A. (2003). Transgenic herbicide tolerant canola—the Canadian experience. *Crop Sci.* 43, 1590–1593. doi: 10.2135/cropsci2003.1590

Struger, J., Grabuski, J., Cagampan, S., Sverko, E., McGoldrick, D., and Marvin, C. H. (2017). Factors influencing the occurrence and distribution of neonicotinoid insecticides in surface waters of southern Ontario, Canada. *Chemosphere* 169, 516–523. doi: 10.1016/j.chemosphere.2016.11.036

Sun, S., Delgado, M. S., and Sesmero, J. P. (2016). Dynamic adjustment in agricultural practices to economic incentives aiming to decrease fertilizer application. *J. Environ. Manag.* 177, 192–201. doi: 10.1016/j.jenvman.2016.04.002

Tilman, D., Balzer, C., Hill, J., and Befort, B. L. (2011). Global food demand and the sustainable intensification of agriculture. *Proc. Natl. Acad. Sci. U.S.A.* 108, 20260–20264. doi: 10.1073/pnas.1116437108

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., and Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature* 418, 671–677. doi: 10.1038/nature01014

Wickham, H. (2017a). *ggplot2: Elegant Graphics for Data Analysis*. Available online at: https://CRAN.R-project.org/package=tidyverse (accessed June 6, 2020).

Wickham, H. (2017b). *Tidyverse: Easily Install and Load the 'Tidyverse'. R Package Version 1.2.1*. Available online at: https://CRAN.R-project.org/package=tidyverse (accessed June 6, 2020).

Yao, Y., Tuduri, L., Harner, T., Blanchard, P., Waite, D., Poissant, L., et al. (2006). Spatial and temporal distribution of pesticide air concentrations in Canadian agricultural regions. *Atmos. Environ.* 40, 4339–4351. doi: 10.1016/j.atmosenv.2006.03.039

Yu, C., Huang, X., Chen, H., Godfray, H. C. J., Wright, J. S., Hall, J. W., et al. (2019). Managing nitrogen to restore water quality in China. *Nature* 567, 516–520. doi: 10.1038/s41586-019-1001-1

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
published: 02 March 2021
doi: 10.3389/fsufs.2021.541590

Ecosystem Services in Working Lands of the Southeastern USA

Alisa W. Coffin[1]*, *Vivienne Sclater*[2], *Hilary Swain*[2], *Guillermo E. Ponce-Campos*[3] and *Lynne Seymour*[4]

[1] United States Department of Agriculture-Agricultural Research Service, Southeast Watershed Research Laboratory, Tifton, GA, United States, [2] Archbold Biological Station, Venus, FL, United States, [3] School of Natural Resources and the Environment, The University of Arizona, Tucson, AZ, United States, [4] Department of Statistics, The University of Georgia, Athens, GA, United States

Agriculture and natural systems interweave in the southeastern US, including Florida, Georgia, and Alabama, where topographic, edaphic, hydrologic, and climatic gradients form nuanced landscapes. These are largely working lands under private control, comprising mosaics of timberlands, grazinglands, and croplands. According to the "ecosystem services" framework, these landscapes are multifunctional. Generally, working lands are highly valued for their provisioning services, and to some degree cultural services, while regulating and supporting services are harder to quantify and less appreciated. Trade-offs and synergies exist among these services. Regional ecological assessments tend to broadly paint working lands as low value for regulating and supporting services. But this generalization fails to consider the complexity and tight spatial coupling of land uses and land covers evident in such regions. The challenge of evaluating multifunctionality and ecosystem services is that they are not spatially concordant. While there are significant acreages of natural systems embedded in southeastern working lands, their spatial characteristics influence the balance of tradeoffs between ecosystem services at differing scales. To better understand this, we examined the configuration of working lands in the southeastern US by comparing indicators of ecosystem services at multiple scales. Indicators included measurements of net primary production (provisioning), agricultural Nitrogen runoff (regulating), habitat measured at three levels of land use intensity, and biodiversity (supporting). We utilized a hydrographic and ecoregional framework to partition the study region. We compared indicators aggregated at differing scales, ranging from broad ecoregions to local landscapes focused on the USDA Long-Term Agroecosystem Research (LTAR) Network sites in Florida and Georgia. Subregions of the southeastern US differ markedly in contributions to overall ecosystem services. Provisioning services, characterized by production indicators, were very high in northern subregions of Georgia, while supporting services, characterized by habitat and biodiversity indicators, were notably higher in smaller subregions of Florida. For supporting services, the combined contributions of low intensity working lands with embedded natural systems made a critical difference in their regional evaluation. This analysis demonstrated how the inclusion of working lands combined with examining these at different scales shifted our understanding of ecosystem services trade-offs and synergies in the southeastern United States.

Keywords: working lands, ecosystem services, trade-off, land use, USDA-LTAR, Southeast USA

OPEN ACCESS

Edited by:
Johann G. Zaller, University of Natural Resources and Life Sciences Vienna, Austria

Reviewed by:
Seth Anders Spawn, University of Wisconsin-Madison, United States
Jennifer Costanza, United States Forest Service (USDA), United States

***Correspondence:**
Alisa W. Coffin
alisa.coffin@usda.gov

Specialty section:
This article was submitted to Agroecology and Ecosystem Services, a section of the journal Frontiers in Sustainable Food Systems

Received: *09 March 2020*
Accepted: *07 January 2021*
Published: *02 March 2021*

Citation:
Coffin AW, Sclater V, Swain H, Ponce-Campos GE and Seymour L (2021) Ecosystem Services in Working Lands of the Southeastern USA. Front. Sustain. Food Syst. 5:541590. doi: 10.3389/fsufs.2021.541590

INTRODUCTION

In the southeastern United States (southeastern US, or Southeast), including Georgia, Florida, and Alabama, natural systems are largely embedded and tightly coupled with more intensive land uses: timber harvesting is common in "natural" upland and riparian areas that also function as important habitat; pastures often include ponds and wetlands; and agricultural fields, irrigated and dry land, provide open foraging sites for wildlife inhabiting adjacent grassed and forested riparian areas. While the southeastern US lacks the extensive tracts of federal protected lands found in many western states, the gradients of land use intensity and the heterogeneity of the southeastern landscape is in stark contrast to the uninterrupted tilled croplands of the upper mid-west. Southeastern "working landscapes," with less intensive land uses and embedded natural areas provide an array of ecosystem services including provisioning, regulating, supporting and cultural (Millennium Ecosystem Assessment, 2005). Characterization of the values and dynamics of these ecosystem services provides insight into an understanding of how to accomplish sustainable intensification of US agriculture. This endeavor is critical to meeting the production demands of future populations while conserving soil, water and biological resources on working lands (Kleinman et al., 2018; Spiegal et al., 2018), work that is being undertaken at a national scale by the US Department of Agriculture's (USDA) Long-Term Agroecosystem Research (LTAR) Network.

Conservation of species, natural habitats, and the protection of ecosystem processes in the US has typically focused on parks and other protected areas, emphasizing the national inventory of terrestrial and marine protected areas dedicated to the preservation of biological diversity, as well as other natural, recreation and cultural uses (USGS Gap Analysis Project, 2018). Assessment of ecosystem services from these protected areas is biased toward infertile soils, extreme climates, and mountainous regions (Knight and Cowling, 2007). In contrast, lands devoted to agriculture and silviculture, generally associated with fertile soils and at lower elevations, have often been viewed as antithetical to conservation. Although agricultural ecosystems in the US are recognized as providing a variety of ecosystem services, such as soil and water quality, carbon sequestration, biodiversity and cultural, they are also depicted as intensive agro-industrial operations with numerous disservices including habitat loss, nutrient runoff, sedimentation of waterways, greenhouse gas emissions, and extensive pesticide use (Power, 2010; Lark et al., 2020). National conservation assessments such as the GAP/LANDFIRE National Terrestrial Ecosystems address detailed vegetation and land cover patterns for the conterminous United States (CONUS) but often minimize analyses of the conservation values of agricultural land uses, both the less intensive uses, such as grazing lands, or the more intensive croplands (e.g., Pearlstine et al., 2002). In contrast, in the European Union, multifunctional agriculture and human modified working lands have long been viewed as contributing to the protection of the environment and the sustained vitality of rural areas (e.g., Burrell, 2001), and the provision of "landscape amenities" produced by agriculture is important to value (Vanslembrouck and Van Huylenbroeck, 2005).

Here we argue, in line with Robertson et al. (2014), that the landscape context within which agricultural lands lie matters a lot: intrinsic services provided by working lands (from low to high intensity) are tightly coupled with the multifunctional ecosystem services of embedded and surrounding natural areas. Regionally, forested upland and riparian areas lying between agricultural fields provide regulating services by reducing loads in nutrient rich runoff. Regulating services are coincident with supporting services, such as habitat essential to pollinators that, in turn, improve local production–a coupled synergistic relationship (Robertson et al., 2014). Less intensive land uses for cattle production, both rangeland and pastures, also provide wildlife habitat and regulating services of fire and carbon sequestration (Fargione et al., 2018; Sanderson et al., 2020). However, while the land mosaic of low intensity agriculture may provide services, existing configurations of agricultural landscapes are often insufficient to effectively buffer the effects of intensive land uses, as evidenced by the increasing levels of pollution and hypoxia in downstream coastal areas, and the legacy of past fertilizer use in pasture soils that still results in downstream nutrient loading (Zhang et al., 2007; Rabalais et al., 2010; Swain et al., 2013). Increasing land area for regulating and supporting services might help solve regional environmental problems, but at the expense of land for provisioning services, which underpin regional economies. This creates a dilemma of resolving tradeoffs between competing land uses. Furthermore, the perceived need to barter between production and other environmental services invokes a win-lose scenario, which has been widely recognized to oversimplify the challenge of balancing conflicting land uses such as conservation and development (De Groot et al., 2010). In this view, working landscapes of the Southeast, covering the full gradient from intensively managed to semi-natural, constitute a vast reservoir of land area, which are both part of the problem and also, part of the solution.

The challenge to manage balances of ecosystem services in working lands requires a more nuanced understanding of landscapes and ecosystems over time and space, with an adequate frame of reference to capture both the spatial and the temporal dynamics of a region. Over time, the balance of services change in response to changes in their underlying drivers such as land

Abbreviations: ABS-UF, Archbold Biological Station – University of Florida; C, Carbon; CDL, Cropland Data Layer; CONUS, conterminous United States of America; CPEco, study area within the Southern Coastal Plain Level III Ecoregion (ecoregion); CV, coefficient of variation; FL-HUC8s, Florida study area comprising HUC8 watersheds (regional basins); FL-HUC10s, Florida study area comprising HUC10 watersheds (local watersheds); LTAR Core area; FG, fragmentation geometry; G1|G2, species that are Imperiled (NatureServe) or Listed (US Endangered Species Act); GA-HUC8s, Georgia study area comprising HUC8 watersheds (regional basins); GA-HUC10s, Georgia study area comprising HUC10 watersheds (local watersheds); LTAR Core area; GACP, Gulf Atlantic Coastal Plain; GIS, geographic information system; GPP, gross primary production; HUC, hydrologic unit code; LREW, Little River Experimental Watershed; LTAR, Long-Term Agroecosystem Research; N, Nitrogen; NCED, National Conservation Easement Database; NLCD, National Land Cover Database; NPP, net primary production; PAD-US, Protected Area Database of the U.S.; SE, study area within the Southeast; SPEco, study area within the Southeastern Plains Level III Ecoregion (ecoregion).

use (Sohl et al., 2010), which has been well-documented for the Southeast (Southworth et al., 2006; Drummond et al., 2015). Spatially, we conceptualize the dynamics among services in the Southeast as not unlike the fictional "pushmi-pullyu" character, sporting two heads on either end of its body (Lofting, 1920), in which coupled ecosystem services of some areas "push" (provide positive services or benefits) while others "pull" (essentially disservices) in a dynamic interplay of trade-offs and synergies. An example of this might be hydrological restoration of seasonal wetlands in grazed pastures in Florida, which "pushes" greater biodiversity of plants, fish and frogs, although it "pulls" or reduces yields of more nutritional forage grasses (Boughton et al., 2019) and increases natural ecosystem emissions of methane, a potent greenhouse gas (Chamberlain et al., 2016). In a cropping system, this could be visualized as a naturalized buffer area that provides habitat for insect pollinators and natural enemies (Xavier et al., 2017), while also serving a reservoir for "weed" species.

The challenges of characterizing tradeoffs and synergies among ecosystem services in working lands also involves questions of landscape structure and scale (Forman, 1995). Landscape ecological research focuses heavily on the effects of habitat loss/habitat fragmentation on biodiversity [see review by Fahrig (2019)], though less attention is paid to effects of landscape structure on other regulating and supporting services. There is also the question of landscape scale at which a benefit or a cost accrues. For example, a benefit such as forage production might accrue at the scale of the field/pasture or the ranch/farm (the enterprise) but can incur environmental costs locally, at a regional or downstream scale, or, in the case of greenhouse gas emissions, the global scale (Swain et al., 2013; Heffernan et al., 2014).

Essentially, trade-offs and synergies in ecosystem services occur within and across all landscapes, including working lands, and understanding them requires their alignment by selecting services that can be measured consistently at all scales, and identifying a useful grain for aggregating at meaningful common scales. In this research, we address these questions for working lands in the southeastern US: What are the ecosystem services associated with working lands? How do characteristics and variability of ecosystem services change as the focus is shifted from one spatial scale to the next? What are the tradeoffs among ecosystem services and does the nature of the tradeoffs change with scale? To respond, we conceptualized pairwise relationships among provisioning, regulating, and supporting services at various scales (**Figure 1**, **Supplementary Table 1.1**). In general, we expect to see negative relationships among provisioning and both supporting and regulating services, while a synergistic relationship is expected among regulating and supporting services, and that the strength of these relationships will change depending on scale. The objectives of this study were, first, to characterize, using descriptive statistics, the ecosystem services associated with working lands, aggregated at multiple scales. Second, we evaluated the tradeoffs and synergies among ecosystem services and compared the observed pairwise relationships with our hypothesized concept. Third, we evaluated whether the ecosystem services measured in the vicinity of two of the USDA LTAR Network research sites (in Georgia and Florida) were representative of their locales, their regions, and of the southeastern US.

MATERIALS AND METHODS

Study Area

This study compares ecosystem services documented at multiple scales in the southeastern US, from local to regional scales (**Figure 2**). At its broadest extent, the study area includes coastal plain regions extending from southern Virginia to southern Florida and west from coastal Georgia to the eastern bank of the Mississippi River, and into the alluvial plains in western Tennessee. The study areas associated with this research pertained to hierarchical spatial frameworks which allowed us to scale up measurements. Scaling-up started with the local scale and then moved up in area to regional frameworks in four steps. This scaling process resulted in a set of seven areas of interest, corresponding to regional spatial frameworks described below, whose locations were driven by two of the 18 USDA LTAR Network sites where we have field measurements.

Long-Term Agroecosystem Research (LTAR) Network Sites

For detailed local scale comparisons, we used two focal areas with comprehensive long-term measurements of production and other ecosystems services. First, the 4,200-ha Buck Island Ranch, managed by Archbold Biological Station, is one part of the ∼12,000 ha Archbold-University of Florida LTAR site, referred to here as the ABS-UF. Buck Island Ranch lies within the beef cattle grazing lands of south-central Florida. Second, the 334 km^2 USDA-ARS Little River Experimental Watershed (LREW) instrumented by USDA for agricultural watershed experimentation and monitoring since 1968 (Bosch et al., 2007), lies at the heart of the Gulf Atlantic Coast Coastal Plain LTAR site south central Georgia. This site, comprised primarily of croplands, rivers and streams, and pine forests, is referred to here as the GACP.

Regional Frameworks

This analysis required us to summarize and compare data across the entire region and among subregions. To accomplish this, we used boundaries that describe hydrological characteristics and, at broader scales, ecological characteristics. We used the hierarchical Hydrologic Unit Code (HUC) system to establish the grain of our analysis (USGS, 2015) summarizing data from small area HUC-12 units (referred to here as HUC-12s), our most basic unit of analysis, within a grouping of HUC-10 units, and again within a larger area of several HUC-8 units. The number and area of HUC-12s incorporated into each scale step is given in **Figure 2A**. The HUC system is well-established and, for the scale of our analysis the HUC-12s provided a stable, consistent spatial framework allowing us to compare areas of increasing sizes, while maintaining consistency as we moved up and down the spatial hierarchy. At each scale increment, measurements from incorporated HUC-12s were summarized using simple descriptive statistics of central tendency and variability.

The first scale was conceptualized as an "LTAR Core" area, an area of local watersheds defined as the area of intersecting

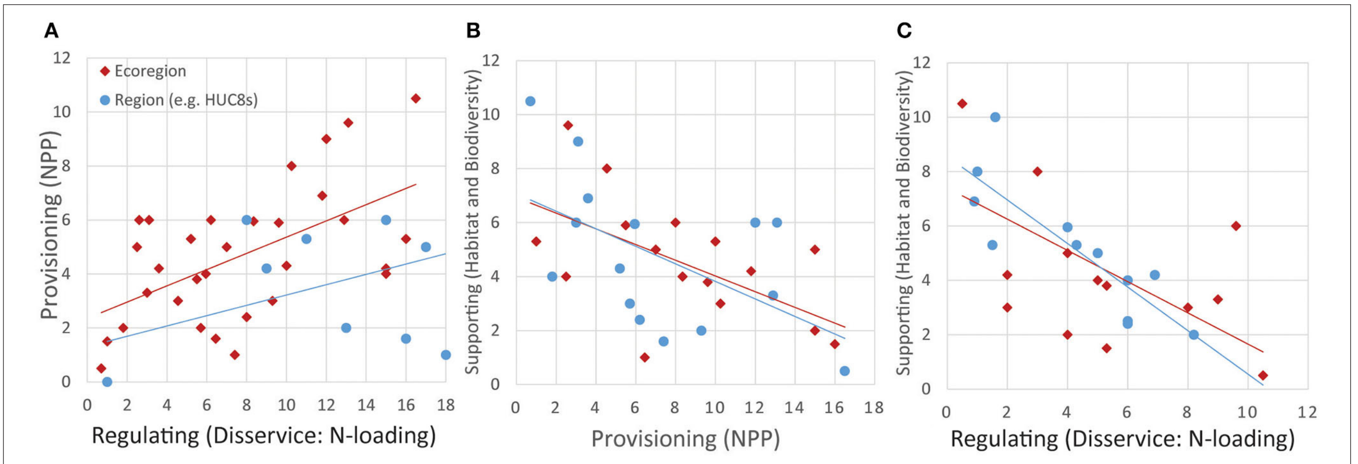


FIGURE 1 | Hypothesized relationships, displayed as scatterplots and linear trendlines, among provisioning, regulating and supporting ecosystem services in the southeastern US, at ecoregion and regional (e.g., HUC-8) scales. Regulating services are represented as a disservice, e.g., nitrogen (N) loading of aquatic systems. Provisioning services are represented as net primary production (NPP). Supporting services are represented as measures of habitat and biodiversity. Ecoregional values (in red) were simulated as a bivariate distribution, and regional values were a random selection of the ecoregion. **(A)** Hypothesized positive relationship between regulating (disservices) and provisioning services at ecoregional (red) and regional (blue) scales. **(B)** Hypothesized negative relationship between provisioning and supporting services at ecoregional (red) and regional (blue) scales. **(C)** Hypothesized negative relationship between regulating and supporting services at ecoregional (red) and regional (blue) scales.

HUC-10 basins that include the GACP or ABS-UF sites, and we referred to as GA-HUC10s and FL-HUC10s, respectively. While we did not attempt to scale point measurements from either ABS-UF or GACP to these core areas, which would have involved an extensive amount of field verification, we used some local measures at these LTAR sites to verify the range of values observed in the HUC-level data.

For the second, regional scale, we selected a series of related HUC-8 regional basins that included our LTAR Core areas, but also related to a single larger HUC-6 level basin, and we referred to as GA-HUC8s and FL-HUC8s. In Georgia, the selected regional basins were all within the Suwannee Basin (HUC-6: 031102), and in Florida, the HUC-8s all pertain to the Kisimmee Basin (HUC-6: 030901). The rationale for study areas bounded by larger hierarchical units at this step, was to maintain a level of consistency in the assumptions underlying our analysis of ecosystem services and disservices. Because these measurement units relate to hydrology, the factor driving our decision to nest the smaller HUC-10 and HUC-8 study areas within a single larger HUC-6 region related to limiting the introduction of confounding issues of broader-scale cross-basin watershed dynamics.

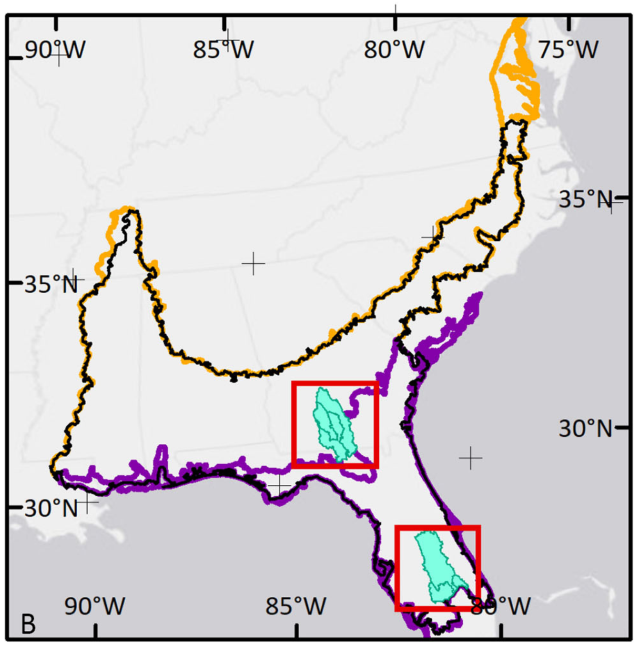
The third scale involved a jump to the ecoregion level including most of the Southeastern Plains (SPEco), and the Southern Coastal Plain (CPEco) Level III ecoregions (Omernik and Griffith, 2014). A final, and fourth scale was the amalgamation of both ecoregions into a unifying southeastern US (SE) mega-region. At these larger regions, we initially defined the study area boundaries as the collection of HUC-12s which intersected the ecoregions. Further refinement led to a final subset of HUC-12s across the entire southeast, comprising 4,596 units. The entire list of HUC-12s used is provided in **Supplementary Material 2—Data Table**, and their area is identified by the black outline in **Figure 2B**. In some cases, such as the coastal plain of South Carolina and the rolling coastal plain of Virginia (north of the James River), HUC-12s were excluded from the SE, SPEco, and CPEco study areas because we judged them to be either highly uncharacteristic of our LTAR sites, or they were better represented by another LTAR site (e.g., Lower Chesapeake Bay LTAR). Units that intersected the boundary of the ecoregion were included if their centroids were within the ecoregion, but certain HUC-12s were excluded from the analysis if their land cover was mostly open water, such as large lakes or barrier islands.

Data

To accomplish this work, we identified several indicators of provisioning, regulating and supporting ecosystem services (or disservices), for which existing data were available at both the scale and grain required (**Table 1**, **Supplementary Table 1.1**). Source data for this analysis included published data available for download from public repositories (**Supplementary Table 1.2**), which we processed using geographic information systems (GIS; Esri, ArcGIS Pro 1.X - 2.X, and ArcGIS Desktop 10.6-7, Advanced licenses), and the Google Earth Engine (GEE) data and analysis platform. Boundary maps for contrasting regional analyses were created from the intersection of the hydrologic basin framework with the ecoregional framework as noted. Data describing land cover and ecosystem services consisted of gridded datasets produced and published in previous research or as operational land cover products. Basic criteria for these data included: scope—datasets had to be available for the CONUS, or for the entire SE mega-region; grain—datasets resolution had to

Region, scale, color	HUC 12 units		
	N	m area (km²)	st dev (km²)
SE, megaregion, black	4596	95.8	53.86
CPEco, ecoregion, purple	1219	111.13	82.63
FL-HUC8s, regional, green	93	112.74	53.85
FL-HUC10s, local, blue	23	106.19	29.75
SPEco, ecoregion, gold	3377	90.29	37
GA-HUC8s, regional, green	116	93.78	34.52
GA-HUC10s, local, blue	27	80.81	25.44

A



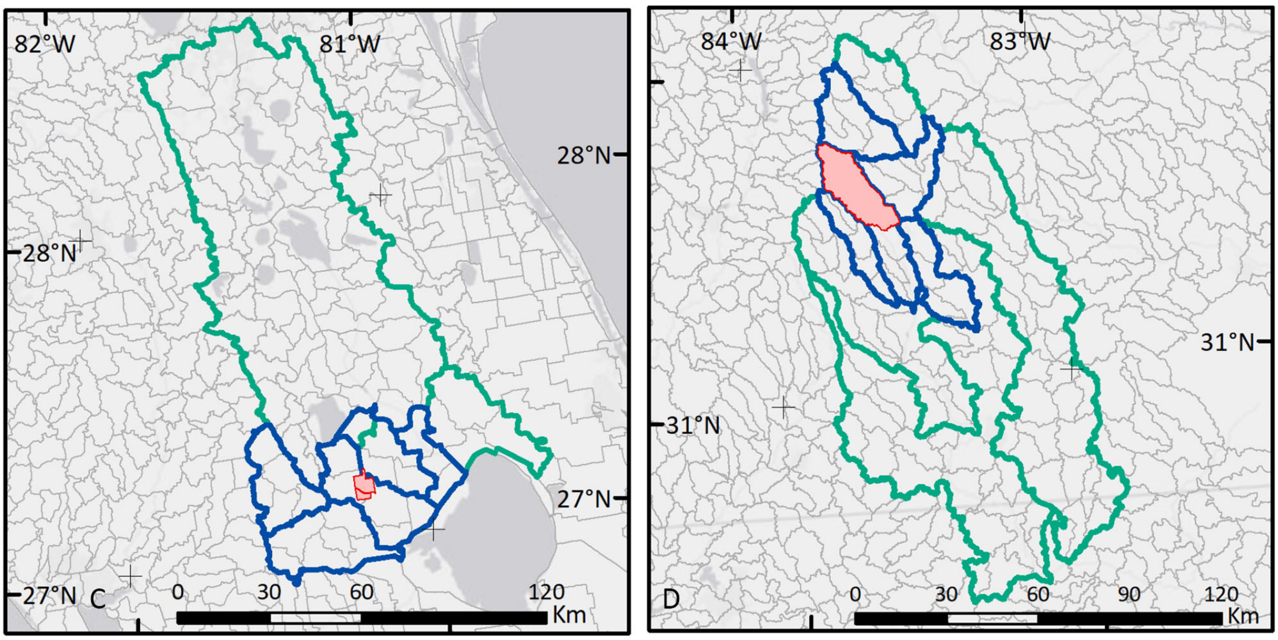


FIGURE 2 | The southeastern US showing ecoregion and study area boundaries with HUC-12 unit summary. **(A)** Table describing study area names, scale, and map color, and showing count, mean and standard deviation of HUC-12 areas within the study area boundaries. **(B)** Southeast mega-region (SE, black outline), with Southeastern Plains Ecoregion (SPEco, gold outline), and Southern Coastal Plain ecoregion (CPEco, purple outline). **(C)** Florida study areas—FL-HUC8s (green outline), and FL-HUC10s (blue outline). **(D)** Georgia study areas—GA-HUC8s (green outline), and GA-HUC10s (blue outline).

be fine enough to provide estimations of land cover for areas >15 km^2, the size of the smallest HUC-12 basin included in the SE; and time period—data were fairly recent (within a decade).

Land Cover

We used land cover information to compare the overall composition of land cover types for each region, and to estimate the fragmentation in landscapes within each study area, under three different characterizations of land use. For these analyses, we used both the USGS National Land Cover Dataset (NLCD) and the USDA Cropland Data Layer (CDL) datasets to derive information about land cover and land use in the region (Boryan et al., 2011; Yang et al., 2018; Jin et al., 2019). For our initial land cover characterization, we summarized land cover values

TABLE 1 | Derived datasets used for analysis (see **Supplementary Material 1** for sources and land cover reclassification; **Supplementary Material 2—Data Table**).

Variable names	Description; Dataset name	Ecosystem service type
FG1	Fragmentation geometry 1. Proportion of natural areas in HUC12: includes only wetlands, forested areas not managed for timber production and shrublands. Fragmenting elements include all anthropogenic classes, roads, and open water (e.g., production forests, croplands, urban); Map1Nat	Supporting
FG2	Fragmentation geometry 2. Proportion of natural areas with low intensity working lands in HUC12: Combination of natural areas with low intensity working lands including production forests and areas used for hay and grazing. Fragmenting elements include developed lands, croplands, roads, and open water; Map2NatLo	Supporting
FG3	Fragmentation geometry 3. Proportion of natural areas and all working lands in HUC12: Combination of natural areas and low intensity working lands with croplands. Fragmenting elements include developed lands, roads, and open water.; Map3NatLoHi	Supporting
NPP and NPP CV	Five year average of net primary production on working lands (mean and coefficient of variation [CV]); NPPwl_2014, … NPPwl_2018	Provisioning
N	Nitrogen from surface and subsurface agricultural sources; AgriN	Regulating
TerrG1|G2 and AqG1|G2	Count per 100 km^2 of imperiled G1|G2 species (terrestrial and aquatic) summarized by HUC12; Imper_Spp	Supporting
MCL	Proportion of managed conservation lands in HUC12; MCL	Supporting

Fragmentation geometry definitions follow methods from Jaeger et al. (2008).

for each study area using the NLCD, which does not include detailed categories of crop type. For this classification, we reduced the number of classes from 15 to 12 by combining "Barren" with open and low intensity "Developed" classes and combining medium with high intensity "Developed" classes. For subsequent, more detailed analysis inferring land use, we used the CDL data to differentiate finer categories of agricultural land cover type, as described below.

Ecosystem Service Indicators

Provisioning: Net Primary Production (NPP)

Provisioning ecosystem services were evaluated using net primary production (NPP) as an indicator of provisioning services from working lands. Although provisioning services can imply much more than simply NPP (or gross primary production, GPP)—for example, some consider yield to be a provisioning service—NPP is useful to describe a measure of biological productivity across widely divergent ecosystems (Running et al., 2000). In terms of ecosystem services, terrestrial primary production is considered the foundation of provisioning services directly related to agriculture, including production of food, fuel, and fiber (Smith et al., 2012), and the main reason for selecting NPP as one of the primary datasets to support the spatial scale of this analysis. There is a constant need to estimate values of terrestrial primary production at regional to global scales, and currently this is only possible by using remote sensing-based models. The traditional components for these estimates are GPP and NPP, where GPP is considered the total amount of carbon captured by plants while NPP defined as the amount of carbon allocated in plants after accounting for autotrophic respiration (Ruimy et al., 1994; Running et al., 2000).

Robinson et al. (2018) developed the Landsat-NPP model based on the MOD17 (MODIS-MODerate-resolution Imaging Spectroradiometer) algorithm (Haberl et al., 2007) producing a high-resolution (30 m pixel size) gridded dataset of annual NPP for the CONUS. This model relies mainly on the work developed by linking GPP and NPP to the amount of solar energy absorbed by plants along with atmospheric factors. Unlike the GPP product, which is produced at 16-days temporal resolution, NPP provides annual estimates that can be incorporated seamlessly into analyses requiring annual aggregation. The annual availability of NPP estimates along with the spatial resolution (30 m) and coverage (CONUS) made this satellite-based product the main candidate as a proxy of a provisioning ecosystem service indicator. We used mean annual NPP for 2014–2018 for this analysis. Data for NPP were tabulated for each HUC-12 area in units of kg carbon (C) per square meter. A mask of working lands was used to limit extract NPP data to areas associated with agricultural production. A more detailed description of the workflow and analytical methods used to calculate values for NPP is provided in **Supplementary Material 1, 2.1**.

Regulating: Nitrogen (N) Loading

To characterize regulating services, we considered the role of pollutant load, specifically agricultural nitrogen (N), as an indicator of the disservice provided by working lands. Landscape buffers adjacent to stream corridors provide important regulating services by purifying water running off intensively used lands. Where N is an essential macronutrient for primary production, and the most common ingredient in agricultural fertilizers, it is an environmental contaminant, so its measurement in aquatic systems indicates the extent to which the adjacent lands and soils are able (or unable) to purify water. Agricultural N concentrations in a watershed can serve, therefore, as an indicator of the effectiveness of regulating services within that area, where high N concentrations indicate a level of disservice.

The Environmental Protection Agency's (EPA) EnviroAtlas provides a nationwide data source for N concentrations to be used as a regulating service indicator. EnviroAtlas mapped modeled agricultural N (2002 values) removed by surface and subsurface flow (metric ton per HUC-12). These data were related to on-the-ground measurements from the LTAR sites after converting units to kg/ha. Data from ABS-UF were acquired from 4 years of collection in a field experiment between 1998 and 2003 (excluding 2000, a drought year) (Bohlen and Villapando, 2011). Data from GACP include N loading from stream collection data published for the LREW for 1978–2014 (Bosch and Sheridan, 2007; Bosch et al., 2020).

Supporting: Landscape Structure, Biodiversity, and Habitat

Supporting ecosystem services include habitat for species and the maintenance of genetic diversity. Numerous measures of biodiversity and habitat conditions exist, which can provide some indicator for the level of supporting services in a region. For this study, we used well-known indicators of both to describe supporting services. Habitat was characterized as the amount of land protected for conservation and by using landscape ecological indices to describe the fragmentation of natural and working lands. Landscape fragmentation is caused by landscape elements that bisect patches of otherwise habitable areas (usually single land cover or habitat types), or form barriers or impediments to landscape flows (Forman, 1995), including the movement of animals. Conversely, landscape connectivity is the result of landscape elements that facilitate connections, such as the ability of animals to disperse, mate and survive. While fragmentation *per se* is often associated with higher levels of biodiversity overall (Fahrig, 2017), landscape barrier characteristics strongly influence the nature and magnitude of fragmentation effects on animal species and populations. Roads are fragmenting elements (Rytwinski and Fahrig, 2013) with particularly negative effects on large carnivores such as Florida panther (*Puma concolor coryi*) and Florida black bear (*Ursus americanus floridanus*) (Ceia-Hasse et al., 2017; Murphy et al., 2017), both of which are endemic to the Southeast. For this study, biodiversity was characterized using outputs from habitat models for large regions, supported with counts of imperiled species in smaller areas. Together, these indicators provided different facets of supporting services and allowed for a more nuanced understanding of the comparative balance of ecosystem services associated with working lands in the Southeast.

Supporting: fragmentation, and landscape structure To understand the role of working lands in providing supporting ecosystem services related to wildlife especially terrestrial vertebrates, we characterized habitat using landscape ecological metrics and the area of "connecting elements" in each HUC-12. The landscape metrics used included patch number, area weighted mean patch area, and effective mesh size (McGarigal and Marks, 1995; Jaeger, 2000). Patch number is a well-known index of landscape structure that exhibits predictable patterns over a range of scales (Wu et al., 2000), and for which, increasing patch numbers are usually associated with increasing fragmentation over time, with implications for habitat connectivity and species diversity (Forman, 1995; Fahrig, 2003). Area weighted mean patch area (AWMPA), quantifies proportional amounts of patch area, and can be used to compare the proportion of habitat types in a management unit (McGarigal and Marks, 1995). In this case, however, we used it to compare the amount of patch (or habitat) area among analysis units. Effective mesh size (m_{eff}) characterizes fragmentation in a given unit of analysis, independent of its size (Jaeger, 2000), thus allowing for cross-scale comparisons. It incorporates scenarios based on land cover types permitting comparisons of alternative scenarios and assumptions, where negative correlations have been found between the degree of fragmentation and levels of species richness (Schmiedel and Culmsee, 2016).

We used land cover (CDL) and forest management data (Marsik et al., 2017, 2018) to derive datasets that characterized three levels of landscape fragmentation (Jaeger et al., 2008), described by increasing intensities of land use. To measure fragmentation, the geometry of fragmenting elements needs to be explicitly provided by stipulating the landscape elements that form the fragmentation geometry, or FG (Jaeger et al., 2008). "Fragmenting elements" comprise classes of land cover types that cause fragmentation in a landscape by breaking apart habitable environments. Conversely, the land cover classes that are not inhospitable are "connecting elements." For example, urban areas, frequently used roads, and intensively farmed cropland are inhospitable for some species. However, the movements of other species may not be hindered by cropland, although their habitat may be fragmented by highways. Together, the spatial arrangement of fragmenting and connecting elements form the FG and are designated with a number differentiating the groupings of connecting and fragmenting elements, as in FG1, FG2, and so on. Final units of analysis included proportion of connecting elements in each HUC-12, and landscape metrics were described only for aggregated levels of analysis (i.e., -HUC10s, -HUC8s, ecoregion). A more detailed description of the workflow and analytical methods used to calculate values for landscape metrics, including the production of the FG layers is provided in **Supplementary Material 1, 2.2**.

Supporting: imperiled species Biodiversity is a key indicator of supporting services, describing, in general terms, the diversity of life that exists to support the long-term viability of populations and ecosystems, including the genetic building blocks that support livestock and cropping systems. While numerous measures of biodiversity exist, imperiled species data provide information to help gauge levels of biodiversity. EPA's EnviroAtlas provides a national map of the number of "At-Risk" species with potential habitat within each HUC-12 in the CONUS, which was used as one biodiversity indicator related to supporting services (US EPA, 2011). These data include species that are Imperiled, as defined by NatureServe (https://explorer.natureserve.org/AboutTheData/Statuses), or Listed under the US Endangered Species Act (ESA), and are indicated by the designation G1|G2. The values are based on habitat models, not wildlife counts, but could be compared with lists of known species for the ABS-UF area, and the list of G1|G2 species from the Georgia Biodiversity Portal, for the GA-HUC8s area (Georgia Department of Natural Resources, 2020). Both G1|G2 terrestrial species (EnviroAtlas: TR_TOT) and aquatic species (EnviroAtlas: AQ_TOT) were used for this analysis. The final datasets were counts weighted by the area of the HUC-12 unit providing a number per 100 square kilometers.

Supporting: habitat and lands protected for conservation Lands protected for conservation were also used as another indicator of biodiversity/habitat as a supporting service. To quantify this, we created a managed conservation lands (MCL) layer consisting of public lands as defined by the USGS Protected Areas Database of the US (PAD-US 2.0), and conservation easements documented in the National Conservation Easement Database (NCED) (https://www.conservationeasement.us). Conservation easements are important to include in this dataset because such lands support biodiversity and can cover large swaths of land, especially in Florida. We recognize that not all land trusts choose to share their conservation easements with the NCED due to lack of funding and technical capacity, or privacy concerns, and that it may vary from state to state (Rissman et al., 2019). This may present an unknown bias in Georgia, Alabama, and other states, however in Florida the area of easements acquired by land trusts are very minor in comparison to those held by state and federal agencies. The NCED draws from exhaustive mapping of all conservation "managed lands" by the Florida Natural Areas Inventory for the state. Based on reviewing known conservation easements in our areas we determined that the NCED was the best available region-wide dataset for our needs. Before combining these two datasets into one, we refined the PAD-US dataset to exclude US Department of Defense sites that were <10,000 acres and Recreation Management Areas from all public agencies, including water features such as lakes, rivers, and reservoirs, as these areas are not necessarily managed with a goal of supporting conservation. This combined dataset does not include other management practice incentive programs, such as the Conservation Reserve Program (CRP) and the Environmental Quality Incentives Program (EQIP) because the areas in those programs are afforded only a temporary conservation status. The final datasets consisted of the proportion of HUC-12s protected for conservation.

Analysis

It was not our intention to produce a multivariate statistical analysis of the cross-scale relationships among indicators, which would have introduced additional statistical challenges associated with the modifiable aerial unit problem (Fotheringham and Wong, 1991; Gotway and Young, 2002). However, we examined bivariate relationships between variables (**Table 1**), and compared the distributions of variables, plotting their kernel density functions to discover the likelihood of similarity between subregions and the larger regions within which they were nested (Scholz and Stephens, 1987).

Data preparation involved standardizing all datasets in the common HUC-12 spatial framework. Since national EnviroAtlas datasets (Agricultural N and Imperiled Species) are provided in this format, data preparation involved collating values from multiple attributes to calculate area-weighted summary measures of total N and G1|G2 species counts. Analyses of datasets included summary statistics over space and, for the NPP datasets, over a period of 5 years, including mean, median, standard deviation and CV. We then mapped and charted the spatial variability of summary measures to visually compare the study areas. We also conducted 216 pairwise bivariate regressions of the nine variables reviewed in this analysis to observe trends between variables at all scales (**Supplementary Material 3**).

Kernel density estimation was used as a non-parametric way to evaluate and compare the distributions of the datasets from the seven study areas for the nine indicators. The kernel density estimator (Silverman, 1986) of an unknown density f is given by:

$$\hat{f}_h(x) = \frac{1}{n}\sum_{i=1}^{n} K_h(x - x_i) = \frac{1}{nh}\sum_{i=1}^{n} K\left(\frac{x - x_i}{h}\right)$$

where x_1,....,x_n is a sample drawn independently from the distribution with density f, h is the bandwidth, and K is the kernel. We used the function density() in the R base package (R Core Team, 2019) with its default settings, which include using a normal kernel.

Dataset density functions were statistically compared using the two-sample Anderson-Darling test (Scholz and Stephens, 1987), as implemented in the R (R Core Team, 2019) package kSamples (Scholz and Zhu, 2019). The null hypothesis for this test is that the samples come from the same (continuous) distribution. The test is based on the goodness of fit statistic of Anderson and Darling (1954) and is non-parametric—no underlying distribution needs to be specified. Such statistics often are used as a measure of distance between distributions or datasets. We used it in this sense to compare our datasets across scales (**Supplementary Material 4**).

To compare the indicator datasets for each LTAR study area, we constructed radar plots of the summary values of the HUC-12s. Values were normalized to a 0-1 scale within each category and plotted together in a radar plot configuration, providing a graphical comparison of the data space covered by ecosystem service indicators for each study area. This comparison included normalized values for: (1) mean annual NPP (primary production); (2) the inverted values of agricultural N loading (reduced N); (3) the ratio of FG1 to FG3 areas (proportion of natural lands within working lands); (4) the FG2 effective mesh size (a measure of connectivity); (5) the proportion of managed conservation lands (area conserved); and (6) terrestrial rare and imperiled species. Values for agricultural N were inverted prior to normalizing the scale so that values close to one indicated low levels of N runoff. This provided a consistent interpretive index where higher values could be associated with higher levels of ecosystem services to enable a visual comparison of tradeoffs and synergies.

RESULTS

Land Cover and Working Lands Connectivity

Land Cover

Agricultural land covers in the study areas of the Southeast (**Figure 3**) are interspersed with extensive forests and wetlands. In the southernmost areas, agricultural land uses are associated with freshwater herbaceous wetlands, characteristic of humid, subtropical climates (Chen and Gerber, 1990; Beck et al., 2018)

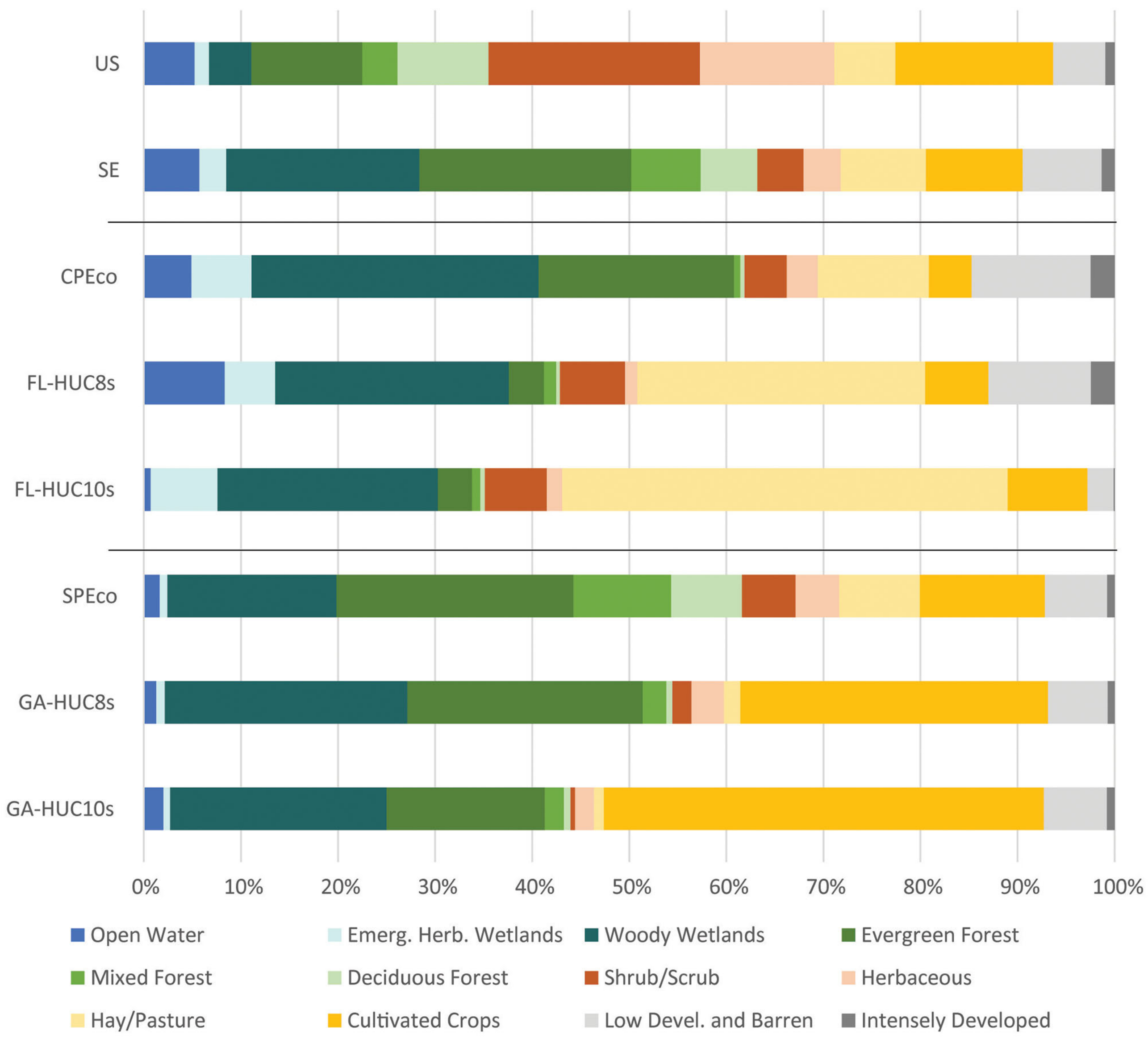


FIGURE 3 | Proportion of land cover type by study region based on the 2016 USGS National Land Cover Database.

where water is a key driver. This association transitions to a coupling with riparian forested ecosystems as one moves northward. While the area of open water in the SE is only slightly greater than the US, 23% of the land area is covered with wetlands (herbaceous and woody) in the SE compared to about 6% for the US. These differences are even more marked for the CPEco, where 35% of the area is covered by these land cover categories. The other distinguishable feature of the SE is that the proportion of developed land classes is higher than the US, driven by high rates of urbanization in the CPEco where the amount of developed cover is 14.7%, more than double the US, and SPEco levels of 6.3% and 7.2%, respectively.

In the SE mega-region, cultivated crops and hay/pasture land covers ~19% of the area, slightly less than the US value of 23% (**Figure 3**). However, the proportion of these areas are lower in the southern ecoregion (CPEco = 16%), than the northern ecoregion (SPEco = 21%). At a smaller extent though, the regional basins (HUC8s) and local watersheds (HUC10s) are more dominated by crop and hay classes than the ecoregions they lie within. These lands occupy 36% of the FL-HUC8s study area and 33% of the GA-HUC8s area, increasing to 54% and 46%, respectively at the HUC-10s level. Of note is the shifting composition of crop and hay classes in moving down the hierarchy of focal areas. At the CPEco and SPEco levels, the proportion of crop and hay is fairly evenly split between the classes. However, at finer scales, hay/pasture classes dominate in the Florida study areas (30% of FL-HUC8s and 46% of FL-HUC10s), and cultivated crops in the Georgia study areas (32% of FL-HUC8s and 45% of GA-HUC10s).

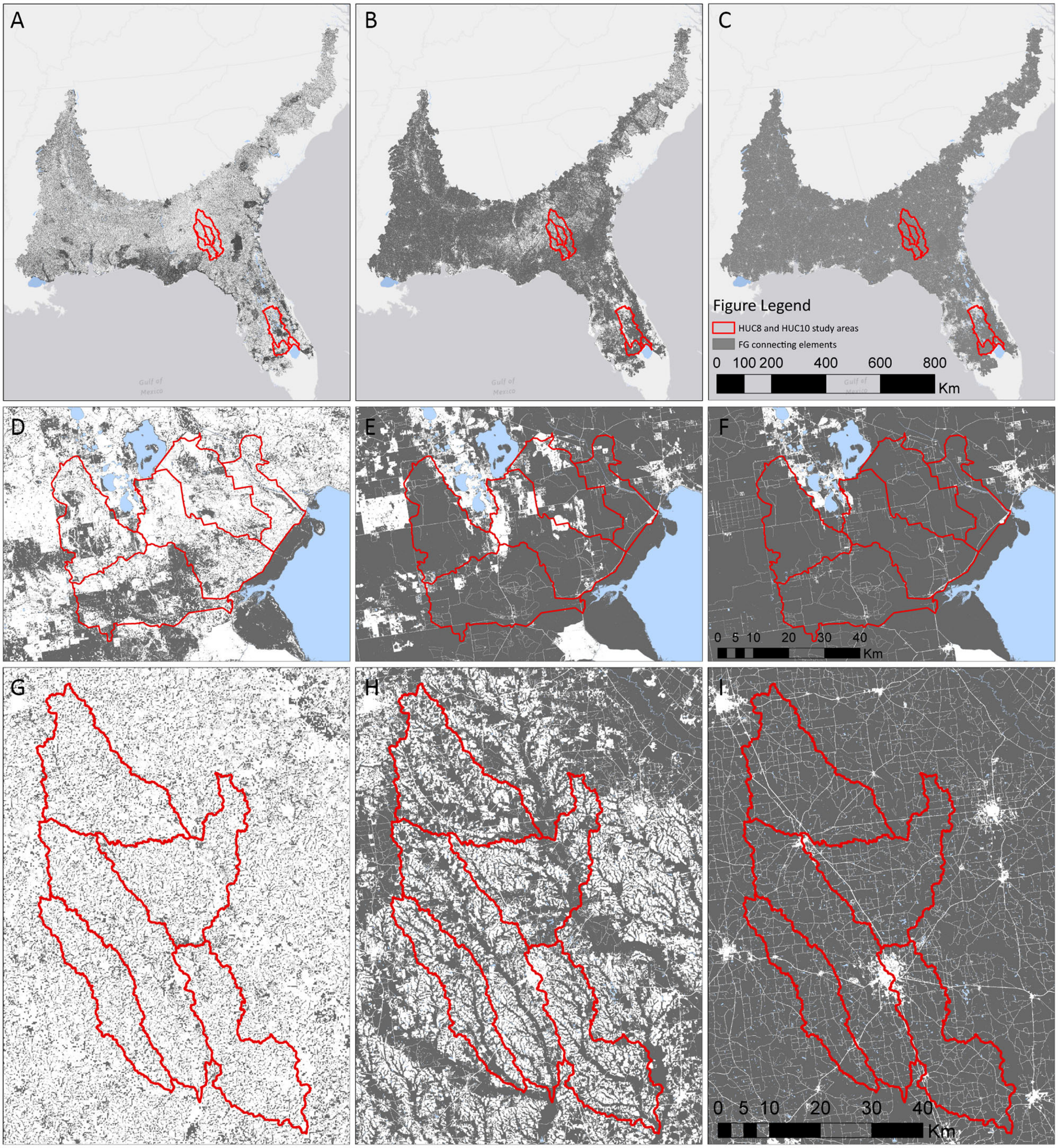


FIGURE 4 | Land use fragmentation geometries (FGs) in the southeastern US (top row), Florida (center row), and Georgia (bottom row), with increasing intensity of use (columns). Left column shows FG1, consisting of wetlands, shrublands and unmanaged forests (gray) for the Southeast **(A)**, Florida **(D)**, and Georgia **(G)**. Middle column shows FG2, consisting of areas in FG1 plus all forests, shrublands, grasslands, and hay production areas (gray) for the Southeast **(B)**, Florida **(E)**, and Georgia **(H)**. Right column shows FG3, consisting of areas in FG2 plus croplands (gray), for the Southeast **(C)**, Florida **(F)**, and Georgia **(I)**, so that areas excluded (white) are developed and open water land cover classes. See **Supplementary Material 1** for land cover classification and GIS workflow.

Fragmentation and Connectivity of Working Lands From Landscape Metrics

Across the SE and its subregions, remaining natural areas, characterized here as FG1 connecting elements (gray areas in **Figure 4**), occupy, on average, about a third or less of HUC-12 areas, except for HUC-12s in the CPEco study area, where the average area of connecting elements is over 40% (**Supplementary Table 1.6**). Despite having 30–40% of the landscape in natural areas, the southeastern US is highly fragmented, as demonstrated by high numbers of patches and low m_{eff}. This value changed markedly across all study areas upon the inclusion of low intensity working lands as connecting elements (i.e., FG2), when the proportion of connecting element area increased substantially to more than 50% (**Figure 5**). For all study areas, the number of patches declined substantially, while patch size and m_{eff} increased (**Figure 5**). Trends for AWMPA were nearly identical to those for m_{eff}. This was especially true in the SPEco, where production forestry is so common and those uses tend to be highly interspersed more "passively" managed areas. Generally, fragmentation is more pronounced in the Georgia study areas, however, the inclusion of low intensity working lands provided a critical "boost" to connectivity in local watersheds of both Georgia and Florida study areas (GA-HUC10s and FL-HUC10s). In Georgia, the increase in connectivity was clearly due to the inclusion of riparian forests in the analysis (**Figure 4**), while in Florida, the inclusion of grasslands used for pasture was the key factor increasing levels of landscape connectivity.

Including all working lands (high intensity croplands as well as low intensity) in FG3 effectively reduced fragmenting elements to a very small proportion of the overall landscape with the exception of the Florida ecoregion (CPEco) and regional basins (FL-HUC8s). These are, however, areas where hard urban edges, including roads, form substantial fragmenting elements and represent a significant proportion of developed land cover classes, keeping connecting elements to about 80% of the landscape.

Spatial Variability of Ecosystem Service Indicators

Provisioning Services Characterized by NPP on Working Lands; Mean and CV Over Time

Net primary production (NPP) in the SE generally varied considerably over space and from 1 year to the next during 2014–2018. The data were normally distributed with mean and median values closely related. The five-year mean of annual NPP averaged across the SE study area was ∼7,176 kg C/ha and the mean CV was 5.8%. The range of CV (1–13%) for working lands in the SE stands in contrast to the CV of 4–38% for the entire 1987–2018 dataset, which includes all years and all pixels. General comparisons (**Figure 6**) of the data show that NPP in the Florida HUC-12s was far more variable than in the Georgia HUC-12s, especially within the local study areas. In Georgia, NPP was consistently higher and far less variable. While ecoregional CV values in both ecoregions ranged widely, variability for the regional basin and local watershed areas in Georgia (i.e., GA-HUC8s and GA-HUC10s) were far lower than the ecoregional means, an unsurprising result related to the modifiable areal unit problem, previous mentioned. Within the LTAR sites, GPP was measured from eddy-covariance flux towers. At the ABS-UF site, annual GPP was measured in 2013–2015 at 24.589, 17.995, and 16.131 Mg C/ha in improved pastures, semi-native pastures and wetlands, respectively (Chamberlain et al., 2016; Gomez-Casanovas et al., 2018). Similar values were reported at the GACP site where in 2016, annual GPP for miscanthus and maize were 30.73 and 26.43 Mg C/ha, respectively (Maleski et al., 2019). While these data do not account for respiration C losses, they provide ground validation of production within the study areas, and indicate similar levels of production for the areas of working lands in Florida and Georgia.

Regulating Services Characterized by Nitrogen Runoff

Modeled values of agricultural N runoff (**Figure 7**) in the SE mega-region were skewed right with a median of 3 kg/ha, and the top 1% with values of 70–260 kg/ha. In ecoregions the median values were lower in CPEco than SPEco (1.5 and 3.5 kg/ha, respectively) but overall values were more variable in the CPEco where the SE maximum value was found, well-exceeding that of SPEco maximum of 183 kg/ha. At the regional basins and local watershed extents (HUC8s and HUC10s), the extreme values were more constrained but, the study areas in Florida had higher median N runoff estimates and greater variability. At the ABS-UF study site, based on pretreatment data for eight pastures from a previous study. average annual values of total N were 9.155 kg/ha (see Bohlen and Villapando, 2011). In Georgia, riparian forests are well-established throughout the area, particularly in the local watersheds region (GA-HUC10s), serving as buffers for agricultural runoff, and resulting in lower N runoff in streams. This is consistent with the GACP measured N loading values published for the LREW of ∼4.2 kg/ha for 1978–2014 (Bosch et al., 2020). For one HUC-12 located in Florida, N values were extremely high, which may have skewed trends slightly in the CPEco, and, based on personal knowledge of this area, we suspect this was an aberrant value in the dataset of 4,596 records.

Supporting Services Characterized by *Imperiled Species*

The distributions of G1|G2 imperiled aquatic and terrestrial species varied markedly across the SE mega-region. The EnviroAtlas data show the CPEco ecoregion supports higher values for both terrestrial and aquatic imperiled species, with terrestrial species strongly associated with scrub and sandhill habitats of the peninsula's ancient sand ridges, as well as the Apalachicola and Ocala National Forests, and aquatic species closely associated with springs habitat in the northern peninsula (**Figure 8**, **Supplementary Figure 1.2**). Buck Island Ranch, at the ABS-UF site, supports four G3 species but no G1|G2 species, although it has five federally threatened and endangered species on site. It also lies within a region of the CPEco, where many HUC-12 watersheds within the regional basins (FL-HUC8s) are each associated with 5+ imperiled species (**Figure 8C**), a nationally high level of rarity. Working lands in Florida both

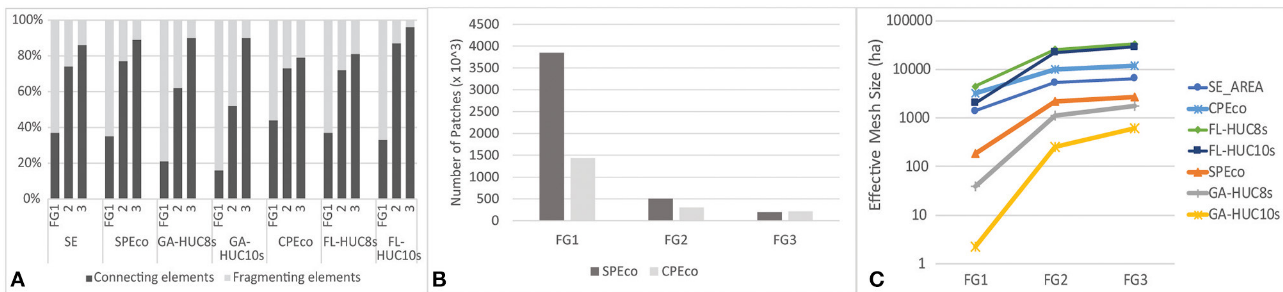


FIGURE 5 | Landscape ecological indices of connectivity and fragmentation for three fragmentation geometry (FG) levels. **(A)** Proportion of total mean HUC-12 area of connecting vs. fragmenting elements for each study area and for each FG level. **(B)** Graph of number of patches by ecoregion. **(C)** Graph of effective mesh size (m_{eff}). Tabulated data provided in **Supplementary Table 1.6**.

directly support and are embedded within a region of high conservation value for rare species.

Conversely, the EnviroAtlas data suggest that the SPEco ecoregion supports lower numbers of terrestrial imperiled species; the GACP site has no known G1|G2 terrestrial species listed. Although the data show that areas in the local watersheds are estimated to have either one or no G1|G2 terrestrial species, Georgia Biodiversity Conservation Data (https://georgiabiodiversity.a2hosted.com) suggests that each of the HUC-8s within the GA-HUC8s regional basins host 2–3 terrestrial G1|G2 animal species, and 3–5 similar plants species, an apparent contradiction of the datasets. Even though our study area excluded the coastal HUCs from both ecoregions, the SPEco clearly supports many rare G1|G2 aquatic species in inland riverine and headwater basins (**Supplementary Figure 1.2**). These freshwater systems and stream corridors with rare aquatics raise important challenges for working lands that may affect downstream water quality.

Lands Protected for Conservation

In the SE mega-region, the average proportion of HUC-12s protected for conservation is <11% (**Figure 9**). However, this proportion protected is heavily right-skewed, with 50% of the HUC-12s having <1.5% of their areas protected for conservation.

There is an extreme disparity evident between HUC-12 areas primarily in Florida vs. those in Georgia. Both areas have many HUC-12 units with <2% of the area in conservation lands. However, in the CPEco, half of HUC-12 units in the region have 11% or more of their area protected for conservation, and at the scale of the FL-HUC8s and HUC10s, many HUC-12s include over 20% of their land protected for conservation. Working lands in the FL LTAR Core region lie within an extensive landscape of public and private conservation lands.

In contrast, although the SPEco includes a few HUC-12s with substantial areas protected, the majority of HUC-12s have <1.4% of their area in protected status, less than the SE generally. The low proportion of land protected for conservation in the ecoregion is especially evident in the regional basins (GA-HUC8s), where only a handful of HUC-12 units include any land protected for conservation, and the majority have none. Working lands in the GACP region and their embedded natural areas represent the most valuable areas remaining for conservation although they are unprotected.

Distributions of Ecosystem Services Across Scales

Bivariate Pairwise Comparisons of Selected Ecosystem Services

Of the 216 pairwise comparisons of ecosystem service indicators at different scales, a subset is included here (**Figure 10**) to illustrate results related to our hypothesized relationships (**Figure 1**). The full collection of pairwise comparisons is included in **Supplementary Material 3**.

Regulating vs. provisioning services: N loading vs. NPP As hypothesized, downstream N loading increased with increasing NPP productivity at the ecoregion, regional basin, and local watershed scales. NPP was higher in general in the SPEco than in CPEco, and more HUC-12s had both high productivity and low downstream N loading in SPEco (**Figures 10A,B**). Drilling down, the GA-HUC8s (regional) and -HUC10s (local) had lower modeled N loadings than the Florida areas, and the relationship with NPP was clearer, appearing asymptotic at lower N levels in Georgia versus Florida. The variance in NPP (NPP_CV) showed an apparent negative relationship with NPP in Florida, potentially driven by the variability in grassland productivity, vs. no obvious relationship in Georgia.

Supporting vs. provisioning services: aquatic biodiversity vs. NPP A high proportion of Florida and Georgia HUC-12 units had zero aquatic imperiled species (AqG1|G2) present, but in contrast to predictions for both ecoregions, there were more AqG1|G2 species at intermediate NPP levels (**Figures 10C,D**) although there was a longer right tail with low AqG1|G2 numbers at high N for Florida HUC units. The same broad pattern of more rare aquatic species at intermediate NPP values could be seen at all landscape scales from local watersheds (HUC10s) to the SE mega-region.

Supporting vs. regulating: FG2 vs. N loading The combination of natural areas with low intensity working lands (described by

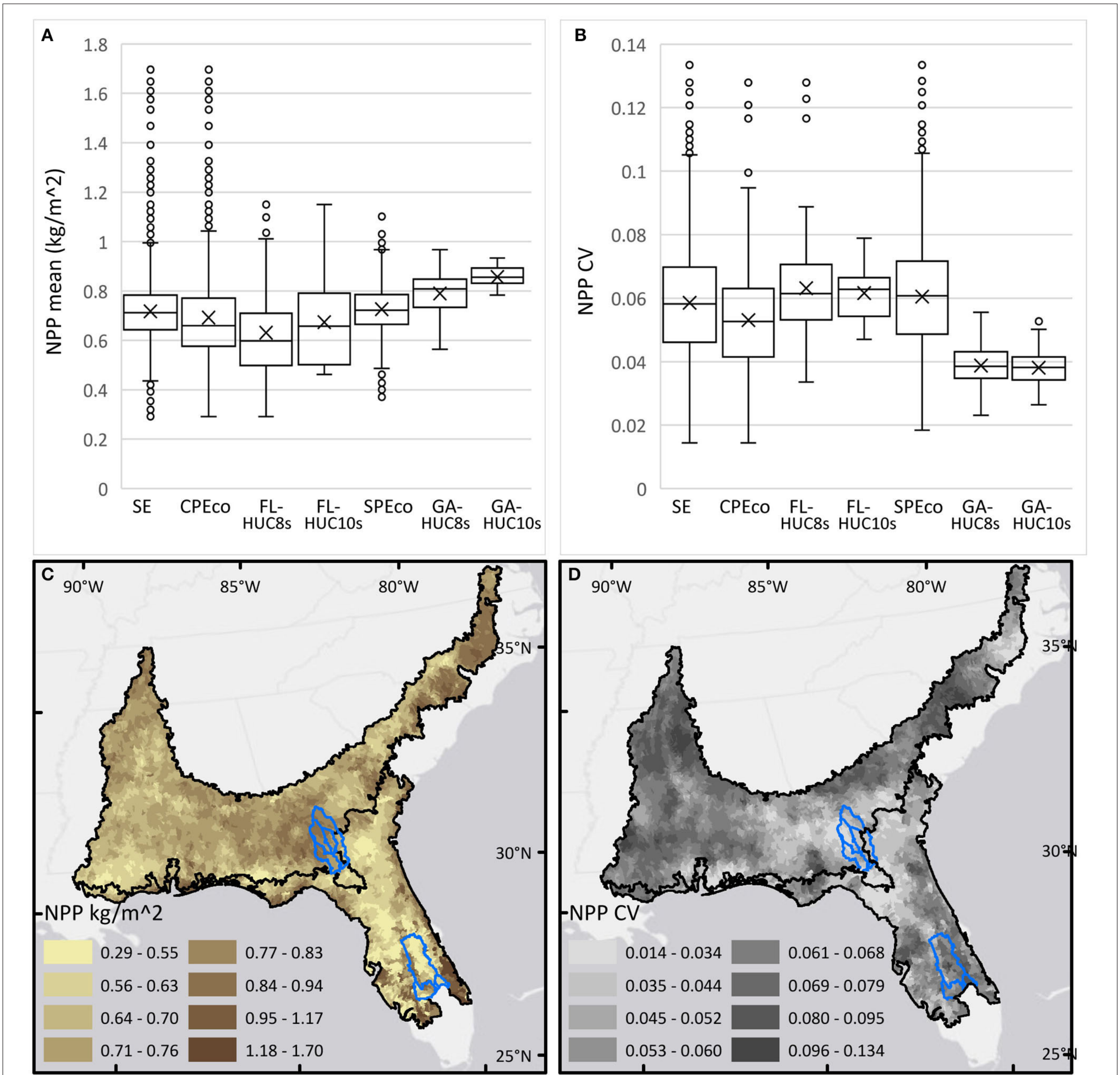


FIGURE 6 | Boxplots and maps of net primary production (NPP, kg C/m^2) and coefficient of variation (CV%) average annual mean values, 2014–2018, calculated for HUC-12 areas and summarized for each study area (clockwise from top left). **(A)** Boxplots of NPP for each study area. **(B)** Boxplots of NPP CV for each study area. **(C)** Map of NPP calculated for HUC-12s in the Southeast (SE). **(D)** Map of NPP CV calculated for HUC-12s in the Southeast (SE). Blue outlines show HUC-8 areas.

FG2) means there is a higher proportion of contiguous land covers available for regulatory ecosystem services. As expected, there was a negative relationship between the proportion of these areas and N loading (**Figures 10E,F**). This effect was observable at all scales from local watersheds to the entire SE but was most marked in Georgia regional basins and local watersheds (GA-HUC8s and -HUC10s, respectively; **Figures 10F,H**) These areas help buffer streams and offset the N loading effects, as found in other published studies from this region (Lowrance et al., 1984; Bosch et al., 2020; Pisani et al., 2020).

Kernel Density Functions of Ecosystem Services Across Scales

While box plots (**Figures 6–9**) allow for visual comparisons of the mean and variance for selected ecosystem service indicators, the Anderson-Darling test was used to test the goodness of fit or distance between pairs of kernel density functions in the datasets (Scholz and Stephens, 1987). We

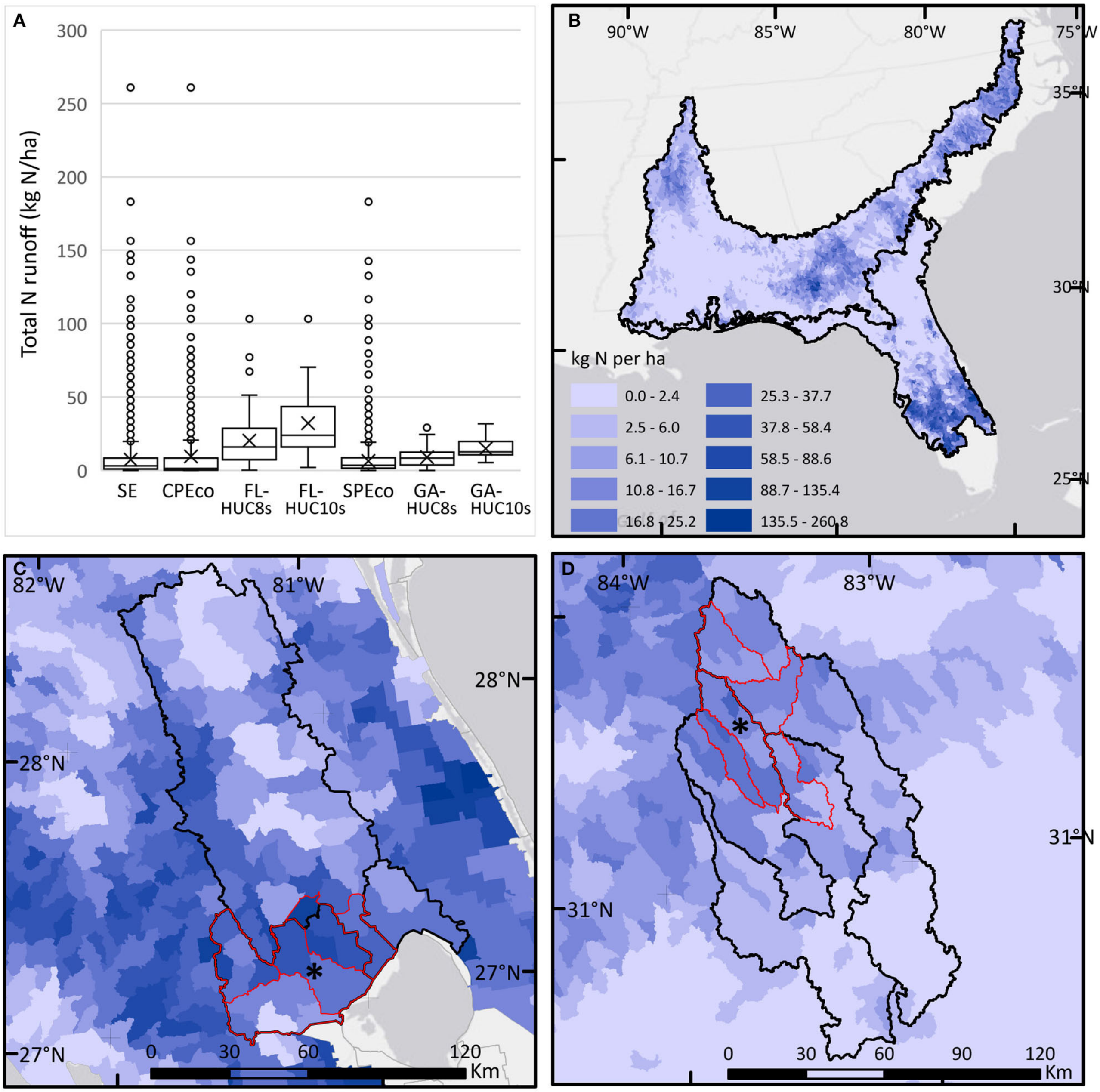


FIGURE 7 | Boxplots and maps of modeled agricultural nitrogen runoff (N, kg/ha) calculated for HUC-12 areas and summarized for each study area. **(A)** Boxplots of N for each study area. Asterisk (*) shows location of LTAR sites. **(B)** Map of N calculated for HUC-12s in the Southeast (SE). **(C)** Map of N calculated for HUC-12s in the Florida study areas. **(D)** Map of N calculated for HUC-12s in the Georgia study areas.

produced 81 pairwise comparisons (nine variables × nine scale pairs), of which, only 17% had similar density distributions (i.e., larger *P*-values; **Supplementary Material 4**) suggesting that most ecosystem services do not scale concordantly. In cases where concordant distributions were found, this was most often among the smaller local, HUC-10, and regional, HUC-8, areas (**Figure 11**). The distribution of mean NPP values in the local watersheds and regional basins of Florida were similar, which was, in turn, similar to the CPEco ecoregion (**Figure 11A**), unlike the Georgia extents which were all dissimilar (**Figure 11B**). The distribution of CV values, describing the variability in mean NPP, was also similar among the smaller extents both in Florida and Georgia (**Supplementary Table 4.2**). Although the records of G1|G2 species include many 0 values, the distribution

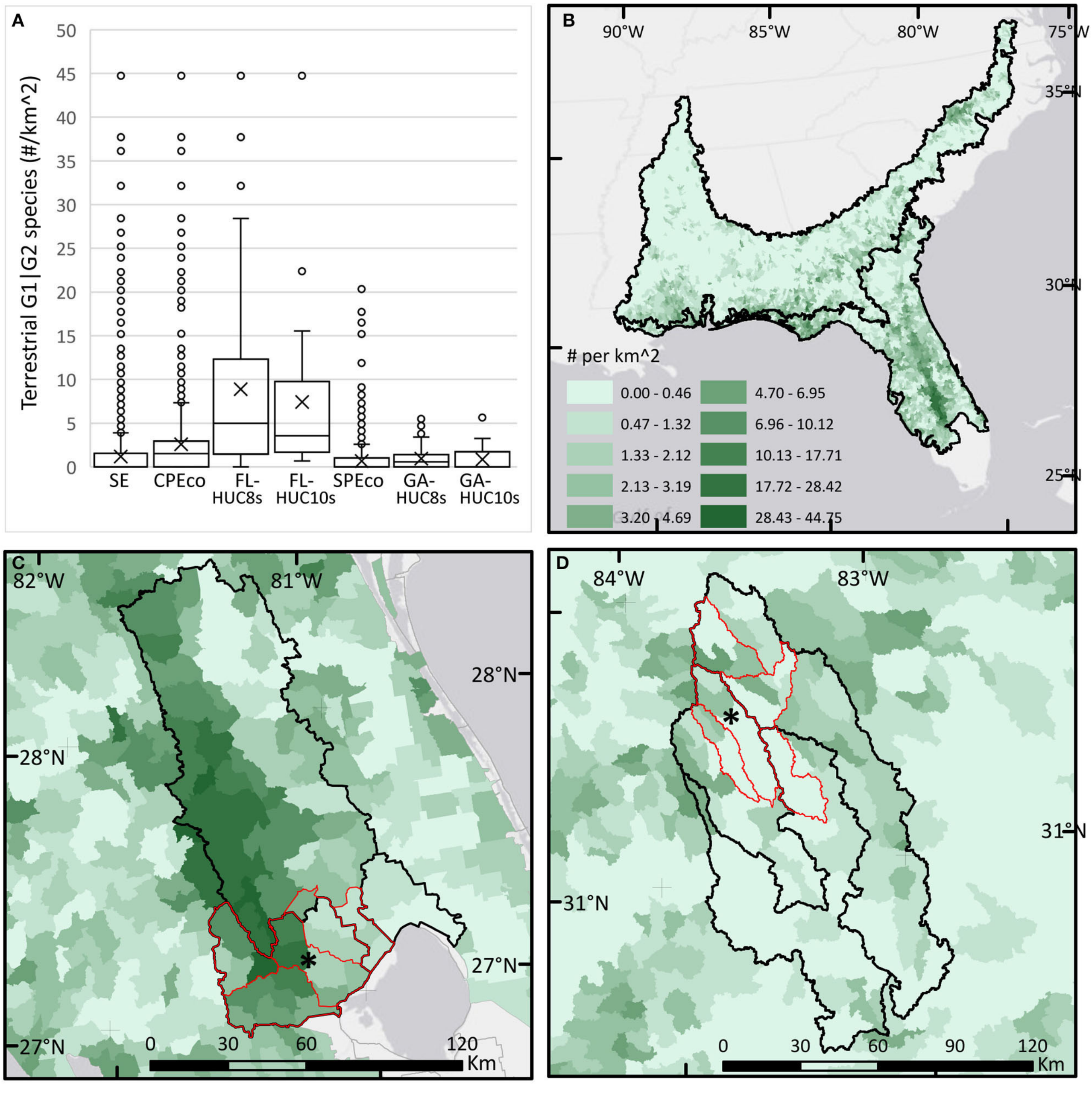


FIGURE 8 | Boxplots and maps of numbers of rare and imperiled terrestrial species (G1|G2) calculated for each HUC-12 (n /100 km^2) and summarized for each study area (clockwise from top left). Asterisk (*) shows location of LTAR sites. **(A)** Boxplots of terrestrial G1|G2 species for each study area. **(B)** Map of terrestrial G1|G2 species calculated for HUC-12s in the Southeast (SE). **(C)** Map of terrestrial G1|G2 species calculated for HUC-12s in the Florida study areas. **(D)** Map of terrestrial G1|G2 species calculated for HUC-12s in the Georgia study areas.

in the Florida local and regional basins (FL-HUC10s and -HUC8s; **Figure 11C**) were smooth with low numbers of species. In contrast, the GA-HUC10s and -HUC8s (**Figure 11D**) were similar, but distributions were spiky because of large numbers of zeroes. The distributions of the proportion of FG2 connecting elements were similar in the CPEco and FL-HUC8s but were dissimilar at other scales in Florida and Georgia (**Figures 11E,F**). However, the proportion FG3 connecting elements were similar for the smaller extents in Georgia (**Supplementary Table 4.9**).

Multivariate Comparison of Ecosystem Services in Radar Plots

Radar plots showing a graphical representation of six values of ecosystem service indicators illustrated the tradeoffs and synergies among these services (**Figure 12**; **Supplementary Table 1.3**). Not surprising, ecosystem services derived from the SPEco, totaling 70% of the SE, were more similar to those from the entire SE mega-region than services from the Florida ecoregion (CPEco), which represents the

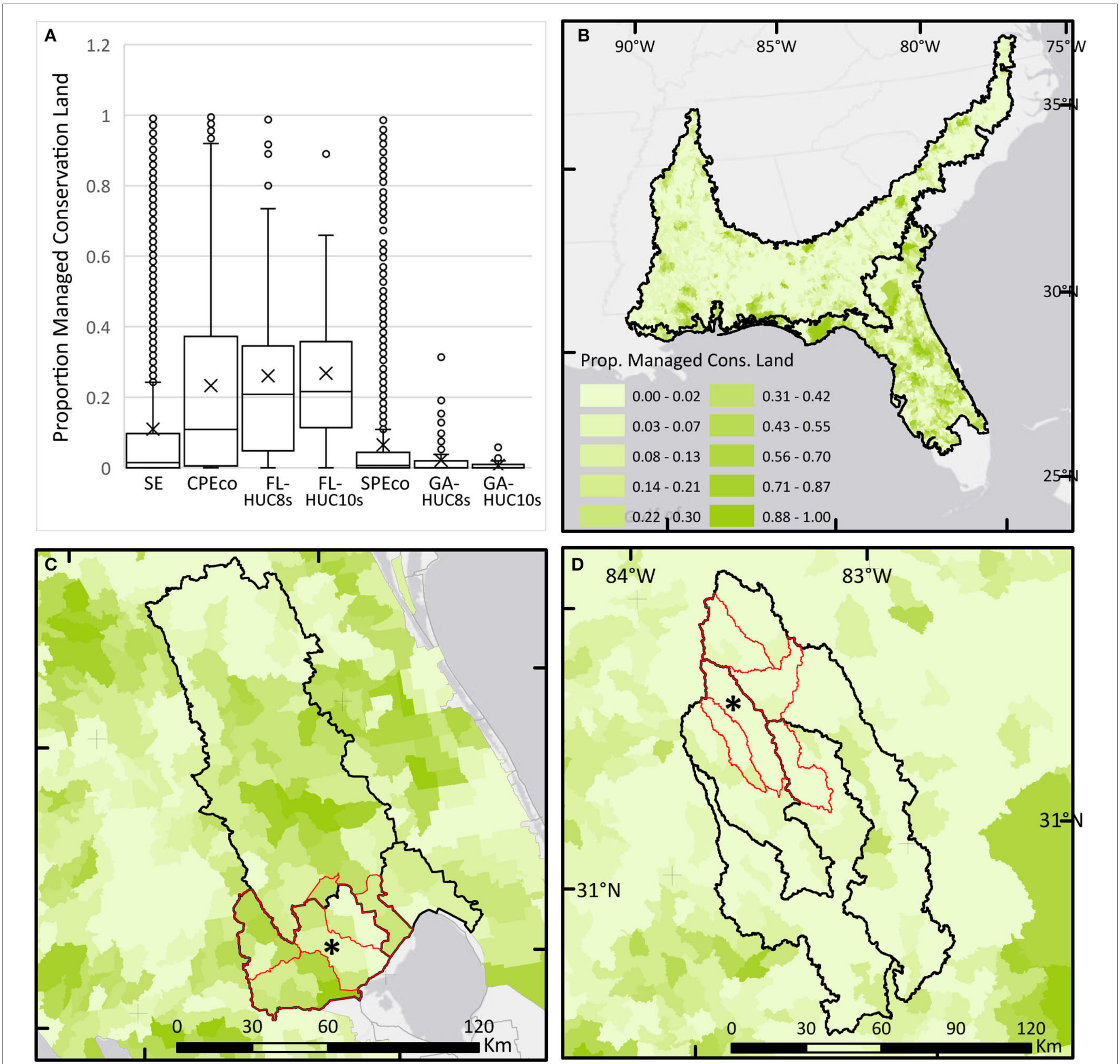


FIGURE 9 | Boxplots and maps of percent area of lands protected for conservation calculated for HUC-12 areas and summarized for each study area (clockwise from top left). Asterisk (*) shows location of LTAR sites. **(A)** Boxplots of percent area of lands protected for conservation for each study area. **(B)** Map of percent area of lands protected for conservation calculated for HUC-12s in the Southeast (SE). **(C)** Map of percent area of lands protected for conservation calculated for HUC-12s in the Florida study areas. **(D)** Map of percent area of lands protected for conservation calculated for HUC-12s in the Georgia study areas.

remaining 30%. The balance of ecosystem services is not evenly distributed across the SE.

Compared with the Georgia ecoregion (SPEco) and the SE, the local watersheds and regional basins (GA-HUC10s and -HUC8s) appeared quite different, showing higher productivity and higher agricultural N runoff (lower values on the radar plot) than mega-regional median values. In terms of habitat and biodiversity indicators, they were similar to the values for the SE or the SPEco. But they differed in that a greater proportion of working lands were high intensity croplands as opposed to the less intensive, but more extensive, production forests found in the rest of the SPEco.

In the local watersheds and regional basins of Florida (FL-HUC10s and -HUC8s), productivity was extremely low, and yet these areas still showed higher downstream N loading. However, the Florida regions showed high levels of supporting services, as measured by conservation indicators, with more rare species, a

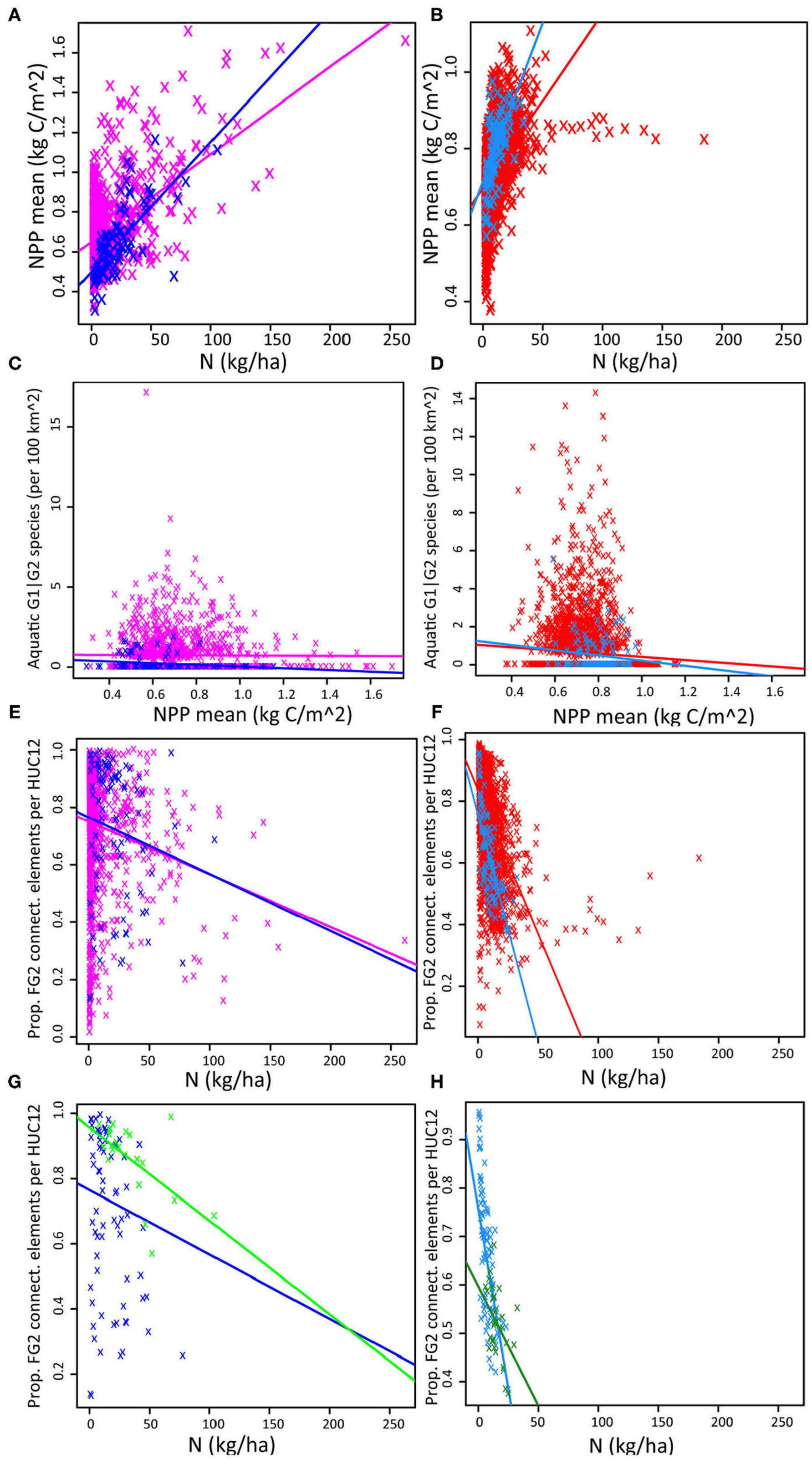


FIGURE 10 | Pairwise comparisons of selected ecosystem service indicators (**Supplementary Material 3**). **(A)** Provisioning vs. Regulating: Mean NPP, 2014–2018 (kgC/m^2) vs. Agricultural N runoff (kg/ha^1), in CPEco (magenta, n = 1,219), and FL-HUC8s (blue, n = 94) study areas. **(B)** Provisioning vs. Regulating: Mean NPP,

(Continued)

FIGURE 10 | 2014–2018 (kgC/m^2) vs. Agricultural N runoff (kg/ha), in SPEco (red, n = 3,377) and GA-HUC8s (dodger blue, n = 116) study areas. **(C)** Supporting vs. Provisioning: Aquatic G1|G2 species (n/100 km^2) vs. Mean NPP, 2014–2018 (kgC/m^2), in CPEco (magenta, n = 1,219), and FL-HUC8s (blue, n = 93) study areas. **(D)** Supporting vs. Provisioning: Aquatic G1|G2 species (n/100 km^2) vs. Mean NPP, 2014–2018 (kgC/m^2), in SPEco (red, n = 3,377) and GA-HUC8s (dodger blue, n = 116) study areas. **(E)** Supporting vs. Regulating: Proportion of FG2 connecting elements vs. Agricultural N runoff (kg/ha), in CPEco (magenta, n = 1,219), and FL-HUC8s (blue, n = 94) study areas. **(F)** Supporting vs. Regulating: Proportion of FG2 connecting elements vs. Agricultural N runoff (kg/ha), in SPEco (red, n = 3,377) and GA-HUC8s (dodger blue, n = 116) study areas. **(G)** Supporting vs. Regulating: Proportion of FG2 connecting elements vs. Agricultural N runoff (kg/ha), in FL-HUC8s (blue, n = 94), and FL-HUC10s (green, n = 23) study areas. **(H)** Supporting vs. Regulating: Proportion of FG2 connecting elements vs. Agricultural N runoff (kg/ha), in GA-HUC8s (dodger blue, n = 116), and GA-HUC10s (forest green, n = 27) study areas.

greater proportion of conservation lands, and large patch sizes within low intensity working lands. In one respect the local watersheds in Florida were more similar to the SE study area overall and differed from Georgia, retaining a higher proportion of natural areas within working lands. Comparing this value on the radar plots for both SPEco and CPEco regions, the CPEco had the effect of "pulling" the entire SE to a higher rank on the axis. Likewise, higher values for NPP in the SPEco, such as seen in the GA-HUC10s and -HUC-8s presumably had the effect of pulling the SPEco region to higher levels on the NPP axis.

DISCUSSION

Globally the agricultural sector is challenged to no longer simply maximize productivity, but rather to optimize across multiple goals including environmental stewardship, and the prosperity and well-being of rural communities (Pretty et al., 2010). Optimization requires a better understanding of the contributions of working lands toward multifunctional ecosystem services at local, regional and national scales (Petersen and Snapp, 2015). We described the tension resulting from this optimization with the image of the fictional "pushmi-pullyu" character in the Introduction, in which some land management actions "pull" ecosystem services, while at the same time unintentionally "pushing" disservices. But, in terms of the overall balance of ecosystem services, the pushmi-pullyu character has only two heads and no scaling issues, whereas comparative analyses of synergies and trade-offs among production and other ecosystem services cannot ignore issues of scale and complexity. The challenge to analyze tradeoffs and synergies of ecosystem services in working lands is more complex, and requires a framework for scaling, analysis, and comparison. While the ecosystem services framework provides a unifying concept for comparing diverse outcomes from agroecosystems, the HUC spatial framework is useful for evaluating how well these ecosystem services do or do not scale. This research constitutes an attempt by two LTAR sites in a common geographic zone to compare outcomes of ecosystem services, probing the limits of how representative they are of the larger context, an understanding which is essential for accomplishing the Network's goals related to national scale agroecological research.

We selected the HUC-12 hydrologic unit (USGS, 2015), as the spatial grain for comparing multiple empirical and modeled datasets across scales, from site to regional areas of interest in the southeastern US. In this, we were strongly influenced by US EPA (2011) which also used the HUC-12 for its nationwide analysis. Our analysis used areal measurements of indicators to summarize and characterize regions, and so, an alternate selection of boundaries would have likely changed our characterizations of the regions, (Fotheringham and Wong, 1991). However, our selection of regions was not arbitrary, but was based on indicators for which we had relatable *in situ* measurements from the two LTAR sites in Georgia and Florida, and which related to the ecoregional and hydrologic frameworks of our analyses.

After considerable evaluation of available data, we chose five factors to derive nine indicators of provisioning, regulating and supporting ecosystem services, for which we could acquire empirical data throughout the southeastern US at the HUC-12 grain of analysis. Ultimately, we used: (1) net primary production (annual mean and CV), (2) agricultural Nitrogen runoff, (3) imperiled species (terrestrial and aquatic), (4) the proportion of areas managed for conservation, and (5) connecting landscape elements (three types). While we used land cover data extensively in this analysis, land cover characterization was one result used to compare study areas, and these data were combined with other datasets as described in our methods and in **Supplementary Material 1**.

Characterizing Ecosystem Services Associated With Working Lands

Our characterization of ecosystem services associated with working lands in the Southeast was constrained to those for which we were able to produce adequate datasets that followed across scales for all our study areas. Provisioning was characterized by mean annual NPP, and mean CV of NPP (over 5 years) since otherwise aligning crop yields and grassland productivity is challenging. Regulating services were indicated by agricultural N (modeled), an ecosystem disservice that we inverted for consistency in the radar plot comparison (less N = high service). Other ecosystem service indicators are of great interest, such as pollinator populations or species biodiversity but to date these lack complete spatial coverage and often do not incorporate data from working lands. Similarly, although we had some habitat specific data on greenhouse gas emissions from our LTAR research, it was not enough to characterize regulatory services over the heterogeneity of an entire HUC-12 around a LTAR site for comparative purposes. We fell back on the supporting services of species rarity (G1|G2 species), the proportion of conservation lands protected, and landscape

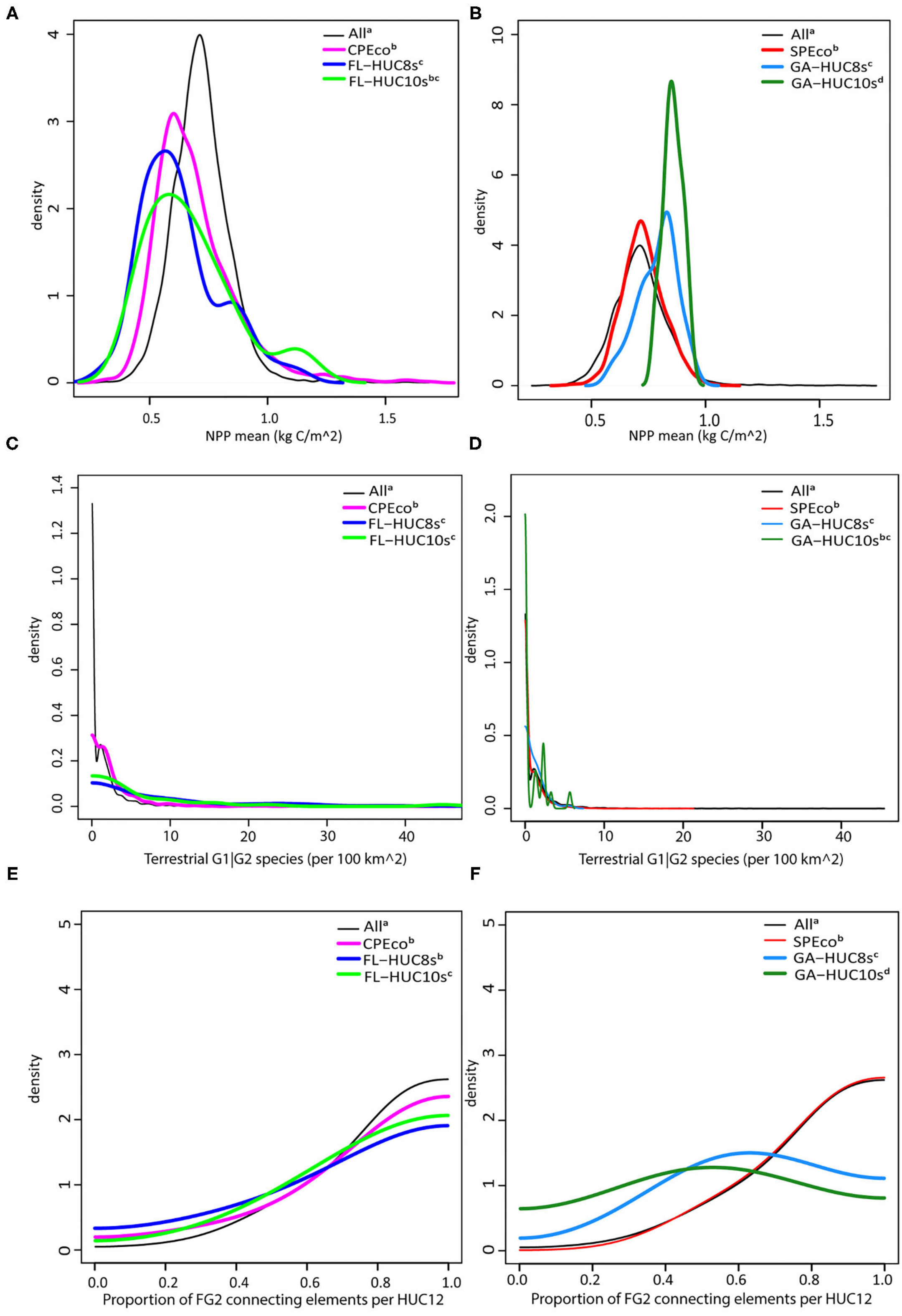


FIGURE 11 | Kernel density function comparisons where superscripts indicate similar distributions (Anderson-Darling statistic $P > 0.05$; **Supplementary Material 4**). **(A)** Mean NPP (kg C/m^2)—all data with Florida sub-regions. **(B)** Mean NPP (kg C/m^2)—all data with Georgia sub-regions. **(C)** Terrestrial Imperiled Species (n/100 km^2)—all data with Florida sub-regions. **(D)** Terrestrial Imperiled Species (n/100 km^2)—all data with Georgia sub-regions. **(E)** Proportion of FG2 connecting elements—all data with Florida sub-regions. **(F)** Proportion of FG2 connecting elements—all data with Georgia sub-regions.

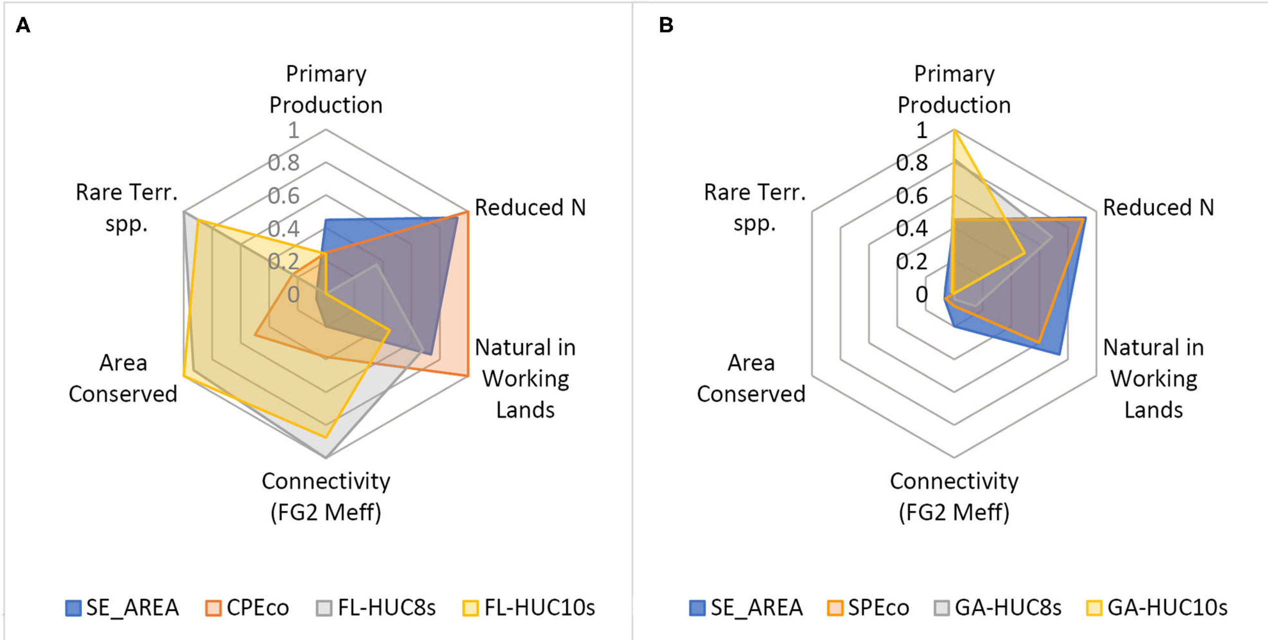


FIGURE 12 | Radar plots of six ecosystem service indicators for each of seven ecosystem service indicators: (clockwise from top) Primary production (mean NPP); Reduced N, the inverse of agricultural nitrogen runoff; Natural in Working Lands (ratio FG1:FG3); Connectivity (FG2 m_{eff}, effective mesh size of the FG2 connecting elements, i.e. natural areas and low intensity working lands); Area Conserved (prop. MCL, proportion of lands protected for conservation); and Rare Terrestrial Species (TerrG1|G2, terrestrial G1|G2 imperiled species). **(A)** Florida study areas: SE megaregion, CPEco ecoregion, FL-HUC8s regional basins, and FL-HUC10s local watersheds. **(B)** Georgia study areas: SE megaregion, SPEco ecoregion, GA-HUC8s regional basins, and GA-HUC10s local watersheds.

factors, including the proportion and connectivity of natural lands vs. agricultural lands.

Indicators of provisioning services in the CPEco and SPEco demonstrated the variability of production in the region (**Figures 6**, **10A,B**). The agricultural regions of Georgia, described by the GA-HUC10s and -HUC8s, generally occupy the higher ranges of mean NPP values in the region, while those of Florida, in the FL-HUC10s and -HUC8s, are in the lower ranges, and are much more variable. Together, production values from these two smaller areas in Georgia and Florida, coincident with the LTAR Network sites, cover the range of NPP values in all but the most extreme outliers of the SE. While the distribution of mean NPP values was similar only among the Florida study areas, the amounts of change within those annual mean values, as described by CV, was similar between local scales (-HUC10s and -HUC8s) in both Florida and Georgia.

Regulating services in working lands of the Southeast are provided largely in the natural areas buffering riparian and aquatic systems throughout the region. This effect is seen wherever "natural" lands and low intensity working lands follow adjacent to waterways. The distribution of N runoff values in our local and regional study areas were higher than the overall SE region. Keeping in mind that both the FL-HUC10s and GA-HUC10s study areas comprise mostly agricultural lands cover classes at more than twice the proportion than the SE (**Figure 3**), the difference in distributions of total N runoff values is not surprising.

The southeastern US is distinguished by highly heterogenous land covers with natural areas that are strongly associated with forested and wetland land covers. Together these natural lands total 37% of the southeastern US study area, well in excess of the 30% for the continental US. Services derived from less intensive agricultural land uses as well as from silviculture generally have fewer environmental costs than intensive agricultural operations (Power, 2010). In the southeastern US, when we combined these less intensive working lands with natural habitats (**Figure 4**) the extent of the landscape from which supporting ecosystem services could be expected essentially doubled from 37 to 74% (**Figure 5**, **Supplementary Table 1.6**). This percent was similar in the two ecoregion study areas: 77% of the SPEco, and 73% of the CPEco. Further considering this addition, the spatial configuration of the "ecosystem service landscape" was transformed, with patch areas providing services greatly increased, and fragmentation decreased.

Largely in response to the pressures of urbanization, another human dimension that will affect drivers of ecosystem services, more land is protected for conservation in the CPEco than SPEco (23 vs. <2%). This stems from decades of massive investment in public land acquisition and purchase of conservation easements in Florida by the federal government and the state (Farr and Brock, 2006). In comparison, with fewer conservation land purchases by the state of Georgia and elsewhere in the southeastern coastal plain, remaining natural habitats important for ecosystem services lie disproportionately within working lands and are typically unprotected.

Trade-Offs and Synergies of Landscape-Scale Ecosystem Services

Results from the pairwise comparisons of services (provided in **Supplementary Material 3**) were broadly in the directions predicted (**Figure 1**) and showed the same general patterns from the local (-HUC10s) to the ecoregion scale, although the trendlines for the relationships were often markedly different in the CPEco region in Florida other parts of the SE (**Figure 10**).

The impacts of the inclusion of working lands, and particularly the shift to combine connectivity of natural areas with low intensity and then high intensity working lands (from FGI to FG2 and FG3), highlights obscured environmental service levels at different landscape configurations. Pairwise comparisons of productivity against an increasing proportion of the connected landscape in an ecoregion, from FGI to FG2, then to FG3, showed: first, a strong negative relationship, i.e., lower productivity with a high proportion of natural areas (FG1); then, a weaker but still negative relationship when low level intensity agricultural lands were added (FG2); and finally, the expected steep positive asymptote for high productivity with all agricultural lands (FG3) included. The relationships are more extreme in the SPEco region with more croplands than the CPEco in Florida. Similarly for N runoff comparisons, going from the FG1 to FG2-configured landscape, shows the respective transition from a negative relationship with a high amount of natural areas (more natural areas, less N) to a mixed relationship depending on the ecoregion (slightly positive in CPEco, slightly negative in SPEco). Subsequently the FG-N relationship becomes obviously positive with the inclusion of high intensity agricultural lands.

Other data also indicate differences in ecosystem service trade-offs, for example the extensive semi-native grazing lands and scattered seasonal wetlands in Florida support high biodiversity including rare species, but yet these regions still produce high downstream N loadings. While croplands in Georgia have high productivity, streams and forested wetland habitats dissect the landscape, buffering and lowering N nutrient loading from adjacent crop fields, revealing a synergistic relationship among regulating and supporting services in these regions.

Representativeness of USDA LTAR Network Sites

To accomplish the task of understanding the interactions among indicators in broad domains of production, environment, and rural well-being in US agroecosystems (Kleinman et al., 2018), it is necessary to characterize the LTAR Network locations. For the two LTAR sites included in this study, ABS-UF (Florida) and GACP (Georgia), our data and analyses addressed two aspects of representativeness of the regions in which they lie. First, we quantified their landscape configurations and ecosystem services, including comparisons with site-specific data collection at the LTAR locations. Second, we characterized the tradeoffs and synergies among ecosystem services at the LTAR sites versus the increasing spatial extents across the region.

For the ecosystem service indicators we analyzed, observations measured within the LTAR sites fell within the ranges of observed or modeled data values. For those indicators, we concluded that the LTAR sites were represented well by the data summarized in the HUC-12s immediately surrounding the LTAR, i.e., the LTAR Core areas, or local watersheds (**Figures 2C,D**), and to some extent the broader regional basins, or -HUC8s.

Our analyses showed how the two LTAR Core areas represent specific conditions of agriculture-dominated watersheds within the range of values encountered in the southeastern US. Given the huge variability evident in the data for the SE, it is not surprising to find that the ABS-UF and GACP LTAR sites are not representative of the whole. But we have found that, regionally, measurements at these sites offer a good representation of the surrounding watersheds. This conclusion was enabled by the hierarchical nature of the analysis and our ability to relate data summaries to *in situ* measurements within the study areas.

Our compilation of ecosystem service indicators, visualized in the radar plots, gives a simplistic comparison of the tradeoffs and synergies among two LTAR sites. The graphical analysis highlights clear differences in the "space" occupied by them, which we were able to show because we compared sub-regions of the same mega-region (i.e., the SE). It shows that synergies can be found among supporting and regulating services, while tradeoffs exist among provisioning and supporting services. Natural lands embedded in agricultural landscapes may result in lower regional production values, but they provide important regulating services of N loading reduction in some areas (Georgia) and critical habitat for biodiversity in others (Florida).

The degree to which LTAR Network is representative of US agriculture is the subject of intense work (Bean et al., 2021). The 18+ LTAR sites across the national network vary considerably in production system, physio-geographic setting, land-use histories, and drivers of change. This study is not presented as a new scaling method for LTAR analysis and synthesis. But ideas here should challenge future analyses of synergies and trade-offs among ecosystem services across LTAR and other national networks, suggesting how to handle issues of scale and landscape complexity in agro-ecosystems. For example, by using scaling methods one can avoid making conclusions about a small area based upon aggregated results, thus avoiding the "ecological fallacy" (Wong, 2008).

Future Directions and the Case for Working Lands in Ecosystem Services Research

An important distinction of the SE is that the proportion of developed land classes is far higher than in the rest of the

US, likely driven by high rates of urbanization, especially in the CPEco, where it is expected to increase in the coming decades (Zhao et al., 2013). Land use is highly dynamic compared with the rest of the US (Sleeter et al., 2013), and some land cover changes are recurrent processes such as in forested areas where land use is heavily focused on silviculture (Drummond et al., 2015; Marsik et al., 2018). Analyses and forecasting of changing ecosystem services from coupled agricultural-natural land covers in the Southeast will have to account for the drivers of land use intensification, in addition to climate change. Indeed, landscape approaches to balancing land uses are refocusing from environment and development tradeoffs to increasing inclusion of societal concerns (Sayer et al., 2013).

The consideration of low intensity working lands and their role in delivering and protecting ecosystem services could be a major contribution to planning future land use, including the sustainable intensification of agriculture (e.g., Rockström et al., 2017), increasing carbon storage, and reducing greenhouse gas emissions (Fargione et al., 2018; Sanderson et al., 2020). Expanding our understanding of the "ecosystem services matrix" of natural habitats combined with working lands allows us to recognize the roles of working lands, such as habitat, for large area-requiring species like top predators. Low intensity working lands are not as biodiverse as the lands they replace, but higher "countryside" ecosystem service values might improve our understanding of how to balance production with agroecosystem conservation (Vanslembrouck and Van Huylenbroeck, 2005). We also gain a better appreciation for the extensive landscapes over which large scale ecosystem processes such as prescribed fires and floods may occur.

Explicitly including contributions of working lands is important for natural capital ecosystem accounting, such as the National Ecosystem Services Classification System (https://seea.un.org/home/Natural-Capital-Accounting-Project) (Olander et al., 2017), enabling tradeoffs from working lands to be assessed more clearly. In a recent application of natural capital accounting, analysis of trends in ecosystem extent, condition, and ecosystem services supply and use accounts were prepared for a 10-state region in the Southeast by Warnell et al. (2020), using extensive ecosystem service indicators such as bird species richness, wild pollinator habitat, and natural habitats that may purify water.

Our analysis was restricted to the southeastern US. Although most of these working lands are neither conserved nor publicly protected, the average proportion of natural and low intensity working lands in HUC-12s across the SE (77%), exceeds the ambitious goal of the Half-Earth Project, which is working to conserve half the land and sea to safeguard the bulk of the world's biodiversity (Wilson, 2016). Conducting similar analyses of agroecosystems across the continental US could provide new and interesting comparative indicators with which to assess ecosystem services, understand responses to drivers of change, and evaluate potential outcomes of alternative scenarios. Using an approach like the one developed here would allow scientists to array agroecosystems along gradients, quantifying the tradeoffs and synergies of ecosystem services across multiple scales, and informing our understanding of the dynamics of ecosystem services in working lands.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available in **Supplementary Material 2—Data Table**, and GIS layers are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

AC, HS, and VS conceived of and designed the research. VS, AC, GP-C, and LS acquired and analyzed data. AC, HS, VS, GP-C, and LS wrote and edited manuscript. All authors contributed to the article and approved the submitted version.

FUNDING

This research was a contribution from the Long-Term Agroecosystem Research (LTAR) Network. LTAR is supported by the United States Department of Agriculture.

ACKNOWLEDGMENTS

We wish to thank Chloe Dela Cerna and Coby Smith for support with data compilation and summaries and graphics development. We also thank David Bosch, Betsey Boughton, and Xukai Zhang for access to local data on water quality and primary production in the local LTAR sites. We are also grateful to USDA-ARS and peer reviewers for their comments, improving the quality of this manuscript.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2021.541590/full#supplementary-material

Presentation 1 | This file contains **Supplementary Materials 1**, including a .docx file with: Tables 1.1–1.6; Figures 1.1, 1.2; and detailed methods describing the Net Primary Production (NPP) data analysis, and the fragmentation geometry (FG) analysis and landscape metrics.

Presentation 2 | This file contains **Supplementary Materials 2—Data Table**, including all of the data used in this analysis in the form of a spreadsheet (.xlsx).

Presentation 3 | This file contains **Supplementary Materials 3**, including: a .docx file with Tables 3.1–3.7 providing legend and figure number information for 216 pairwise comparisons of variables; a folder of 216 .jpg images of pairwise comparisons of variables.

Presentation 4 | This file contains **Supplementary Materials 4**, including: a .docx file with R code and Tables 4.1–4.9 Anderson-Darling statistics (section 1) of kernel density function comparisons, Table 4.10, a general legend for all figures in the supplement (section 2), and Table 4.11, file names and titles for all figures in the supplement (section 2); and a folder of 18 .jpg images of kernel density functions.

REFERENCES

Anderson, T. W., and Darling, D. A. (1954). A test of goodness of fit. *J. Am. Statist. Assoc.* 49, 765–769. doi: 10.1080/01621459.1954.10501232

Bean, A. R., Coffin, A. W., Arthur, D. K., Baffaut, C., Holifield Collins, C., Goslee, S. C., et al. (2021). Regional frameworks for the USDA long-term agroecosystem research (LTAR). *Front. Sustain. Food Systems.* 4:612785. doi: 10.3389/fsufs.2020.612785

Beck, H. E., Zimmermann, N. E., Mcvicar, T. R., Vergopolan, N., Berg, A., and Wood, E. F. (2018). Present and future Köppen-Geiger climate classification maps at 1-km resolution. *Sci. Data* 5, 180214–180214. doi: 10.1038/sdata.2018.214

Bohlen, P. J., and Villapando, O. R. (2011). Controlling runoff from subtropical pastures has differential effects on nitrogen and phosphorus loads. *J. Environ. Quality* 40, 989–998. doi: 10.2134/jeq2010.0127

Boryan, C., Yang, Z., Mueller, R., and Craig, M. (2011). Monitoring US agriculture: the US Department of Agriculture, National Agricultural Statistics Service, Cropland Data Layer Program. *Geocarto Int.* 26, 341–358. doi: 10.1080/10106049.2011.562309

Bosch, D. D., Pisani, O., Coffin, A. W., and Strickland, T. C. (2020). Water quality and land cover in the Coastal Plain Little River watershed, Georgia, United States. *J. Soil Water Conserv.* 75, 263–277. doi: 10.2489/jswc.75.3.263

Bosch, D. D., and Sheridan, J. M. (2007). Stream discharge database, Little River Experimental Watershed, Georgia, United States. *Water Resourc. Res.* 43:5833. doi: 10.1029/2006WR005833

Bosch, D. D., Sheridan, J. M., Lowrance, R. R., Hubbard, R. K., Strickland, T. C., Feyereisen, G. W., et al. (2007). Little river experimental watershed database. *Water Resourc. Res.* 43:5844. doi: 10.1029/2006WR005844

Boughton, E. H., Quintana-Ascencio, P. F., Jenkins, D. G., Bohlen, P. J., Fauth, J. E., Engel, A., et al. (2019). Trade-offs and synergies in a payment-for-ecosystem services program on ranchlands in the Everglades headwaters. *Ecosphere* 10:e02728. doi: 10.1002/ecs2.2728

Burrell, A. (2001). Multifunctionality and agricultural trade liberalisation. *J. Tijdschrift voor Sociaalwetenschappelijk Onderzoek van de Landbouw* 16, 77–95.

Ceia-Hasse, A., Borda-De-Água, L., Grilo, C., and Pereira, H. M. (2017). Global exposure of carnivores to roads. *Glob. Ecol. Biogeogr.* 26, 592–600. doi: 10.1111/geb.12564

Chamberlain, S. D., Gomez-Casanovas, N., Walter, M. T., Boughton, E. H., Bernacchi, C. J., Delucia, E. H., et al. (2016). Influence of transient flooding on methane fluxes from subtropical pastures. *J. Geophys. Res. Biogeosci.* 121, 965–977. doi: 10.1002/2015JG003283

Chen, E., and Gerber, J. F. (1990). "Climate," in *Ecosystems of Florida*, eds R. L. Myers, and J. J. Ewel (Orlando, FL: University of Central Florida Press) 11–34.

De Groot, R. S., Alkemade, R., Braat, L., Hein, L., and Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol. Complexity* 7, 260–272. doi: 10.1016/j.ecocom.2009.10.006

Drummond, M. A., Stier, M. P., Auch, R. F., Taylor, J. L., Griffith, G. E., Riegle, J. L., et al. (2015). Assessing landscape change and processes of recurrence, replacement, and recovery in the Southeastern Coastal Plains, USA. *Environ. Manag.* 56, 1252–1271. doi: 10.1007/s00267-015-0574-1

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Ann. Rev. Ecol. Evol. Systemat.* 34, 487–515. doi: 10.1146/annurev.ecolsys.34.011802.132419

Fahrig, L. (2017). Ecological responses to habitat fragmentation *per se*. *Ann. Rev. Ecol. Evol. Systemat.* 48, 1–23. doi: 10.1146/annurev-ecolsys-110316-022612

Fahrig, L. (2019). Habitat fragmentation: a long and tangled tale. *Glob. Ecol. Biogeogr.* 28, 33–41. doi: 10.1111/geb.12839

Fargione, J. E., Bassett, S., Boucher, T., Bridgham, S. D., Conant, R. T., Cook-Patton, S. C., et al. (2018). Natural climate solutions for the United States. *Sci. Adv.* 4:eaat1869. doi: 10.1126/sciadv.aat1869

Farr, J., and Brock, O. (2006). Florida's landmark programs for conservation and recreation land acquisition. *Sustain* 14:35–44.

Forman, R. T. T. (1995). *Land Mosaics: The Ecology of Landscapes and Regions*. Cambridge MA: Cambridge University Press. doi: 10.1017/9781107050327

Fotheringham, A. S., and Wong, D. W. S. (1991). The modifiable areal unit problem in multivariate statistical analysis. *Environ. Plan. A* 23, 1025–1044. doi: 10.1068/a231025

Georgia Department of Natural Resources (2020). *Rare Natural Elements by Locations*. Georgia Biodiversity Portal, Wildlife Resources Division, Wildlife Conservation Section, Social Circle. Available online at: https://www.georgiabiodiversity.a2hosted.com/natels/natural-element-locations?group=all_groups (accessed November 6, 2019)

Gomez-Casanovas, N., Delucia, N. J., Bernacchi, C. J., Boughton, E. H., Sparks, J. P., Chamberlain, S. D., et al. (2018). Grazing alters net ecosystem C fluxes and the global warming potential of a subtropical pasture. *Ecol. Appl.* 28, 557–572. doi: 10.1002/eap.1670

Gotway, C. A., and Young, L. J. (2002). Combining incompatible spatial data. *J. Am. Statist. Assoc.* 97, 632–648. doi: 10.1198/016214502760047140

Haberl, H., Erb, K. H., Krausmann, F., Gaube, V., Bondeau, A., Plutzar, C., et al. (2007). Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. *Proc. Natl. Acad. Sci.* 104, 12942–12947. doi: 10.1073/pnas.0704243104

Heffernan, J. B., Soranno, P. A., Angilletta M. J. Jr., Buckley, L. B., Gruner, D. S., Keitt, T. H., et al. (2014). Macrosystems ecology: understanding ecological patterns and processes at continental scales. *Front. Ecol. Environ.* 12, 5–14. doi: 10.1890/130017

Jaeger, J. A. G. (2000). Landscape division, splitting index, and effective mesh size: new measures of landscape fragmentation. *Landscape Ecol.* 15, 115–130. doi: 10.1023/A:1008129329289

Jaeger, J. A. G., Bertiller, R., Schwick, C., Müller, K., Steinmeier, C., Ewald, K. C., et al. (2008). Implementing landscape fragmentation as an indicator in the Swiss monitoring system of sustainable development (Monet). *J. Environ. Manag.* 88, 737–751. doi: 10.1016/j.jenvman.2007.03.043

Jin, S., Homer, C., Yang, L., Danielson, P., Dewitz, J., Li, C., et al. (2019). Overall methodology design for the United States National Land Cover Database 2016 products. *Remote Sensing* 11:2971. doi: 10.3390/rs11242971

Kleinman, P. J. A., Spiegal, S., Rigby, J. R., Goslee, S. C., Baker, J. M., Bestelmeyer, B. T., et al. (2018). Advancing the sustainability of US agriculture through long-term research. *J. Environ. Quality* 47, 1412–1425. doi: 10.2134/jeq2018.05.0171

Knight, A. T., and Cowling, R. M. (2007). Embracing opportunism in the selection of priority conservation areas. *Conserv. Biol.*21, 1124–1126. doi: 10.1111/j.1523-1739.2007.00690.x

Lark, T. J., Spawn, S. A., Bougie, M., and Gibbs, H. K. (2020). Cropland expansion in the United States produces marginal yields at high costs to wildlife. *Nat. Commun.* 11, 4295. doi: 10.1038/s41467-020-18045-z

Lofting, H. (1920). *The Story of Doctor Dolittle*. New York, NY: Frederick A. Stokes Company.

Lowrance, R., Todd, R., Fail, J., Hendrickson, O., Leonard, R., and Asmussen, L. (1984). Riparian forests as nutrient filters in agricultural watersheds. *BioScience* 34, 374–377. doi: 10.2307/1309729

Maleski, J. J., Bosch, D. D., Anderson, R. G., Coffin, A. W., Anderson, W. F., and Strickland, T. C. (2019). Evaluation of miscanthus productivity and water use efficiency in southeastern United States. *Sci. Total Environ.* 692, 1125–1134. doi: 10.1016/j.scitotenv.2019.07.128

Marsik, M., Staub, C., Kleindl, W., Fu, C. -S., Hall, J., Yang, D., et al. (2017). "Regional-scale forest management maps for the continental United States," in: Supplement to: Marsik, M et al. (2018): Regional-Scale Forest Management Maps for the Continental United States. *Sci. Data* 5:180165. doi: 10.1038/sdata.2018.165

Marsik, M., Staub, C. G., Kleindl, W. J., Hall, J. M., Fu, C. -S., Yang, D., et al. (2018). Regional-scale management maps for forested areas of the Southeastern United States and the US Pacific Northwest. *Sci. Data* 5:180165.

McGarigal, K., and Marks, B. (1995). *FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure*. Portland, OR: USDA Forest Service. doi: 10.2737/PNW-GTR-351

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being: A Framework for Assessment*. Washington, DC: World Resources Institute.

Murphy, S. M., Augustine, B. C., Ulrey, W. A., Guthrie, J. M., Scheick, B. K., Mccown, J. W., et al. (2017). Consequences of severe habitat fragmentation on density, genetics, and spatial capture-recapture analysis of a small bear population. *PLoS ONE* 12:e0181849. doi: 10.1371/journal.pone.0181849

Olander, L., Bagstad, K., Characklis, G. W., Comer, P., Effron, M., Gunn, J., et al. (2017). *Data and Modeling Infrastructure for National Integration of Ecosystem Services into Decision Making: Expert Summaries*. Durham, NC: National Ecosystem Services Partnership.

Omernik, J., and Griffith, G. (2014). Ecoregions of the conterminous United States: evolution of a hierarchical spatial framework. *Environ. Manag.* 54, 1249–1266. doi: 10.1007/s00267-014-0364-1

Pearlstine, L. G., Smith, S. E., Brandt, L. A., Allen, C. R., Kitchens, W. M., and Stenberg, J. (2002). Assessing state-wide biodiversity in the Florida Gap analysis project. *J. Environ. Manag.* 66, 127–144. doi: 10.1006/jema.2002.0551

Petersen, B., and Snapp, S. (2015). What is sustainable intensification? Views from experts. *Land Use Policy* 46, 1–10. doi: 10.1016/j.landusepol.2015.02.002

Pisani, O., Bosch, D. D., Coffin, A. W., Endale, D. M., Liebert, D., and Strickland, T. C. (2020). Riparian land cover and hydrology influence stream dissolved organic matter composition in an agricultural watershed. *Sci. Total Environ.* 717:137165. doi: 10.1016/j.scitotenv.2020.137165

Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philos. Trans. Royal Soc. B Biol. Sci.* 365, 2959–2971. doi: 10.1098/rstb.2010.0143

Pretty, J., Sutherland, W. J., Ashby, J., Auburn, J., Baulcombe, D., Bell, M., et al. (2010). The top 100 questions of importance to the future of global agriculture. *Int. J. Agri. Sustain.* 8, 219–236. doi: 10.3763/ijas.2010.0534

R Core Team (2019). *R: A language and environment for statistical computing*. Vienna: R Foundation for Statistical Computing.

Rabalais, N. N., Díaz, R. J., Levin, L. A., Turner, R. E., Gilbert, D., and Zhang, J. (2010). Dynamics and distribution of natural and human-caused hypoxia. *Biogeosciences* 7, 585–619. doi: 10.5194/bg-7-585-2010

Rissman, A. R., Morris, A. W., Kalinin, A., Kohl, P. A., Parker, D. P., and Selles, O. (2019). Private organizations, public data: land trust choices about mapping conservation easements. *Land Use Policy* 89:104221. doi: 10.1016/j.landusepol.2019.104221

Robertson, P. G., Gross, K. L., Hamilton, S. K., Landis, D. A., Schmidt, T. M., Snapp, S. S., et al. (2014). Farming for ecosystem services: an ecological approach to production agriculture. *BioScience* 64, 404–415. doi: 10.1093/biosci/biu037

Robinson, N. P., Allred, B. W., Smith, W. K., Jones, M. O., Moreno, A., Erickson, T. A., et al. (2018). Terrestrial primary production for the conterminous United States derived from Landsat 30 m and MODIS 250 m. *Remote Sens. Ecol. Conserv.* 4, 264–280. doi: 10.1002/rse2.74

Rockström, J., Williams, J., Daily, G., Noble, A., Matthews, N., Gordon, L., et al. (2017). Sustainable intensification of agriculture for human prosperity and global sustainability. *Ambio* 46, 4–17. doi: 10.1007/s13280-016-0793-6

Ruimy, A., Saugier, B., and Dedieu, G. (1994). Methodology for the estimation of terrestrial net primary production from remotely sensed data. *J. Geophys. Res. Atmos.* 99, 5263–5283. doi: 10.1029/93JD03221

Running, S. W., Thornton, P. E., Nemani, R., and Glassy, J. M. (2000). "Global terrestrial gross and net primary productivity from the earth observing system," in *Methods in Ecosystem Science,* eds O. E. Sala, R. B. Jackson, H. A. Mooney, and R. W. Howarth (New York, NY: Springer New York), 44–57. doi: 10.1007/978-1-4612-1224-9_4

Rytwinski, T., and Fahrig, L. (2013). Why are some animal populations unaffected or positively affected by roads? *Oecologia* 173, 1143–1156. doi: 10.1007/s00442-013-2684-x

Sanderson, J. S., Beutler, C., Brown, J. R., Burke, I., Chapman, T., Conant, R. T., et al. (2020). Cattle, conservation, and carbon in the western Great Plains. *J. Soil Water Conserv*. 75, 5A—12A. doi: 10.2489/jswc.75.1.5A

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J.-L., Sheil, D., Meijaard, E., et al. (2013). Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. *Proc. Natl. Acad. Sci.* 110, 8349–8356. doi: 10.1073/pnas.1210595110

Schmiedel, I., and Culmsee, H. (2016). The influence of landscape fragmentation, expressed by the "Effective Mesh Size Index," on regional patterns of vascular plant species richness in Lower Saxony, Germany. *Landscape Urban Plan.* 153, 209–220. doi: 10.1016/j.landurbplan.2016.01.012

Scholz, F. W., and Stephens, M. A. (1987). K-sample Anderson-Darling tests. *J. Am. Statist. Assoc.* 82, 918–924. doi: 10.1080/01621459.1987.10478517

Scholz, F. W., and Zhu, A. (2019). "*kSamples: K-Sample Rank Tests and their Combinations." R package version 1.2-9 ed.* Available online at: https://cran.r-project.org/package=kSamples

Silverman, B.W. (1986). *Density Estimation*. London: Chapman and Hall. doi: 10.1007/978-1-4899-3324-9

Sleeter, B. M., Sohl, T. L., Loveland, T. R., Auch, R. F., Acevedo, W., Drummond, M. A., et al. (2013). Land-cover change in the conterminous United States from 1973 to 2000. *Glob. Environ. Change* 23, 733–748. doi: 10.1016/j.gloenvcha.2013.03.006

Smith, W. K., Zhao, M., and Running, S. W. (2012). Global bioenergy capacity as constrained by observed biospheric productivity rates. *BioScience* 62, 911–922. doi: 10.1525/bio.2012.62.10.11

Sohl, T. L., Loveland, T. R., Sleeter, B. M., Sayler, K. L., and Barnes, C. A. (2010). Addressing foundational elements of regional land-use change forecasting. *Landscape Ecol.* 25, 233–247. doi: 10.1007/s10980-009-9391-3

Southworth, J., Cumming, G. S., Marsik, M., and Binford, M. W. (2006). Linking spatial and temporal variation at multiple scales in a heterogeneous landscape. *Professional Geographer*. 58, 406–420. doi: 10.1111/j.1467-9272.2006.00578.x

Spiegal, S., Bestelmeyer, B. T., Archer, D. W., Augustine, D. J., Boughton, E. H., Boughton, R. K., et al. (2018). Evaluating strategies for sustainable intensification of US agriculture through the Long-Term Agroecosystem Research network. *Environ. Res. Lett.* 13:034031. doi: 10.1088/1748-9326/aaa779

Swain, H. M., Boughton, E. H., Bohlen, P. J., and Lollis, L. O. G. (2013). Trade-offs among ecosystem services and disservices on a Florida ranch. *Rangelands* 35, 75–87. doi: 10.2111/RANGELANDS-D-13-00053.1

U.S. Geological Survey (USGS) Gap Analysis Project (GAP) (2018). *Protected Areas Database of the United States (PAD-US): U.S. Geological Survey data release*. doi: 10.5066/P955KPLE

US EPA (2011). *EnviroAtlas - EnviroAtlas - NatureServe Analysis of Imperiled or Federally Listed Species by HUC-12 for the Conterminous United States*. Research Triangle Park, NC: US EPA.

USGS, USDA-NRCS, and US EPA (2015). *National Watershed Boundary Dataset (WBD)*. http://datagateway.nrcs.usda.gov (accessed January 25, 2021).

Vanslembrouck, I., and Van Huylenbroeck, G. (2005). Landscape Amenities: Economic *Assessent of Agricultural Landscapes*. Netherlands: Springer. doi: 10.1007/1-4020-3172-6

Warnell, K. J. D., Russell, M., Rhodes, C., Bagstad, K. J., Olander, L. P., Nowak, D. J., et al. (2020). Testing ecosystem accounting in the United States: a case study for the Southeast. *Ecosyst. Services* 43:101099. doi: 10.1016/j.ecoser.2020.101099

Wilson, E. O. (2016). *Half-Earth: Our Planet's Fight for Life.* New York, NY: Liveright Publishing Corporation, a division of W. W. Norton & Company.

Wong, D. (2008). "Ecological fallacy," in *Encyclopedia of geographic information science*, ed K. K. Kemp (SAGE Publications, Inc.) 119–120. doi: 10.4135/9781412953962.n53

Wu, J., Jelinski, D. E., Luck, M., and Tueller, P. T. (2000). Multiscale analysis of landscape heterogeneity: scale variance and pattern metrics. *Geogr. Inform. Sci.* 6, 6–19. doi: 10.1080/10824000009480529

Xavier, S., Olson, D., Coffin, A., Strickland, T., and Schmidt, J. (2017). Perennial grass and native wildflowers: a synergistic approach to habitat management. *Insects* 8:104. doi: 10.3390/insects8040104

Yang, L., Jin, S., Danielson, P., Homer, C., Gass, L., Bender, S. M., et al. (2018). A new generation of the United States National Land Cover Database: requirements, research priorities, design, and implementation strategies. *ISPRS J. Photogrammetr. Remote Sensing* 146, 108–123. doi: 10.1016/j.isprsjprs.2018.09.006

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., and Swinton, S.M. (2007). Ecosystem services and dis-services to agriculture. *Ecol. Econ.* 64, 253–260. doi: 10.1016/j.ecolecon.2007.02.024

Zhao, S., Liu, S., Sohl, T., Young, C., and Werner, J. (2013). Land use and carbon dynamics in the southeastern United States from 1992 to 2050. *Environ. Res. Lett.* 8:044022. doi: 10.1088/1748-9326/8/4/044022

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Advantages of publishing in Frontiers

OPEN ACCESS
Articles are free to read for greatest visibility and readership

FAST PUBLICATION
Around 90 days from submission to decision

HIGH QUALITY PEER-REVIEW
Rigorous, collaborative, and constructive peer-review

TRANSPARENT PEER-REVIEW
Editors and reviewers acknowledged by name on published articles

Frontiers
Avenue du Tribunal-Fédéral 34
1005 Lausanne | Switzerland

Visit us: www.frontiersin.org
Contact us: frontiersin.org/about/contact

REPRODUCIBILITY OF RESEARCH
Support open data and methods to enhance research reproducibility

DIGITAL PUBLISHING
Articles designed for optimal readership across devices

FOLLOW US
@frontiersin

IMPACT METRICS
Advanced article metrics track visibility across digital media

EXTENSIVE PROMOTION
Marketing and promotion of impactful research

LOOP RESEARCH NETWORK
Our network increases your article's readership