DIVERSIFYING FARMING SYSTEMS FOR ADAPTIVE CAPACITY

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DIVERSIFYING FARMING SYSTEMS FOR ADAPTIVE CAPACITY

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EDITORIAL
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Editorial: Diversifying Farming Systems for Adaptive Capacity

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Editorial on the Research Topic

Diversifying Farming Systems for Adaptive Capacity

The COVID-19 pandemic underscores how novel shocks to agrifood systems compound the already dire global impacts of climate change, biodiversity loss, and food insecurity. Agrifood systems are not merely threatened by these interconnected crises—they drive them. In the pursuit of maximizing yield and profit, the ecologies, cultures, and markets that comprise our food systems have been made ever more uniform. Despite broad scientific agreement that agrifood systems must transform to mitigate climate change, conserve biodiversity and natural resources, and provide food and nutrition security in sustainable and equitable ways (Kremen and Merenlender, 2018; Shukla, 2019), much research remains agnostic toward *how* to do this. This collection of articles targets this gap by examining the social and ecological structures that shape adaptive capacity in farming systems and determine whether adaptive processes advance sustainability, resilience, and equity.

Agricultural adaptive capacity represents the extent to which farming systems can respond to climate change, biodiversity loss, and food insecurity in ways that not only preserve but holistically enhance their social-ecological functions. Petersen-Rockney et al. present a framework to analyze the relationship between farming system processes and adaptive capacity. By applying this framework to five diverse cases of agrifood system stressors, the authors find that adaptive capacity exhibits divergent qualities, such as heightened reliance on external inputs, in contrast to strengthening *in-situ* ecosystem service provisioning or place-based expertise, depending on the processes through which it emerges. Critically, the way we adapt to crises today shapes the range of possibilities for future adaptation. Returning to the former, usually simplified, state in response to crisis (i.e., a classic resilience perspective) may perpetuate crises. Instead, adaptations that diversify farming systems serve as long term investments from which robust in nimble adaptive capacity can emerge, helping social-ecological systems grow in more sustainable and equitable ways.

The 11 articles in this collection expand on these core insights, highlighting how adaptive capacity operates at various scales and levels, and emerges from diversification processes across ecological, social, and institutional dimensions of farming systems.

At the field and farm levels, diversification entails leveraging synergies and reducing tradeoffs among ecosystem, nutritional, and economic functions. Classically, farms can diversify through farming practices that strategically increase biodiversity across time and space to provision ecosystem services (Basche et al.; Stratton et al.). Diversification through practices like crop rotation can facilitate adaptive capacity to stressors like drought (Wauters et al.; Mortensen and Smith). For instance, Renwick et al. demonstrate how intercropping maize with a legume increases overall yield, improves nutritional quality, and increases adaptive capacity in the face of extreme weather—regardless of fertilizer availability. Renewed attention to crop maturity cycles offers another strategy to correct

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myopic focus on fast-maturing annual crops through reintroduction of long-duration (i.e., produce for several seasons), indeterminant (i.e., continuously producing), and perennial crops (Snapp).

Particularly among smallholders, the strategic (re)introduction of "neglected and underutilized crops" can balance on-farm nutritional needs with cash crop production, bolstering food sovereignty opportunities (van Zonneveld et al.). Q'eqchi' Maya smallholders who implemented ecological adaptation strategies, such as cover cropping or agroforestry, were better able to meet both nutritional and ecological goals compared to other farmers (Stratton et al.). Even conventional, highly industrialized farms can improve their ability to meet multiple goals simultaneously by, for example, incorporating lifecycle and habitat considerations for honeybees and other organisms (Durant and Ponisio).

Contributing authors also emphasized the importance of diversifying markets, values and goals, knowledge systems, farm business and management models, governing institutions and policies, and farmer demographics for improving adaptive capacity (Petersen-Rockney et al.). Multifunctional goals can help build structural diversification iteratively, aiding the emergence of adaptive capacity over time (Snapp). In contexts where multiple goals are prioritized together, coordinated for example through robust social networks and governance capacity, "total loss appears to provide a window of opportunity for reinventing agricultural systems" (Rodriguez-Cruz et al.). van Zonneveld et al. identify strengthening farmer-to-farmer networks, magnifying "lighthouse farmers" (e.g., custodians of traditional agroecological knowledge), and empowering women farmers as preconditions for on-farm diversification that safeguards household food security. Transformative potential also depends on the diversity of claims and claimants to resources like farmland (Calo). This point illuminates an implied corollary to van Zonneveld et al.'s argument that diversification begins with farmer goals: on-farm diversity is constrained by the diversity of functions that are sought from the land, and therefore on the diversity of people who have control over land use decisions.

While research articles in this collection find considerable evidence for the emergence of more broad and nimble adaptive capacity through diversifying processes, they also identify critical barriers to diversification. Mortensen and Smith emphasize the need to address factors that lock farmers into monocultures and narrow the range of adaptive possibilities. Snapp shows how maladaptive policies—such as India's Public Distribution System that "privileges" wheat and rice monocultures in the name of calorie-centric food security—lead to disabling lock-in by pushing farmers into dependence on fossil fuels and chemical pest regulation. Basche et al. critique institutions that focus too narrowly on measuring singular agrifood production functions, calling instead for multifunctional metrics and investments that value a plurality of environmental and soil health co-benefits. Structural forces like land tenure regimes and crop incentive policies constrain individual farmers' agency (Rodriguez-Cruz et al.; Calo).

Contributing authors consistently agree that for broad and nimble adaptive capacity to emerge from farming systems, diversification efforts must embrace regionally specific messaging (Durant and Ponisio), site-specific implementation (Wauters et al.), and the unique contexts and goals of individual farmers (van Zonneveld et al.). Such multi-faceted and context-specific approaches can better identify synergies, as well as tradeoffs, between, for example, ecological and nutritional resilience functions (Stratton et al.).

As the articles in this collection demonstrate, diversifying farming systems recognizes a spectrum of socio-ecological practices that farmers and other agrifood workers can flexibly employ to increase adaptive capacity. Yet critical questions remain. Future work should assess diversifying approaches beyond the farm scale that can complement economic diversification of crop portfolios and biodiversification of genetic assets and ecosystem functions. Calo, for example, highlights the potential to diversify property relationships, while other authors nod toward market and insurance diversification (van Zonneveld et al.), heterogeneity of social networks and informal relationships (Rodriguez-Cruz et al.), diverse forms of knowledge and expertise (Petersen-Rockney et al.; van Zonneveld et al.), and new criteria and metrics for policy formation (Snapp; Basche et al.).

An important intervention this collection offers is distinguishing the *state* of being diversified or simplified from the *processes* of diversification and simplification. Obstacles to building socially just and environmentally sustainable adaptive capacity loom large. And low rates of adoption of diversification practices, such as basic legally required Best Management Practices among almond growers in California (Durant and Ponisio), may appear disheartening. However, focusing on processes helps illuminate pathways of opportunity. Instead of framing farmers as the "problem" in need of a "fix" through education, incentives, and regulation, we join Calo in suggesting a need to address the root causes of farming system vulnerability and inequity. We encourage ourselves, other scholars, and you, the reader, to engage in reflexive inquiry that always asks, adaptive capacity by whom and for whom?

AUTHOR CONTRIBUTIONS

MP-R suggested the Research Topic and outlined this editorial. MP-R, TB, SA, and PB met and discussed the topic and all contributions. All authors closely reviewed several articles in the collection, contributed points to the drafting of this text, edited this document, and approved its submission.

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Decision-Making to Diversify Farm Systems for Climate Change Adaptation

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On-farm diversification is a promising strategy for farmers to adapt to climate change. However, few recommendations exist on *how* to diversify farm systems in ways that best fit the agroecological and socioeconomic challenges farmers face. Farmers' ability to adopt diversification strategies is often stymied by their aversion to risk, loss of local knowledge, and limited access to agronomic and market information, this is especially the case for smallholders. We outline seven steps on *how* practitioners and researchers in agricultural development can work with farmers in decision-making about on-farm diversification of cropping, pasture, and agroforestry systems while taking into account these constraints. These seven steps are relevant for all types of farmers but particularly for smallholders in tropical and subtropical regions. It is these farmers who are usually most vulnerable to climate change and who are, subsequently, often the target of climate-smart agriculture (CSA) interventions. Networks of agricultural innovation provide an enabling environment for on-farm diversification. These networks connect farmers and farmer organizations with local, national, or international private companies, public organizations, non-governmental organizations (NGOs), and research institutes. These actors can work with farmers to develop diversified production systems incorporating both high-value crops and traditional food production systems. These diversified farm systems with both food and cash crops act as a safety net in the event of price fluctuations or other disruptions to crop value chains. In this way, farmers can adapt their farm systems to climate change in ways that provide greater food security and improved income.

Keywords: on-farm diversification, agroecosystem diversification, climate-smart agriculture, climate variability, crop diversification, diversified farming systems, participatory research, risk management

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INTRODUCTION

On-farm diversification is a promising strategy for farmers to adapt to climate change while also contributing to diverse food production, healthier diets, and a better use of agricultural biodiversity (Vermeulen et al., 2012; Waha et al., 2018; Willett et al., 2019). However, few recommendations exist for farmers, practitioners, and researchers on *how* to diversify farm systems in ways which best fit the agroecological and socioeconomic challenges that farmers face.

In this paper, we outline seven steps on *how* to work with farmers in decision-making about on-farm diversification of cropping, pasture, and agroforestry systems (**Figure 1**). Existing tools

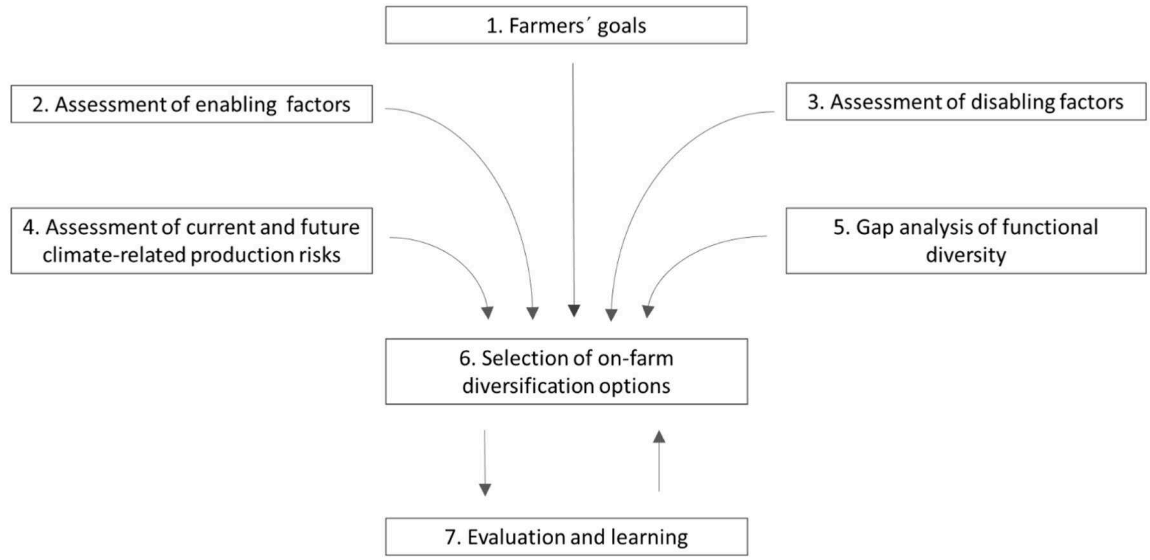

FIGURE 1 | Decision-making framework to develop, select, evaluate, and implement on-farm diversification strategies for climate change adaptation. We propose seven steps for practitioners and researchers to work with farmers in decision-making. Steps 1–5 help farmers and other actors to collect information to select on-farm diversification options in step 6. After the selection of on-farm diversification options, farmers can evaluate them in step 7 and implement or adjust them or replace them by other on-farm diversification options. This is reflected in a feedback loop between step 6 and 7. The arrows indicate which steps influence other steps in the decision-making framework.

to select agroecological practices and plant species for on-farm diversification (Altieri et al., 2015; de Sousa et al., 2019) or to economically optimize crop portfolios (Werners et al., 2011; Knoke et al., 2015) cover different considerations in decision-making on on-farm diversification strategies. These tools are not always linked to farmers' goals and constraints, which are embedded in a range of social, economic, ecological, cultural, and political relationships, and which determine the decisions farmers make about farm management and livelihood options (Gardner and Lewis, 1996; Shiferaw et al., 2009). This paper offers a practical and comprehensive framework, which takes into account these different issues in decision-making about on-farm diversification, and which brings together agroecological, agrobotanical, social, and economic considerations and recommendations.

This decision-making framework is intended for practitioners and researchers in agricultural development. The framework can be used to establish a dialogue with individual farmers or farmer groups to develop on-farm diversification strategies with the use of participatory research approaches, which have proved to be successful approaches in the selection and adoption of new agricultural technologies (Carberry et al., 2002; Grothmann and Patt, 2005; Urwin and Jordan, 2008).

In this framework, we first discuss enabling and disabling factors, which warrant consideration when developing on-farm diversification strategies. Second, we propose straightforward tools and techniques, which can help farmers to select on-farm diversification options. Finally, we explain how researchers, practitioners, and farmers can apply participatory approaches to evaluate on-farm diversification options.

The seven steps are useful for all types of farmer but are particularly relevant to smallholder farmers. Smallholders are often more vulnerable to climate change compared with large-scale farmers and usually face higher risks when adopting new technologies because of lower resource endowments. Smallholder farmers are the main target of interventions, which are collectively known as climate-smart agriculture (CSA). CSA contributes to an increase in global food security and broader development, secondly enhances farmers' ability to adapt to climate change, and finally mitigates greenhouse gas emissions (Lipper et al., 2014). On-farm diversification is a component of CSA, and not only contributes to the realization of the Sustainable Development Goal (SDG) 13: *Climate Action* but also of other SDGs, including SDG 1: *No Poverty*; SDG 2: *Zero Hunger*; SDG 12 *Responsible Consumption and Production*; and SDG 15 *Life on Land*.

APPROACH

The seven steps resulted from the authors' discussions on existing concepts and tools from literature on climate change adaptation and on-farm diversification. These concepts have been presented separately in literature. By connecting these concepts, we establish a practical framework for decision-making to diversify farm systems for climate change adaptation. We focus on tropical and subtropical regions where most smallholders live and work, and on cropping, pasture, and agroforestry systems as principal components of farm systems in these regions. Many examples of crops and traditional production systems in this paper come from Central America and Mexico where each of the authors has over 12 years' work experience complemented by extensive experience from Sub-Saharan Africa, South Asia, and South-East Asia. The seven-step decision-making process is applicable to other tropical and subtropical regions.

Step 1. Defining farmers' goals: Any initiative to work with farmers starts with understanding the goals of the different farm household members, and identifying how on-farm diversification can contribute to these goals (Allen et al., 2011).

Step 2. Assessment of enabling factors: Enabling factors determine the feasibility and potential of on-farm diversification options. Farmers are more willing to select, evaluate, and implement new diversification strategies in the context of an enabling environment consisting of support from farmer organizations and private and public extension services, and access to credit, insurance, and markets.

Step 3. Assessment of disabling factors: Successful adoption of on-farm diversification strategies depends on the extent to which farmers have the possibility and are willing to invest in labor, financial capital, and learning new skills.

Step 4. Assessment of current and future climate-related production risks: On-farm diversification strategies can be tailored to local conditions when farmers, practitioners, and researchers identify the principal climate stresses for current and future agricultural production in their locations (Vermeulen et al., 2013).

Step 5. Gap analysis of functional diversity in farm systems: Farmers and other actors can identify the need for diversifying their farm systems with new crop functional types, such as cereals with C4 photosynthesis (Lavorel and Garnier, 2002) or the need for new management practices, such as the establishment of shade trees to make farm systems more resilient against climate changes (Altieri et al., 2015).

Step 6. Selection of on-farm diversification options: Farmers choose crops on the basis of multiple criteria considering their goals, enabling and disabling factors, climate-related production risks, and gaps in functional diversity (Coe et al., 2014).

Step 7. Evaluation and learning: These activities are part of adaptive management. Farmers continuously evaluate and improve on-farm diversification strategies in dialogue with other farmers, practitioners, and researchers (Allen et al., 2011).

STEP 1. FARMERS' GOALS

Any initiative to work with farmers starts with understanding farmers' goals and identifying how diversification of their farm systems contributes to these goals. Often profit-maximizing approaches, such as Modern Portfolio Theory (MPT) are used to determine the optimal number and type of crops or land-use systems to manage production risks for a certain expected return on investment under climate change (Figge, 2004; Werners and Incerti, 2007; Werners et al., 2011). Farmers, especially smallholders, often perceive benefits from on-farm diversification in ways which profit-maximizing approaches do not necessarily capture. When diversifying their farm systems, farmers often define multiple goals, for example, they consider cereals for food security; pulses and vegetables for nutrition; cash crops for increasing income; off-seasons crops and forages for animal production to stabilize income; and finally intercropping and field scattering to reduce production risks (Schroth and Ruf, 2014). Different household members, such as women and men, may have different goals (van de Fliert and Braun, 2002; Chaudhury et al., 2013). Participatory approaches have proved effective in enabling practitioners and researchers to understand the goals of different members of farm households (Mazón et al., 2016; Dumont et al., 2017). Understanding farmers' goals is thus the basis of working with farmers in developing, selecting, evaluating, and implementing on-farm diversification strategies.

STEP 2. ENABLING FACTORS

Extension

A particular challenge is that our proposal to work in a participatory way with farmers comes at a time when public extension services have been severely eroded in much of the developing world (Umali-Deininger, 1997; Hellin, 2012). Private extension has increased but there has been a tendency to focus efforts on better-off farmers leaving those in marginal areas with limited services (Hellin, 2012). There are, however, growing examples of innovative extension approaches which include both the public and private sector (Chapman and Tripp, 2003). The transformation from specialized to diversified farm systems can be fostered by agricultural innovation systems (Schut et al., 2014). In the absence of extension and agricultural innovation systems, farmers would need to rely largely on neighboring farmers, farmer organizations, and local knowledge to adapt their farm systems to climate change.

Farmer Organization

The organization of farmers in associations, farmer-to-farmer movements, or other types of social organization can be an effective way to scale practices to diversify farm systems because these organizations are conduits for the dissemination of knowledge and information (Shiferaw et al., 2009; Mier et al., 2018), and allow to establish safety nets for farmers through formal and informal insurance programs (Tucker et al., 2010; Bacon et al., 2014) (**Table 1**, Examples 1 and 2). Capacity development on good governance and finance makes farmer organizations more competent, efficient, and transparent, and diminishes dependence on external authorities or donors. With these skills, farmer organizations can reduce the risks on "elite capture," secondly they can access credit from banks and social investors to invest in on-farm diversification, and finally they can connect to networks of agricultural innovation to access markets and external support (**Table 1**, Example 2). Farmer organizations are thus in principle good partners in selecting, developing, evaluating, and implementing on-farm diversification strategies.

Local Knowledge and Neglected and Underutilized Crops

At least 7,000 food plant species have been documented and these provide a rich basket of crop choices (Padulosi et al., 1999). Many

TABLE 1 | Examples of successful societal, public and private initiatives to support farmers in diversifying their farm systems.

Example 1: Over several decades, agroecological farmer-to-farmer networks in Central America, Mexico, and Cuba have reached ten-thousands of farmers (Mier et al., 2018). These networks introduced straightforward agroecological practices enhanced by local experimentation, farmer-to-farmer learning, and training and promotion of farmer extension workers (Holt-Giménez, 2002; Mier et al., 2018). These agroecological practices include the introduction of cover crops and green manures, such as velvet bean (*Mucuna pruriens*) and jack bean (*Canavalia ensiformis*), which reduce the sensitivity of farm soils and productivity to hurricane and flooding exposure (Holt-Giménez, 2001, 2002).

Example 2: Smallholder coffee farmers who are members of associations in Guatemala and Nicaragua have been able to access training on and inputs for agroecological practices, access formal and informal safety nets, and export coffee (*Coffea arabica*) at a premium price (Bacon et al., 2014; Morris et al., 2016; Winget et al., 2017). Among agroecological practices, shade tree species, such as cocoashade (*Gliricia sepium*) and salmwood (*Cordia alliodora*), are commonly used to stabilize above ground temperatures in Mesoamerican coffee systems (Lin, 2007).

Example 3: Associations of gastronomy and avant-garde chefs in Peru have promoted a cuisine with neglected and underutilized crops to a wider public, including native *Capsicum* peppers, native potatoes, and local fruit species (Hellin and Higman, 2005; Matta, 2013).

Example 4: The vegetable seed company East-West Seed successfully scaled and diversified the production of vegetables in Southeast Asia and other regions. East-West Seed produces seeds of 60 crops and 1,000 varieties to support diverse vegetable farm systems. As part of their seed sales, East-West Seed sold 25 million one-dollar seed packs, which are accessible to smallholders (East-West Seed, 2016). In 2019, East-West Seed received the World Food Prize in recognition of their impact in creating sustainable economic opportunities for small farmers around the world over the last four decades.

Example 5: In Kenya and Tanzania, national and international agricultural research organizations, local and international seed companies, governmental and farmer organizations collaborate in a network to promote variety and seed system development of traditional African vegetables, such as African eggplant (*Solanum aethiopicum*), leafy nightshade (*Solanum scabrum*), and spider plant (*Cleome gynandra*) (Dinssa et al., 2016; Stoilova et al., 2019). One of the most-promising traditional vegetables in East Africa is leafy amaranth (*Amaranthus* spp.), a hardy and nutritious C4 crop. In 2017, about 231,000 farmers in Kenya and Tanzania increased their yield by growing improved amaranth varieties. These varieties are developed, distributed, and commercialized through this network in response to increased urban demand for leafy amaranth in East Africa (Ochieng et al., 2019).

Example 6: An example of where index insurance can enhance on-farm diversification is in Ethiopia. The World Food Program (WFP), Oxfam, and partners have initiated the R4 Rural Resilience Initiative. The initiative includes insurance as part of a larger climate-change adjustment program, which includes tree-planting and soil and water conservation. The program uses the work-for-assets model, enabling farmers to accumulate individual and/or group savings, which provide a "risk reserve." The initiative added an insurance component. In return for their work, farmers get access to an insurance scheme (Greatrex et al., 2015).

Example 7: Participatory prioritization and capacity building enhanced farmer uptake of native tree species in Costa Rica, Colombia, and Mexico where cattle ranchers successfully have implemented climate-resilient silvopastoral systems with native tree species (Murgueitio et al., 2011; Bozzano et al., 2014).

Example 8: In Brazil, governmental organizations, NGOs, and agricultural research institutes have collaborated to advocate for policies to promote the consumption of native foods. This has led to the publication of a national ordinance which officially recognizes the nutritional value of more than 60 native food plants (Beltrame et al., 2016). This has led to the inclusion of these species in subnational and local programs of school feeding food procurement. Farmers who participate in these programs can diversify their farms with nutritious food plants because the mediated food-procurement market provides an incentive to do so (Wittman and Blesh, 2017).

Example 9: An example of these agricultural innovation systems are consortia of research institutes and seed companies, which provide farmers with affordable seeds of improved vegetable lines and as a conduit for feedback between seed suppliers and farmers (Schreinemachers et al., 2017b; Ochieng et al., 2019).

are neglected and underutilized (National Research Council, 1989; Clement, 1999). These species could become important for food security under changing climate conditions because they have evolved during a long history of human selection and fluctuating climate conditions (Mercer and Perales, 2010; Padulosi et al., 2011). Some examples of promising species for diversification and climate change adaptation are provided in **Table 2**.

Farmers in traditional communities have commonly diversified their farm systems with these crops to manage production risks related to unpredictable weather cycles (Winterhalder et al., 1999; Matsuda, 2013; Altieri et al., 2015). Much of the local knowledge associated with growing neglected and underutilized crops is at risk of extirpation due to changing diets, reduced interest by young people in agriculture, and shifts in production systems under climate change (Padulosi et al., 2011; Khoury et al., 2014). With this loss, farmers have fewer diversification options. This makes them more vulnerable to climate change. Finally, the decline of production and consumption of these neglected and underutilized crops leads to the disappearance of local varieties whose traits for adaptation to climate stresses are not only important to local farmers but also for research and breeding by the global agricultural research community (**Table 3**, Example 1).

The promotion of these neglected and underutilized crops is complex and requires actions at both the supply side to incite farmers to continue using these crops and demand side to persuade consumers to incorporate these crops in their diets. Here we name three approaches to provide incentives to farmers' use of neglected and underutilized crops to diversify farm systems. First, within each community, commonly a few farmers are knowledge hubs on the management of these neglected and underutilized crops (Altieri and Merrick, 1987; Sthapit et al., 2013). These persons are custodian or lighthouse farmers who merit recognition in society and who can be encouraged to share their knowledge with other farmers as well as with practitioners and researchers. Second, empowerment of women in agriculture increases the options for on-farm diversification because both men and women maintain exclusive and complementary knowledge about crops and farm management (Padulosi et al., 2011). Because female-headed farm systems are not necessary more diverse than male-headed ones (Saenz and Thompson, 2017), it is important to understand the complementary impacts of women and men's choices on the diversification of farm systems (Farnworth et al., 2016). Finally, the identification and development of niche markets and new uses of neglected and underutilized crops can stimulate their production and the maintenance of local knowledge (**Table 1**, Example 3). There are, hence, several strategies to maintain and use local knowledge

TABLE 2 | Crop functional types and crop examples to diversify in response to various climate stresses.

Climate stress	Crop functional type	Trait examples	Crop examples	References
Drought and water scarcity	Dryland hardwood trees	Deep root architecture, phenological drought escape, deciduous	Mesquite *(Prosopis* spp.), glassywood (*Astronium graveolens*)	Borchert, 1994; Holmgren et al., 2006; Nabhan, 2013
	Tropical dryland lightwood trees	Water storage, deep root architecture, phenological drought escape, deciduous	Hog plum (*Spondias* spp.), pochote (*Pachira fendleri*), baobab (*Adansonia digitata*)	Borchert, 1994
	C4 perennial forage grasses	C4 photosynthesis, deep root architecture	Guinea grass (*Panicum maximum*)	Cattivelli et al., 2008; Lopes et al., 2011
	Crassulacean Acid Metabolism (CAM) crops	CAM metabolism, deep root architecture, phenological drought escape, water storage	Nopal (*Opuntia ficus-indica)*, maguey and other agaves (*Agave* spp.), pitayas (*Echinocereus* spp., *Stenocereus* spp. *Hylocereus undatus*)	Yang et al., 2015
	C4 cereals	C4 metabolism, deep root architecture, phenological drought escape	Maize (*Zea mays*), sorghum (*Sorghum bicolor*), teff (*Eragrostis tef*)	Lopes et al., 2011; Cheng et al., 2017
	Legumes	Phenological drought escape, water use efficiency, deep root structure	Chick pea (*Cicer arietinum*), cowpea (*Vigna unguiculata*), mungbean (*V. radiata*), moth bean (*V. aconitifolia*)	Subbarao et al., 1995; Ehlers and Hall, 1997; Graham and Vance, 2003; Iseki et al., 2018; Yundaeng et al., 2019
	Tropical root crops	Stomatal control, shift in leaf size, recovery of photosynthesis	Cassava (*Manihot esculenta*)	Bondeau et al., 2007; El-Sharkawy, 2007
Flooding and waterlogging	Tropical floodplain trees and shrubs	Dormancy and periodic growth, xeromorphic leave traits, starch storage in roots	Camu-camu (*Mycriara dubia*), acupari (*Garcinia brasiliensis*)	Peters and Vásquez, 1987; Parolin, 2009
	Aquatic grasses (forage and grains)	Root aeration, elongation growth response	Rice (*Oryza* spp.) brachiaria grasses (*B. humidicola*), teff, sorghum	Sairam et al., 2008; Bailey-Serres et al., 2012; Cardoso et al., 2013
	Swamp palms	Dormancy and periodic growth, root aerenchyma	Aguaje palm (*Mauritia flexuosa*), chambirilla (*Astrocaryum jauari*)	Kahn, 1991; Schluter et al., 1993
Heat	Tropical leguminous trees	Changes in concentrations of regulatory proteins	Mesquite, cocoashade (*Gliricia sepium*)	Felker et al., 1983; Ortiz and Cardemil, 2001; Nabhan, 2013
	CAM crops	Not found	Pineapple (*Ananas comosus*)	Yamada et al., 1996; Yang et al., 2015
	C4 cereals	Not found	Maize	Wahid et al., 2007
	Tropical Legumes	Heat escape, stabilizing mechanisms of cell membrane integrity, improved pod set under hot conditions	Cowpea, moth bean, yard-long bean (*Vinga unguiculata* group *sesquipedalis*)	Ehlers and Hall, 1997; Wahid et al., 2007; Yundaeng et al., 2019
	Palms	Not found	Cocos (*Cocos nucifera*), date (*Phoenix dactylifera)*	Yamada et al., 1996; Nabhan, 2013
Frost	Temperate cereals	Hardening	Oats (*Avena sativa*)	Rizza et al., 2001; Yadav, 2010
	Temperate legumes	Hardening	Faba bean (*Vicia faba*)	Arbaoui and Link, 2008

This list is not exhaustive and just provide some crop examples per crop functional type.

on neglected and underutilized crops to promote diversified farm systems.

Getting the Right Variety

Farmers often struggle to find planting material of crops with high potential for on-farm diversification even though appropriate varieties are often available at agricultural institutions or maintained by neighboring farmers (Jarvis et al., 2011). Due to weak formal and informal seed systems, farmers are not always able to access germplasm of appropriate varieties and diversify their farm systems. Farmers can access more varied germplasm when they are better connected to public and private germplasm suppliers (Coomes et al., 2015; Stoilova et al., 2019) and when these suppliers strengthen their germplasm production capacity (Schreinemachers et al., 2017a). The desired type of seed system differs between crop groups and should be defined per crop and region (Louwaars and de Boef, 2012). For example, public-private networks of research institutes and local, national, and international seed companies have proven to be successful to scale the supply of affordable and high-quality vegetable seeds (Schreinemachers et al., 2017a) (**Table 1**, Examples 4 and 5). Aside from fostering farmers' access to commercial and public germplasm in formal seed systems, farmer communities across the world successfully establish networks to conserve, use, and exchange germplasm of local varieties and associated knowledge (Coomes et al., 2015; Vernooy

TABLE 3 | Examples of on-farm diversification constraints related to market dynamics.

Example 1: In the central highlands of Mexico, farmers traditionally intercrop maize (*Zea mays*) and common beans (*Phaseolus vulgaris*) with maguey (*Agave atrovirens*), a neglected crop, which is adapted to dry conditions because of its Crassulacean Acid Metabolism (CAM) photosynthetic apparatus. Production of *aguamiel* from maguey, a natural sweetener and raw material for production of a traditionally fermented beverage, can provide an additional source of income (Eakin, 2005). In recent years, the demand for *aguamiel* has decreased as consumer preferences have changed. Without a market, farmers have largely stopped growing maguey and increasingly they grow only maize and common beans. This puts them in a vulnerable position as both crops are more susceptible to drought, frost and hail damage compared with maguey.

Example 2: In 2012 and 2013, many Mesoamerican coffee smallholder families suffered from hunger because coffee rust wiped out their coffee crop (*Coffea arabica*). Coffee rust thrived because of the interplay of poor management as a result of low coffee prices and unfavorable temperatures (Avelino et al., 2015). Many coffee farmers received technical and monetary support because of their affiliation to cooperatives and fair-trade schemes. While these safety nets helped many farmers to compensate for income loss and to manage coffee rust, these safety nets were not sufficient to protect all farmers and farm laborers (Morris et al., 2016). In addition, to further sustain food security, farmer organizations in Nicaragua have established grain banks for Central American smallholder coffee producers who suffer seasonal hunger (Bacon et al., 2014). Food insecurity was highest in households of coffee laborers without alternative income sources and coffee smallholder families who had abandoned or reduced the areas dedicated to traditional food crops (Avelino et al., 2015). Farmers' safety nets can be strengthened when these are combined with technical and financial support to diversify farm systems with food crops for subsistence and income generation from local markets. Farm laborers are the most vulnerable because they lack land for food production and would need to diversify their income sources with other *off-farm* activities.

Example 3: Nutrition of some households In the western highlands of Guatemala has declined when farmers started to grow exclusively high-value vegetable crops for export markets (Webb et al., 2016). Some of these vegetable farmers stopped growing or consuming nutrient-rich crops from traditional diversified farm systems characterized by Milpa system of maize, common beans, and associated crops. High-value crops may require large investments in fertilizer and other inputs; financial pressures may encourage producers to invest in commercial production, abandon traditional agriculture, and consume low-quality processed food (Webb et al., 2016). More research is required to understand when and how the replacement of food by cash crops affects the nutrition status of farm household members.

et al., 2017). The promotion of promising crops to diversify farm systems requires an assessment of the existing formal and informal seed systems to strengthen, where necessary, germplasm quality and supply in collaboration with farmer organizations, NGOs, breeders, genebanks, and private and public suppliers of planting material.

Insurance

Risk aversion on the part of farmers, especially smallholders, is an obstacle to the adoption of new crops, varieties, and novel management practices (Lee, 2005). Weather shocks, such as drought, can trap farm households in poverty because the risk of the shocks limits farmers' willingness and capacity to invest in on-farm diversification strategies (Dick et al., 2011; Carter et al., 2016). For example, fire risk in drought-prone areas limits farmers to diversify farm systems with tree species (Jacobi et al., 2017). As a complement to on-farm diversification, agricultural insurance against yield loss mitigates the risks farmers face and encourages them to diversify their farm systems (Bobojonov et al., 2013).

One approach gaining much attention is index insurance. With index insurance, payouts are based on an index, such as the total seasonal rainfall or average crop yield for a larger area. This index reduces the costs of insuring individual farmers (Bell et al., 2013). Furthermore, the insurance is based on a reliable and independently verifiable index and can be reinsured, allowing insurance companies to transfer part of their risk to international markets (Binswanger-Mkhize, 2012). Index insurance can be bundled with climate-adapted germplasm or cropping systems to encourage farmers to invest in crop productivity (Bobojonov et al., 2013) (**Table 1**, Example 6).

Index insurance, however, is not a perfect predictor of an individual loss. The difference between the farmers' actual losses and the expected payout is known as basis risk; it may result in a farmer suffering a yield loss, but not receiving a payout, or in a payout without the farmer experiencing any loss (Dick et al., 2011; Miranda and Farrin, 2012).

Whole-farm insurances could be another promising insurance measure to provide farmers an incentive to diversify farm systems (Hart et al., 2006; Turvey, 2012). What whole-farm and index insurances have in common is that combining agricultural insurance with on-farm diversification benefits both farmers and insurance providers. Diversified farm systems can stabilize income and productivity and reduce the risks and corresponding premia of insurance. A recommendation is to develop policies and incentives for innovative insurance services, which support and promote on-farm diversification.

Markets

High-value crops, such as fruit and vegetable species, have been identified as promising crops to diversify farm systems and to increase farmers' net income (Joshi et al., 2004; Pingali, 2007; Birthal et al., 2015). Vegetable species are of special interest because in general they have short rotation cycles and can provide quick and year-round returns (Schreinemachers et al., 2018). Market access may, however, be limited to large-scale farmers as smallholders often lack capital to make investments to convert a semi- or fully-subsistence farm system into a commercial farm system (Pingali, 2007; Eakin et al., 2012). Many high-value crops, such as leafy vegetables, are perishable and this often requires additional investments in post-harvesting and transportation. Finally, smallholders can be particularly vulnerable to fluctuating market prices (Eakin, 2003; Carletto et al., 2010). Linking farmers, especially smallholders, to markets therefore requires support by governments, food processors, and distributors to strengthen post-harvesting facilities, distribution channels, stable production supply, and insurance.

Farmers tend to focus on one or a few crops to meet quality demands. However, a sole focus on one or two high-value cash crops in a farm system can be a risk for food security and livelihoods for individual farm households as well as for local economies (Immink and Alarcon, 1991; Chakrabarti and Kundu,

2009) (**Table 3**, Examples 2 and 3). Rather than focusing solely on one or two cash crops, farmers may therefore opt to manage several crops and varieties with different production and price risks, to meet food and nutrition security goals, and increase net income (**Table 4**, Example 1).

STEP 3. DISABLING FACTORS

Scale Effects

Scale effects leading to crop and farm specialization may be stronger drivers than those leading to on-farm diversification. Such specialization can occur in the case of commodities where there is a demand for large quantities and where sophisticated and product-specific technical packages drive monocultures. Such can be the case for oil palm (*Elaeis guineensis*), sugarcane (*Saccharum officinarum*), and soybean (*Glycine max*). Indeed, for several decades, research and development efforts in the agricultural sector of many countries support technologies, which reinforce scale effects and favor specialization (Griffon, 2006; Pingali, 2012). Agricultural subsidies in countries, such as Mexico, Bolivia, and Zambia support large-scale monocultures rather than diversified production systems (Eakin and Wehbe, 2009; Jacobi et al., 2017; Saenz and Thompson, 2017).

With more research investment and policy support, scalable and economically-feasible diversification practices can be developed. So far, scaling of species mixtures has been successful for pasture and cover crops because these mixtures increase productivity without extra management costs (Bybee-Finley et al., 2018) (**Table 4**, Example 2). The wide-scale introduction of high-quality seed of vegetable crops to smallholder farmers in Southeast Asia during the last decades is a successful example on how to scale diversification of farm systems with high-value crops (Schreinemachers et al., 2018) (**Table 1**, Example 4).

Labor Constraints

Any on-farm diversification option should save labor and/or increase and/or stabilize net income to make it an attractive option for climate change adaptation (Lee, 2005). Labor saving is urgent because climate change is predicted to reduce farming labor capacity in tropical regions by up to 50–80% in peak months of heat stress (Dunne et al., 2013; Myers et al., 2017). Diversification with cover crops and shade trees can reduce the labor costs of weed control (Raintree and Warner, 1986; Holt-Giménez, 2006; Liebman and Dyck, 2007) or fertilizer input in the case of cocoa agroforestry systems (Armengot et al., 2016). However, often diversified farm systems require more labor compared with less complex systems (Bacon et al., 2012). This has been the case for diversified rice systems and cocoa systems (Pingali, 1992; Armengot et al., 2016). The introduction of high-value crops, such as fruit and vegetable species could be an alternative diversification strategy to increase or stabilize net income (Joshi et al., 2004). Finally, diversification strategies, which improve on-farm climate conditions, such as the establishment of shade trees can eventually improve labor conditions because while they may require a large initial labor input this tails off substantially after tree establishment.

Farm Size and Land Ownership

Although farm size is thought to be a constraint for diversification, we did not find a clear correlation between farm size and on-farm diversification. As part of a systematic literature review, which included 13 detailed studies, six reported that on-farm diversification increases with farm size; four studies reported no effect; and three studies reported that on-farm diversification reduces with farm size (**Table S1**). There is thus scant evidence that farm size is an enabling factor or constraint for on-farm diversification. Our recommendations to diversify farms are therefore relevant for different farm sizes.

We found only a few studies, which consider land ownership as a factor in diversification (Lawin and Tamini, 2017; Asante et al., 2018). These studies showed no relationship between land ownership and diversified farms. More research is needed to understand better if there is any relation between these two variables.

STEP 4. CURRENT AND FUTURE CLIMATE-RELATED PRODUCTION RISKS

Farmer perceptions of weather cycles and climate change are a good starting point for identifying climate risks. Their knowledge may need to be combined with formal predictions to reduce bias from their recent experiences and to reflect long-term climate trends. Once climate risks are identified, crops, varieties, and management practices can be selected to manage these risks.

Climate models with projections in climate change under different economic and climatic scenarios allow for predictions of climate change impact on crop production for the next decades (Lobell et al., 2008; Baca et al., 2014; de Sousa et al., 2019). The main purpose of these models is to reduce uncertainty in decision-making rather than to give precise predictions (Vermeulen et al., 2013). These models are relevant for planting decisions for both annual and perennial commodities, such as soybean and coffee (*Coffea* spp.), for which a whole infrastructure needs to be maintained or put in place. Even in the case of the introduction of non-commodities, time may be required to develop seed systems and to develop the capacity of farmers who are interested in growing these crops.

Climate models, which use historic climate trends, help to predict trends in climate stress for shorter time spans compared with the decadal predictions of climate models on the basis of projections in climate change. To be effective, the results of these models have to be communicated clearly to farmers (Pulwarty and Sivakumar, 2014). The Famine Early Warning Systems Network (FEWS NET), for example, provides rainfall predictions for the next 10–365 days on the basis of high-resolution rainfall and hydrological models (Senay et al., 2015). These predictions allow farmers and other actors in the value chain to anticipate and adjust cropping systems to water scarcity or surplus. High-quality modeling in combination with good communication is thus essential to provide farmers meaningful information about current and future climate risks.

TABLE 4 | Successful examples of diversified cropping, pasture, and agroforestry systems.

Example 1: In the semi-arid regions of Myanmar, farmers manage a diversified cropping system with cash crops, such as cotton (*Gossypium* spp.) and sesame (*Sesamum indicum*), and food crops, such as rice (*Oryza* spp.), pigeon pea (*Cajanus cajan*), and mungbean (*Vigna radiata*) (Matsuda, 2013). This diversified farm system provides multiple income and subsistence sources under uncertain weather conditions.

Example 2: Species mixtures have a high potential to diversify pasture lands because the diversification of sowing material does not substantially increase labor costs for a farmer and will increase and stabilize productivity. Pot experiments show that diversified pasture lands with multiple genotypes and multiple species increase the stability and productivity for meat and milk production under climate variability (Prieto et al., 2015). Legumes have a high potential to augment the functional trait diversity of tropical pastures (Schultze-Kraft et al., 2018). A large range of legume crops is available for different tropical agroecological zones (Schultze-Kraft et al., 2018).

Example 3: The traditional Milpa system with maize (*Zea mays*), common beans (*Phaseolus vulgaris*), squash (*Cucurbita* spp.), and other crops is still an important cropping system in Mexico and Central America for the food security of many smallholder farmers (Isakson, 2009; Salazar-Barrientos et al., 2016). The Milpa system can be combined with growing export cash crops, such as coffee to get a diversified farm system, which meets multiple farmers' goals related to income and food security (Morris et al., 2016). The Milpa system combines different functional traits including C4 cereals and legumes. The system rotates maize and beans and can be adapted to different climate conditions using different types of varieties and different types of rotation systems (Trouche et al., 2006). Several crops can be intercropped with maize, such as cucurbits (Salazar-Barrientos et al., 2016). When climate conditions are too dry for maize, this crop can be replaced by sorghum (*Sorghum bicolor*) (Trouche et al., 2006).

Example 4: In the high-altitude regions of central Mexico, late season frost is a major threat to maize production. Changing climate has resulted in the late arrival of spring rains, a delay to the planting date and an increase in the risk of late season frost. Mexican farmers in these frost-prone areas minimize risk by diversifying their production area with more frost tolerant crops, such as oats (*Avena sativa*) and fava beans (*Vicia faba*) (Espitia Rangel et al., 2007; Maqbool et al., 2010). Maize is still the preferred crop and has a high market demand, so farmers tend to adjust the crop area based on the planting date; the later the planting date, the smaller the area planted with maize and the greater the area planted to a crop with higher frost tolerance (Eakin, 2005).

Example 5: In the dry corridor of Central America and Yucatan peninsula, fruit trees provide a safety net in the dry season. Indigenous communities traditionally relied on Maya nut (*Brosimum alicastrum*) and other food tree species to cope with failed harvests in dry years (Gómez-Pompa, 1987). These trees were removed from the landscape to make way for more intensive farming practices. Different seed sources of Maya nut have now been identified for replanting in home gardens for food security in times of drought and to have a reliable forage supply for cattle (Vohman and Monro, 2011).

Example 6: In East Africa, a drought-tolerant legume crop, desmodium (*Desmodium intortum*) has been tested successfully as an intercrop to repel stemborer moths from C4 maize-production systems in combination with the perennial C4 grass *Brachiaria* cv *mulato* which is planted in field borders to attract this pest (Midega et al., 2018).

STEP 5. GAP ANALYSIS OF FUNCTIONAL DIVERSITY IN FARM SYSTEMS

By filling functional gaps in farm systems, farmers can stabilize and even increase primary productivity of their farm systems under climate change. This occurs via two distinct but linked agroecological mechanisms. First, diversification with crops and varieties, each with a differential response to climate stresses, stabilizes primary productivity in agroecosystems under climate variability. The second mechanism is related to diversification of crops and management practices to foster ecological functions. Ecological functions increase and stabilize primary productivity in farm systems and include climate regulation, water storage, nutrient cycling, and pest regulation. By understanding these two agroecological mechanisms and translating that knowledge into practical recommendations for decision-making, farmers can make informed choices about adapting their farm systems to climate change.

Crop Choices for Differential Responses to Climate Stresses

Spatial diversification stabilizes primary productivity of farm systems under climate variability when crops with differential responses to climate stresses are grown in polycultures or in separate fields. These crops expand together the physiological range to produce a minimum yield under different climate conditions. In addition to physiological range expansion, positive plant interactions and niche complementary further increase and stabilize agricultural productivity (Brookfield, 2001; Malézieux et al., 2009).

When considering polycultures to diversify farm systems in a specific area, local knowledge on crops and management practices provide a rich source of possibilities for rotations, intercropping, and agroforestry systems (Eakin, 2005; Hellin and Dixon, 2008; Isakson, 2009) (**Table 4**, Example 3). Traditional polyculture systems can fall into disuse because of labor constraints, poor markets, and erosion of local knowledge. It is therefore important to address these economic and cultural constraints in order to maintain and improve traditional systems, and introduce new systems as well.

Crop functional types help to differentiate between crops, which, because of their physiological differences, tolerate different types and different levels of climate stress (**Table 2**). For polycultures, farmers ideally choose crops, which besides their differentiated tolerance to climate stresses, have complementary traits to reduce competition for similar resources, such as different rooting depths, complementary nutrient requirements, and differential light interception patterns (Brooker et al., 2015). In this way, farmers can minimize competition for light, water, and nutrients between crops, and avoid production and income loss.

The upper temperature ranges for the production of many crops is below 40°C while temperature conditions above 40°C become more prevalent in tropical growing areas (Farooq et al., 2017). Only a limited amount of crops can adapt to temperatures above 40°C, either through short growth seasons or by coping with high temperatures during sensitive development stages, such as pollen development, fruit setting, and grain filling (Wahid et al., 2007; Barnabás et al., 2008). **Table 2** gives a few examples of the crops which are reported to be strong candidates for agricultural production under hot conditions. In contrast, low temperatures can cause production risks in mountain areas in tropical and subtropical regions (**Table 4**, Example 4).

Plant production is principally limited by lack or excess of water. Drought and flooding events have occurred with greater frequency over the past 50 years and the trend is predicted to continue (Lobell et al., 2008). Despite the vulnerability of many plant species to drier conditions (McCord et al., 2015), a wide range of species is adapted to dry conditions in rain-fed systems. **Table 2** includes a few examples of species, which are reported to be strong candidates for on-farm diversification of rain-fed systems under increasing drought conditions.

C4-metabolism crops, such as maize (*Zea mays*) and sorghum (*Sorghum* spp.) have in general a high water-use efficiency and are better in tolerating water stress compared with C3-metabolism crops, such as wheat (*Tritricum* spp.) and sunflower (*Helianthus annuus*) because of their more efficient photosynthetic apparatus (Zhang and Kirkham, 1995; Nayyar and Gupta, 2006). This makes C4 crops potential candidates for production under dry and hot conditions, although several C4 crops may be susceptible to water stress because of the wide diversity in C4 plant evolution (Ghannoum, 2009). Crassulacean Acid Metabolism (CAM) crops use significantly less water and can grow in higher temperatures compared with C3 and C4 crops. Some CAM crops, such as pineapple (*Ananas comosus*) are commercial crops. The majority of CAM crops, however, are neglected or underutilized (Mizrahi et al., 2007; Yang et al., 2015).

Tree planting is a common on-farm diversification strategy to improve microclimates after their establishment (Bryan et al., 2009; Meldrum et al., 2018). Native tree species may be preferred candidates for diversification (**Table 1**, Example 7; **Table 2**). Since most tree species are wild or at an incipient stages of domestication, some exotic tree species can become invasive, such as the American species *Prosopis juliflora* in African countries (Richardson, 1998), or can be highly competitive for water, such as *Eucalyptus* spp. and may outcompete understory crops under drought-stress conditions (Saxena, 1991; German et al., 2006). Native food tree species provide also a reliable food source for farmer households in lean months (Graefe et al., 2012; Bacon et al., 2014) (**Table 4**, Example 5). Despite their potential importance for food and nutrition security, there is generally a lack of focus on these tree species in people's diets under climate seasonality and inter-annual variability (Rowland et al., 2015).

As periods of drought become longer and more frequent, farmers may need to replace water-competitive shade trees with species, which are less water demanding. The pruning of tree species reduces water stress and allows farmers to manage shade (Bayala et al., 2002) while also providing mulch to conserve soils and retain soil moisture (Hellin et al., 1999).

With respect to water excess, food tree species from tropical floodplains and swamps, such as many palm species, tolerate long periods of waterlogging (**Table 2**). In a similar line, sugarcane and perennial forage grasses, such as *Brachiaria* spp., can withstand waterlogging conditions (Cardoso et al., 2013; Gomathi et al., 2014). As with tree species, native forage grasses may be preferred because of the risk that exotic ones become invasive (DiTomaso, 2000).

Many traits related to stress tolerance can be found at variety level. Major advances have been made in breeding to increase drought tolerance of main cereal crops, such as maize (Cairns et al., 2013). Nevertheless, farmers may still want to diversify with drought-tolerant minor cereals and legumes (**Table 2**). For some cereals, such as maize, landraces could be good choices in strategies of on-farm diversification because they contain high levels of genetic variation, which enable landraces to evolve under the interplay of human selection and climate change (Mercer and Perales, 2010; Vigouroux et al., 2011). Evaluation of these landraces in different environments helps shed light on their potential for climate change adaptation and in breeding strategies in a similar way to the search for climate-adapted durum wheat landraces (Ceccarelli, 2015; Mengistu et al., 2016).

Even though breeders use advanced technologies, such as genomic selection and editing to develop varieties with multiple traits to tolerate climate stresses (Tester and Langridge, 2010; Mousavi-Derazmahalleh et al., 2019), it remains a challenge to stack these traits in single varieties (Mercer and Perales, 2010). Alternatively, a traditional approach is to grow multiple varieties of the same crop to respond to multiple stresses (Jarvis et al., 2008; Matsuda, 2013; Salazar-Barrientos et al., 2016). Farmers can thus diversify their farm systems by growing both multiple crops and varietal mixtures. In the same line, livestock and feed producers may prefer pasturelands, which are both rich in grass species and rich in genotypes because these pasturelands are more productive and recover better after extreme events, compared with less diverse ones in the same biotope (MacDougall et al., 2013; Prieto et al., 2015).

Crop Choices and Management Practices to Foster Ecological Functions

Diversification of farm systems in space and time can foster ecological functions, such as climate regulation, water storage, nutrient cycling, and pest regulation. Farmers may find it useful to use a straightforward checklist of management practices, which foster ecological functions to improve their farm systems (**Table 5**).

Microclimates can be regulated by tree shade, which buffers against high temperatures above ground and in some cases prevent frost damage (Barradas and Fanjul, 1986; Caramori et al., 1996) (**Table 1**, Example 2). Forage tree and shrub species, which are planted along field borders, provide a wind-break to maintain moisture levels in agriculture fields (Holt-Giménez, 2002), and are a source of animal fodder in times of drought (Kort, 1988; Tamang et al., 2010). Tree species can therefore be selected for multiple goals in farm systems including for food or fodder production and to maintain ecological functions.

On-farm diversification with cover crops and green manures can improve and conserve soil by building up organic matter, adding nitrogen, improving soil structure, and reducing soil erosion (Cong et al., 2014). As a consequence, soil fertility, infiltration, water holding capacity, and soil moisture can increase, and with that the crops' ability to cope with drought (Erenstein, 2003; Waraich et al., 2011). However, under humid conditions and on poorly drained soils, mulching can cause waterlogging resulting in lower yields (Giller et al., 2009). Some cover crops are competitive for water, and if intercropped, they can reduce the yields of the main crop under water limiting conditions. Therefore, selection of soil-improving intercrops or relay crops, which are water efficient, is important in

TABLE 5 | Diversification strategies to maintain or include ecological functions in farm systems.

Ecological function	Climate related stress	Mechanism	Functional types	Diversification strategy
Microclimate regulation and shade provision	Excess heat	Block solar radiation, cooling	Shade producing plants, trees and shrubs	Plant trees to increase canopy density
Disturbance regulation	Strong winds, typhoon	Physical wind break	Trees and shrubs, coastal mangroves	Place of hedgerows and wind breaks
Water regulation	Excess water, extreme rain events	Improved soil structure and drainage	Deep rooting plants, trees and shrubs	
Soil retention	Extreme wind and rain events	Physical soil stabilization, protection of soil surface	Shrubs, trees, grasses, and cover crops	Used as living barriers in sloping land and soil cover in annual systems
Soil formation and nutrient cycling	Drought, cold-associated hydric stress	Improved soil structure and nutrient retention	Biomass-producing crops, leguminous plants	Residue retention and reduced tillage, intercropping, relay cropping, pruning leguminous trees
Biological regulation	Shifts in pest and disease ranges and pressures	Habitat diversification, predator habitat provision, trap crops, microclimate management	Crop/pest specific	Intercropping, planting in field borders

drought-prone environments. Alternative management options in semi-arid regions include external biomass input from hedgerows or woodlots and establishment of rotation schemes with cover crops.

Crop residue incorporation is an important practice to improve soil quality (Turmel et al., 2015). In mixed cropping and livestock systems, especially in semi-arid areas, trade-offs exist between using residues for fodder or soil cover (Giller et al., 2009). In many areas, however, farmers require these residues for animal feed and in some cases they earn more from selling the residues for feed than they can from the maize they grow (Beuchelt et al., 2015). If farmers leave at least a portion of their residues in their fields, then they provide soil cover and build organic matter (Turmel et al., 2015). Alternative biomass-producing crops and sources of forages and soil cover can be introduced in intercropping, agroforestry, or silvopastoral systems to address these needs.

Holt-Giménez (2002) showed how diversification of Nicaraguan farm systems with agroecological practices, such as soil cover, windbreaks, crop rotation, and alley cropping, protected farmers' fields during extreme weather events compared with farmers' conventional practices (**Table 1**, Example 1). This evidence suggests that diversification enables farm systems to recover more quickly from extreme weather events compared with uniform farm systems.

Diversification across multiple spatial scales beyond the farm level is thought to further stabilize micro and mesoclimates and make farm systems more resilient against extreme weather events (Kremen et al., 2012). Forest patches surrounding cropping systems and pasturelands may control rainfall distributions and regulate temperatures at meso-level, but more evidence is needed (Teuling et al., 2010; Lawrence and Vandecar, 2015). Preliminary evidence show that farm systems in a diversified landscape indeed recover more quickly from extreme weather events compared with farm systems in uniform landscapes but the finding are not yet conclusive (Philpott et al., 2008; Gil et al., 2017). Monitoring farm systems in areas with extreme weather events will help to collect more data to understand further how diversification at multiple spatial scales makes farms more resilient against extreme weather events.

Caution is needed when introducing a new crop into a farm system since it can be a host of new crop diseases (Marshall, 1977; Anderson et al., 2004). Often, however, it is only a question of time until a pest or disease arrives because of globalized food export and import, and shifting distributions of pest and diseases due to climate change (Shaw and Osborne, 2011; Bebber et al., 2013). On-farm diversification is therefore a good preparation for when these pests or diseases arrive. First, crop diversification may reduce the risk of pest and disease outbreaks related to monoculture host plants (Rosenzweig et al., 2001). Some pests and pathogens, however, use a wide range of host plants, which limits the potential of crop diversification for preventing these outbreaks (Ratnadass et al., 2011). Second, heterogeneity in vegetation and crops obstruct pest movement and provide habitats for natural pest enemies (Avelino et al., 2012). Finally, a wide range of plant species, which repel or attract pests, is available to farmers. By understanding which climate stresses these plant species tolerate, they can be selected for pest control under changing climate conditions (**Table 4**, Example 6).

STEP 6. SELECTION OF ON-FARM DIVERSIFICATION OPTIONS

To support on-farm diversification, all the relevant information mentioned in steps 1 to 5 can be combined in a decision model, which captures multiple criteria (**Figure 2**). For many crops no exact information about markets and optimal growing conditions exist. Alternatively, ranking and scaling by a group of persons already provides robust estimates and comparisons (Hubbard, 2014; van Etten et al., 2016). These straightforward scoring approaches help determine which crops, varieties, and management practices are more appropriate for farmers' goals, such as income stability, food security, and/or nutrition; which crops and varieties require more or less labor, and so on. Selected diversification options can be further evaluated on-farm to test how well they fit farmers' realities, goals, and aspirations.

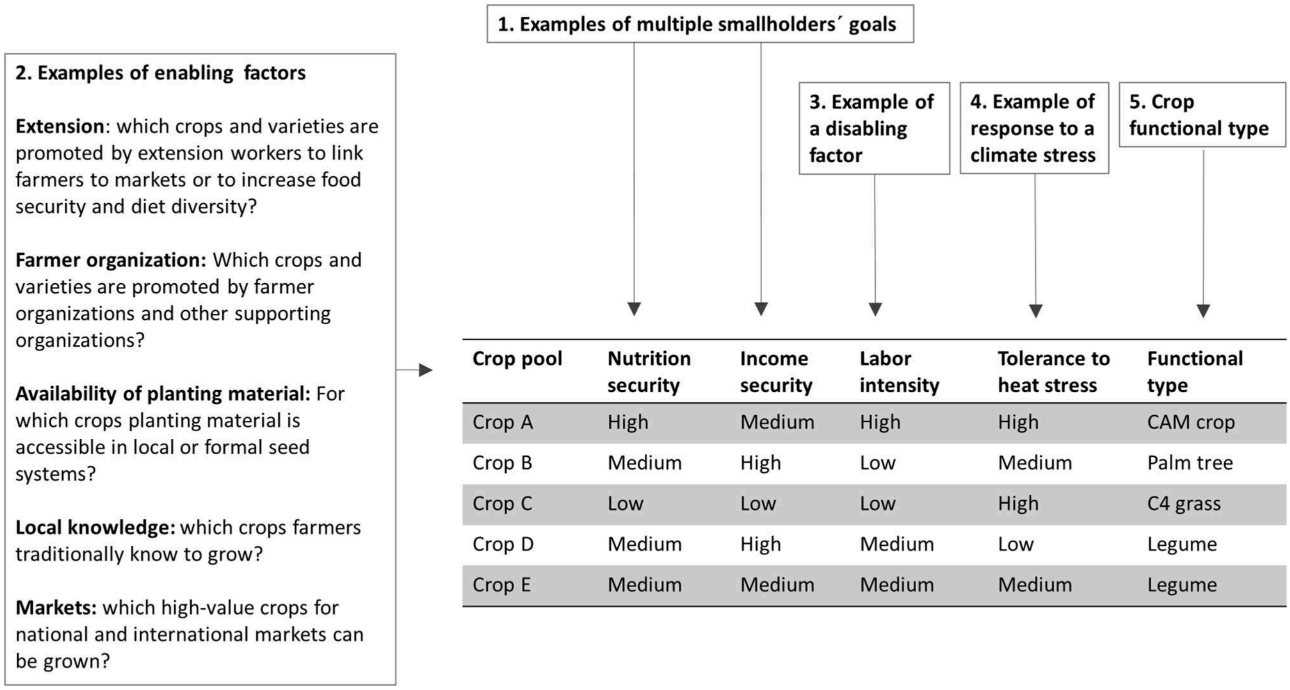

FIGURE 2 | Decision model to select crops and management practices for on-farm diversification. The existing enabling factors as defined in step 2 determine the availability of crop choices. Crops can be chosen using multiple criteria in function of the farmers' goals defined in step 1; disabling factors defined in step 3; climate stresses defined in step 4; and a gap analysis of functional diversity in step 5.

The selection of these options can be done in focus-group discussions in farmer communities with farmers, practitioners, and researchers, and by interviewing key persons from farmer communities, as well external actors, which could support farmers in access to markets, germplasm, climate information, credit, or insurance (Schattman et al., 2015; Morris et al., 2016).

Crop options are available for different agroecological zones. In all these zones, legumes and trees are common functional types to diversify farm systems for climate change adaptation (**Tables 2**, **4**). Some studies suggest that a low optimum number of on-farm diversification options for semi-arid agroecological zones (Waha et al., 2018). Therefore, it would be important to maximize the functional diversity in semi-arid regions within a few crops (see **Table 2**).

Many crops, which are hardy and can tolerate climate stresses, are neglected and underutilized (**Table 2**). The reality is that most of these crops have limited market opportunities. A selection of the crops with most potential for both climate change adaptation and markets, and targeted and long-term efforts to strengthen both supply of and demand for these selected crops, can help to support farmers to diversify their farms with these crops (**Table 1**, Examples 3 and 8).

Among high-value crops, vegetable species are commercially interesting for smallholder farmers and easy to incorporate in farm systems. However, we found little research on climate stress tolerance in vegetable species compared with species from other crop groups. Further research is needed to evaluate the response of vegetable species to different climate stresses because these are potentially interesting crops for diversification.

STEP 7. EVALUATION AND LEARNING

Participatory evaluation is a cost-effective way to evaluate crops, varieties, and management practices despite high transaction costs in communication and information exchange (Almekinders et al., 2007; Thomas et al., 2007). For on-farm testing of new crops, varieties, and management practices, home gardens are convenient because farmers traditionally use these places for experimentation (Williams, 2004; Galluzzi et al., 2010). After evaluation, farmers can decide if they wish adopt these new options and how best to incorporate them in their farm systems.

For uptake and scaling of diversification measures within communities, it is often advantageous to work initially with the most innovative female and male farmers, such as custodian or lighthouse farmers. They are often the most eager to experiment with diversification options and can subsequently inspire others (Hellin and Dixon, 2008). Researchers and practitioners can foster knowledge exchange between farmers by supporting farmer networks. Women and other vulnerable groups in many countries, would need to be involved in these activities to prevent increase in inequality as a consequence of differential access to information and learning opportunities (Tompkins and Adger, 2004).

Agricultural innovation systems are another form to share knowledge and to encourage learning about on-farm diversification options among farmers, and other private, public, and societal actors in value chains (Schut et al., 2014) (**Table 1**, Example 9). Feedback and information exchange on crop and variety performance between germplasm suppliers, farmers, and other actors improves site-specific crop and variety recommendations and enhances farmers' access to high-quality germplasm (van Etten et al., 2019).

DISCUSSION

In this paper, we propose seven steps to work with farmers in making choices about the development, selection, evaluation, and implementation of on-farm diversification strategies for climate change adaptation. These steps are based on existing concepts on climate change adaptation, which are often recommended separately. Complementary to existing tools, which recommend agroecological practices (Altieri et al., 2015), select species (de Sousa et al., 2019), or economically optimize crop portfolios (Werners et al., 2011), this decision-making framework brings together agroecological, agrobotanical, social, and economic considerations and recommendations from different disciplines, and links these to farmers' goals and constraints. The framework, coupled with extensive field experience from Latin America, Sub-Saharan Africa, and Asia, offers a practical and comprehensive tool for researchers and practitioners to establish a dialogue with farm households or with farmer groups to develop on-farm diversification strategies.

We argue that the four most essential elements for selection of appropriate on-farm diversification options are: step 1 on understanding farmers' goals, which is the basis of any adaptation plan; step 2 on identifying enabling factors to identify opportunities to support farmers with financial and technical support; step 5 on assessing gaps in functional diversity in farm systems, which need to be filled to adapt farm systems to climate change; and step 6 on the selection of on-farm diversification options to fill these gaps. These four steps would be the minimum needed to work with farmers in the development and selection of viable on-farm diversification options for climate change adaptation.

Practitioners, policy-makers, and farmer organizations who aim to incite farmers to diversify their farm systems in a specific territory, can use the framework as a check box and follow the steps in this framework on the basis of their existing knowledge and with support of local and international research organizations and networks. For example, the CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS) provides a toolbox to select climate-smart options (https://csa.guide/). Agroecological networks, such as the Community Agroecological Network (CAN), have established guidelines to carry out participatory action research (Méndez et al., 2017).

The framework counts in the lessons learned from successful cases on scaling agroecological practices (Mier et al., 2018). These cases stress the importance to foster farmer organization and external support as two key enabling factors, and to select effective and straightforward agroecological practices. One of the most compelling examples of scaling agroecological practices are agroecological farmer-to-farmer networks in Central America, Mexico, and Cuba (**Table 1**, Example 1). These networks show the importance of horizontal learning from farmer-to-farmer and through the establishment of dialogues between farmers and other actors (Holt-Giménez, 2006; Morris et al., 2016). Therefore, in addition to the four essential steps mentioned above, step 7 in our decision-framework on evaluation and learning is another important step in the diversification of farm systems.

The framework identifies insurance policies and market access as two additional enabling factors for on-farm diversification, in recognition of farmers' needs for enabling institutional environments to incentivize on-farm changes in crop and land management. Network structures for agricultural innovation for sustainable agriculture link farmer organizations to markets and insurance providers (Schut et al., 2014). We are not aware of successful policies to link insurance products to on-farm diversification, and we recommend policy-makers and practitioners to pilot these combinations.

The framework stresses the importance of understanding the goals of different farm household members and their diverse livelihood options and preferences. This provides the basis on which to establish a dialogue on diversifying farm systems, and allows to consider gender in the selection of diversification strategies. We stress this, because this may not always happen, resulting in a focus on profit-maximization in projects biased to narrow economic objectives or to poor linkage between recommended agroecological practices and the objectives of the different members of farm households.

To ensure that recommended practices align with farmers' economic objectives, we recommend practitioners and researchers to work with farmers in estimating the production costs and economic benefits of their existing farm systems in comparison with more diversified systems. Farmers are likely to determine the optimum extent of on-farm diversification by the balance between the labor input and other management costs associated with diversifying their farm systems, and the benefits from increased and more stable productivity leading to enhanced income and food security as a result of on-farm diversification.

Since labor constraints increase with climate change, it will be important to consider these increased labor costs in cost-benefit analysis and the implementation of diversification strategies. Recommended practices to diversify farm systems under climate change should therefore minimize extra labor, more so because of growing labor-scarcity due to rural-urban migration (Bacud et al., 2019). This fits well to the existing lesson in scaling agroecology to promote effective and straightforward agroecological practices (Holt-Giménez, 2001). When these practices minimize extra labor, then this will help to the successful implementation of diversification measures.

On-farm diversification strategies contribute effectively to CSA and SDG policies, which many governments aim to promote to enhance food security, climate change adaptation, and sustainable development (Lipper et al., 2014; Totin et al., 2018; Willett et al., 2019). On-farm diversification contributes less to climate change mitigation, which is another important component of CSA and SDG 13 on *Climate Action*. Although several on-farm diversification strategies, such as agroforestry or growing cover crops already address mitigation by sequestering carbon, this is not their primary goal when adapting farm systems to the adverse effects of global climate change. On-farm diversification in integrated CSA strategies should therefore be evaluated for their mitigation potential and when necessary combined with other mitigation strategies.

In some cases, on-farm diversification will not be sufficient to reduce the vulnerability of farmers to climate change (Harvey et al., 2014); on-farm diversification options simply do not save sufficient labor or sufficiently increase or stabilize net income. In these cases, *off-farm* diversification, such as seasonal labor in the non-agricultural sectors or a permanent exit from agriculture, may be a better option for farmers to adapt to climate change (Hansen et al., 2019).

CONCLUSIONS

On-farm diversification is a key component of a range of climate change adaptation and mitigations practices and technologies known collectively as CSA. Poorer farmers are particularly vulnerable to climate change and it is, hence, even more imperative that diversification options address the resources available to them and their aspirations. Increasing resources are being directed at CSA and we suggest following the seven steps presented in this paper as an approach to working with farmers for appropriate on-farm diversification as part of climate change adaptation and mitigation efforts. The seven steps provide a framework to identify appropriate diversification options in the context of farmers' agroecological and socio-economic conditions: (step 1) defining farmers' goals; (step 2) assessment of enabling factors; (step 3) assessment of disabling factors; (step 4) assessment of current and future climate-related production risks; (step 5) gap analysis of functional diversity; (step 6) selection of on-farm diversification options; and finally (step 7) evaluation and learning.

Governments often have few economic resources to put in force an agenda for CSA and, hence, network structures for agricultural innovation are vital for sustainable agriculture under climate change. Scale effects often favor monocultures. There are, however, several examples how food and feed demand in combination with adequate germplasm supply enables large numbers of farmers to diverse their farm systems and access markets. A successful example is diversified horticultural systems with high-value fruit and vegetable species for urban markets. Networks of agricultural innovation enable farmers to adopt diversification options by connecting with local, national, and international private companies, farmer organizations, public and private extension services, NGOs, as well as research institutes.

The key is to work with farmers in a participatory way and to prioritize their constraints, aspirations, and opportunities for on-farm diversification. A failure to do so, risks stymieing CSA efforts and ultimately perpetuating the vulnerability of those farmers who are often the target group of CSA. This would also result in CSA falling well short of its potential to contribute meaningfully to several of the SDGs including 13: *Climate Action;* SDG 1: *No Poverty*; SDG 2: *Zero Hunger*; and SDG 15 *Life on Land*.

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All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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SUPPLEMENTARY MATERIAL

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GLOSSARY

Agricultural innovation system or network: A network of actors including researchers, input suppliers, extension agents, farmers, traders, processors, and other actors who are engaged in the creation and use of knowledge relevant to agricultural production and marketing (Spielman et al., 2008).

Agroecosystem: A site or integrated region of agricultural production understood as an ecosystem with organisms, such as crop plant individuals, populations of crops, communities of polycultures, and ecosystems as farms or watersheds (Gliessman, 2014).

On-farm diversification refers to the incorporation of species, plant varieties or breeds, and management practices and land-use systems in farm systems in space and time through a range of spatial practices, such as polycultures, agroforestry systems, field scattering, and hedgerows; and temporal diversification through crop rotations (Somarriba, 1992; Vandermeer, 1992; Goland, 1993; Brookfield, 2001; Liebman and Dyck, 2007; Kremen et al., 2012).

Crop functional type: Practical ecological approach to group crops with similar traits and responses to changes in environmental factors (Lavorel and Garnier, 2002; Bondeau et al., 2007; Gilbert and Holbrook, 2011).

Farm system: A decision-making unit comprising the farm household, cropping, agroforestry, and/or livestock systems, which transforms land, capital, and labor into useful products, which can be consumed or sold (adjusted from Fresco and Westphal, 1988).

Germplasm: Living tissue from which new plants can be grown, such as seeds, meristem, or pollen.

Index insurance: Payouts are based on an index (such as the total seasonal rainfall or average crop yield for a larger area) and this reduces the costs of insuring farmers (Bell et al., 2013).

Local knowledge: A collection of certainties and experiences, which relate to a system of concepts, beliefs, and perceptions, which people hold about their environment. This includes the way people observe and measure their surroundings, how they solve problems and validate new information. It includes the processes whereby knowledge is generated, stored, applied, and transmitted to others (Warburton and Martin, 1999).

Modern Portfolio Theory: Optimization technique to determine optimal number and type of crops or land-use systems to manage production risks for specific expected returns on investment under climate change. In MPT, risks are defined as the variance in returns to expected production or gross margin across years.

Polyculture: Multiple cropping systems, such as intercropping systems and multistrata systems.

Resilience: The amount of change a system can undergo and still remain within the same domain of attraction (Gallopín, 2006). This is related to the extent that farmers can adapt their farming systems to climate change (Eakin et al., 2012).

Smallholders: Farmers who own small-based plots of land on which they grow subsistence crops and one or two cash crops and generally rely principally on family labor. Smallholders generally have <2 ha of land in production but farm-size is context-specific. In the western highlands of Guatemala many farm households have access to land well below 2 ha (Hellin et al., 2017) while in parts of Brazil a smallholder farmer may own up to 50 ha. Smallholders often have limited marketing, storage, and processing capacity. The average annual income for commercial smallholder production in generally below 5,000 USD/year (Lowder et al., 2016).

Neglected and underutilized crops: Neglected crops may be globally distributed, but tend to occupy special niches in the local ecology and in production and consumption systems. While these crops continue to be maintained by socio-cultural preferences and use practices, they remain inadequately characterized and neglected by research and conservation. Many underutilized crops were once more widely grown but have fallen into disuse for a variety of agronomic, genetic, economic and cultural factors. Farmers and consumers are using these crops less because these crops are in some way not competitive with other crops in the same agricultural environment (Padulosi et al., 2002). These crops include food and forage tree species and any other agricultural plant species; they are also known as minor, orphan, underexploited, underdeveloped, lost, new, novel, promising, alternative, local, traditional, or niche crops.

Whole farm insurance: A single insurance, which covers the covariate risk of jointly produced farm crop and livestock enterprises (Turvey, 2012).

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Ecological and Nutritional Functions of Agroecosystems as Indicators of Smallholder Resilience

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Smallholder agriculture is the foundation of global food systems, yet smallholders face severe socio-economic and environmental challenges that can destabilize livelihoods and threaten their resilience. Given that smallholder farmers rely on household production to meet their nutritional needs, management of soil fertility, biodiversity, and other ecological characteristics of agroecosystems directly affects smallholders' capacity to produce sufficient crop nutrients for their diets. However, we lack explicit frameworks linking ecological and nutritional functions of agroecosystems, as well as research exploring farmers' adaptive capacity and agency in mediating these functions, and ultimately, agroecosystem resilience. To address these gaps, we developed an indicator framework to evaluate the complementary roles of ecological and nutritional functions of agroecosystems for smallholder resilience. Paired ecological and nutritional indicators were aggregated into an index representing four agroecosystem functions: (1) Productivity, (2) Diversity, (3) Quality, and (4) Functional Diversity. We then applied this framework and index to a case study of Q'eqchi' Maya smallholders in eastern Guatemala, using farm management and crop quality data from 60 households to determine the status of agroecosystem functions and assess coping and adaptive capacities in response to shocks. More than three-quarters of farms in the sample relied solely on household production of staples to meet their nutritional demands. Across farms, ecological and nutritional indicators were significantly related (Kendall's tau $= 0.58$, $z = 5.7$, $p < 0.0001$), and we found both synergistic (Quality, Functional Diversity) and tradeoff (Productivity) relationships between indicator pairs. We found that farmers using ecological adaptation strategies such as cover cropping and agroforestry had significantly higher levels of agroecosystem functioning and resilience than farmers who were coping with shocks by working off-farm or renting land from plantations. Our findings demonstrate the importance of linking ecological and nutritional functions of agroecosystems through diversified management practices to leverage their synergies. Because smallholder agroecosystems underlie a third of the food system, understanding and promoting their resilience is critical for the social, ecological, and nutritional well-being of global populations.

Keywords: adaptive capacity, agroecology, functional diversity, synergies and tradeoffs, crop nutritional quality, indicator framework, food security and nutrition, land grab

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INTRODUCTION

Smallholder agriculture persists as the foundation of global food systems. Farms smaller than 2 hectares produce more than 30% of the world's food and occupy 24% of agricultural land (Herrero et al., 2017; Ricciardi et al., 2018), yet smallholders face severe socioeconomic and environmental challenges that can destabilize livelihoods and threaten their resilience (Scherr, 2000; Blesh and Wittman, 2015; Cohn et al., 2017). In degraded landscapes, where poor soil fertility can jeopardize crop yields, many smallholders are forced to compromise long-term sustainability to meet short-term production needs (Rodriguez et al., 2006; Power, 2010). While food production in the short-term is essential to maintain household nutrition, neglect for long-term ecological sustainability can make land unviable for future production. Thus, there is an interplay between ecological and nutritional functions of agroecosystems that may influence whether smallholders are able to adapt and continue farming within a deteriorating environmental context. Farmer management decisions can either increase or decrease these functions, demonstrating their capacity to adapt, and make incremental adjustments or changes that ultimately affect their resilience. Because smallholder agroecosystems support a third of the food system, understanding and promoting their adaptive capacity and resilience is critical for the social, ecological, and nutritional well-being of rural communities, and, ultimately, the global population.

Several recent studies operationalize agroecosystem resilience by developing indicators and metrics (Büchs, 2003; Cabell and Oelofse, 2012; Urruty et al., 2016; Jacobi et al., 2018). However, such frameworks neglect the ecological and nutritional interactions that contribute to agroecosystem functioning and resilience. Management of soil fertility, biodiversity, and other ecological characteristics on farms can directly affect the capacity of agroecosystems to produce sufficient crop nutrients for smallholder households to meet their nutritional demands (DeClerck et al., 2011; Allen et al., 2014). Smallholders depend largely on their own food production and local market availability for food security and nutrition (Jones, 2017), which tightens relationships between ecological and nutritional functions of local food production. To our knowledge, no prior study has developed an explicit framework to link ecological and nutritional functions of agroecosystems, or explored how, by employing strategies that enhance both ecological and nutritional functions, farmers' adaptive capacity and agency can mediate these agroecosystem functions, and resilience.

Our study developed an indicator framework to evaluate the complementary roles of ecological and nutritional functions of agroecosystems for smallholder resilience (**Figure 1**). We generated pairs of ecological and nutritional indicators of agroecosystem functions based on prior theoretical and empirical work and applied our framework to a case study of Q'eqchi' Maya smallholders in eastern Guatemala. The case study showcases the linkages between smallholder management decisions, adaptive capacity, and underlying ecological and nutritional functions that shape environmental and human health outcomes. While tradeoffs exist between practices that optimize either ecological or nutritional functions of agroecosystems, management strategies can also lead to synergies between them (Power, 2010), demonstrating the potential for smallholders to adapt and enhance resilience in a changing environment.

BACKGROUND

Social-Ecological Resilience of Agroecosystems

Social-ecological resilience offers a useful analytical framework to evaluate the long-term productive capacity of smallholder farms vulnerable to environmental and economic shocks (Folke, 2006; Cabell and Oelofse, 2012). Researchers frequently define social-ecological resilience as a system's capacity to respond to disturbance and shocks and retain its essential components and functions, as well as its capacity for learning and adaptation (Holling, 1973; Adger, 2000; Carpenter et al., 2001). Resilience was popularized in the ecological sciences but has since expanded into the social sciences. The concept has been adapted for social-ecological systems in order to account for differences in ecosystem function and feedbacks resulting from human agency (Berkes and Ross, 2013; Béné et al., 2016).

Agroecosystems represent coupled social-ecological systems that are characterized by complex interactions and feedbacks between components, including ecosystem services and rural livelihoods (Adger, 2000; Liu et al., 2007; Bailey and Buck, 2016). Social-ecological resilience, then, provides an increasingly common framework for linking social and ecological dimensions of food systems (Callo-Concha and Ewert, 2014; Jacobi et al., 2018), and by extension, the concept is useful for linking nutritional and ecological functions of agroecosystems.

Resilience in agroecosystems is achieved through the capacities of actors in the system. Thus, the concepts of coping, adaptive, and transformative capacities are tied to resilience. While all of these capacities offer avenues toward system resilience, they function through different mechanisms and result in distinct outcomes (Béné et al., 2012). For example, coping capacities are normally employed when shocks are minor and the objective is to maintain system stability (i.e., persistence), whereas adaptive capacities are useful when incremental adjustments to the system are necessary to increase flexibility in the face of shocks. When shocks or stresses exceed levels for which coping and adaptive capacities are sufficient, transformative capacities can come into play to shift the nature of the system entirely.

Ecological Functions of Agroecosystems

Basic ecosystem processes, including fluxes of energy and nutrients and interactions among species, drive different ecosystem functions. Some examples of functions that are central to nutrient cycling in agroecosystems include primary production, decomposition, and biological nitrogen fixation by legume species. In a positive state, each of these functions enables agroecosystems to maintain soil fertility and productivity over time. When negative, loss of agroecosystem functioning occurs. Ecological processes happen in part due to abiotic and biotic conditions outside of farmers' control, but farmers are able

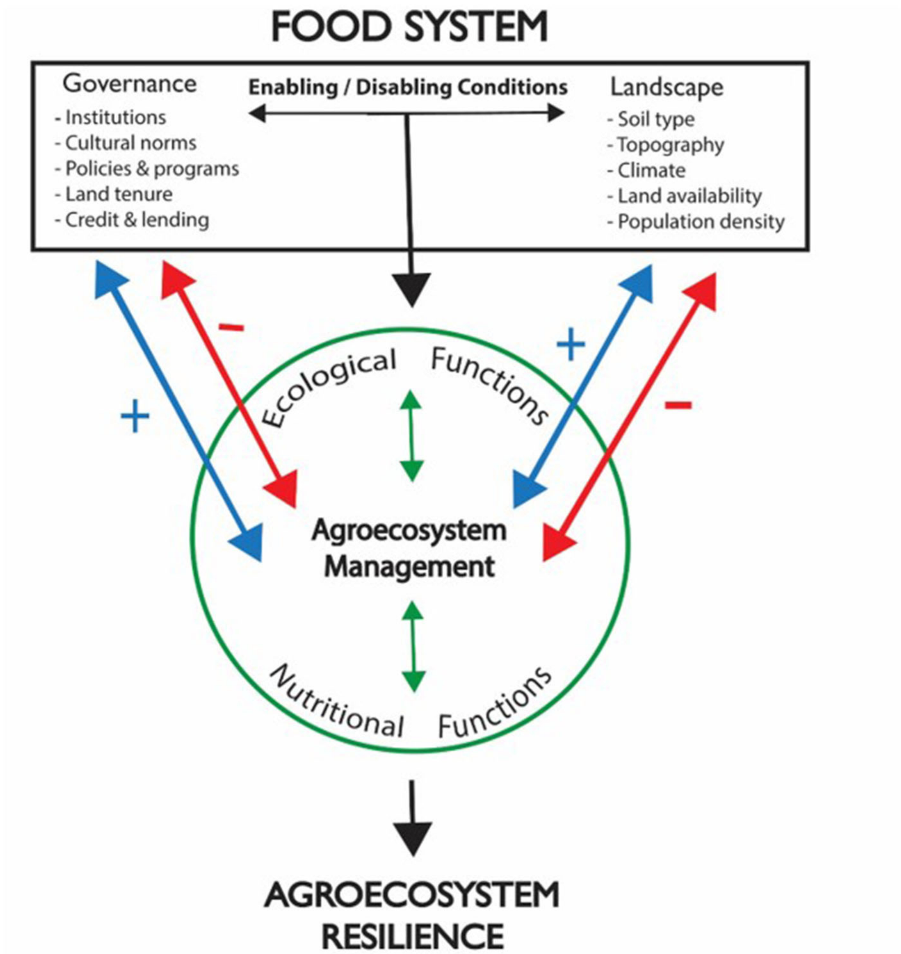

FIGURE 1 | Conceptual diagram of interactions between agroecosystem functioning and resilience. Farm management affects ecological and nutritional functions, which in turn drive changes in agroecosystem resilience. Green arrows inside the circle represent the interactions between ecological and nutritional functions, which are mediated by agroecosystem management practices and the adaptive capacity of farmers. Farm-level management decisions are embedded within food systems and associated governance and landscape conditions, which can trigger positive (+) or negative (–) feedbacks in agroecosystems.

to alter many agroecosystem functions through management practices (intentionally or unintentionally) (Drinkwater et al., 2008). Past work suggests that intentional management of ecological processes, or "agroecological management" (Kremen and Miles, 2012), results in resilience of desirable, or productive, states in agricultural systems (Peterson et al., 2018). Farm management strategies can therefore promote agroecosystem functioning and resilience (Bailey and Buck, 2016).

Agroecological management practices, in particular, can improve crop nutrient uptake in ecologically degraded systems by augmenting biotic interactions to enhance nutrient cycling (Brooker et al., 2016). For example, increasing the diversity of crop rotations with cover crops is a practice that can improve multiple ecosystem functions, or "multifunctionality" (Snapp et al., 2005; Finney and Kaye, 2016; Blesh, 2018). Among other functions, cover crops in the legume family supply nitrogen (N) and carbon (C) to soils through biological N fixation and photosynthesis. These N and C inputs add to pools of bioavailable soil nutrients, as cover crops are generally not harvested but instead are incorporated into the soil at the end of the season as "green manures." This agroecological practice has therefore been shown to increase internal nutrient cycling and nutrient availability to primary crops, with potential to increase productivity over time (Wander et al., 1994; Blesh, 2019). More broadly, the addition of legume cover crops to crop rotations introduces additional plant traits that influence ecosystem functions, contributing to both agricultural biodiversity (agrobiodiversity) and crop functional diversity (Wood et al., 2015).

Nutritional Functions of Agroecosystems

As an extension of ecosystem functioning, DeClerck et al. (2011) proposed that, given agroecosystems' primary goal of food production for human nutrition and health, nutritional functions of agroecosystems should be measured alongside their ecological counterparts. Although their study proposed one indicator of nutritional function—nutritional functional diversity—a broad suite of nutritional functions has rarely been considered in assessments of agroecosystem performance and resilience, nor have nutritional functions been explicitly related to underlying ecological functions.

Other nutritional functions of agroecosystems include the quantity, diversity and nutritional quality of crops produced, as well as maintenance of genetic resources to improve traits of individual crops and diet diversity (Remans et al., 2011; Allen et al., 2014). Importantly, these indicators of nutritional function consider more than just yield or productivity, which has been the dominant metric for assessing agroecosystem performance since the Green Revolution (Cassidy et al., 2013). Favoring productivity as the sole goal of agroecosystems can falsely place household food security and rural livelihoods at odds with critical ecological and nutritional functions (Zhang et al., 2007; Bennett et al., 2009; Nelson et al., 2009).

Just as farm management practices impact ecological functioning, they also affect nutritional functions, including nutritional quality of crops and potential for increased dietary diversity, as well as productivity. Ecological functions therefore affect the overall ability of an agroecosystem to provide nutritional functions to people—through the production of a diverse selection of nutritious foods. Extending beyond the agroecosystem level, recent high-profile reports have highlighted transitioning to agroecological management as an innovative approach to enhance food security and nutrition globally [McIntyre et al., 2009; The High Level Panel of Experts (HLPE) on Food Security Nutrition, 2019].

Agroecological management frequently entails ecological and nutritional diversification of cropping systems. Diversified farms have high levels of interaction between plant species, and between plants and microorganisms, which can maximize the efficiency of nutrient use on farms (e.g., Matson et al., 1997; Shennan, 2008; Kremen and Miles, 2012). When nutrient use efficiency (defined as total nutrient harvested/total nutrient input) of crops increases, there is greater uptake of nutrients by crop species, which can increase crop nutritional quality. Greater nutrient use efficiency also tends to correspond with reduced nutrient losses through runoff, leaching, or other pathways (Robertson and Vitousek, 2009). Such management practices thus have direct impacts on environmental sustainability as well as the quantity and quality of food produced and consumed (i.e., nutritional yield) in an agroecosystem. At the same time, it is important to acknowledge potential tradeoffs between management strategies that maximize ecological or nutritional functions (e.g., Power, 2010; Kremen and Miles, 2012), often by favoring short-term nutritional functions (e.g., crop yield or income from crop sales in a single season) over longer-term ecological ones (e.g., soil organic matter formation, C storage, and nutrient retention) (e.g., Steffan-Dewenter et al., 2007).

Continuing the example of farm diversification with cover crops, we can identify specific links between the ecological and nutritional functions derived from this practice. Nutritional functions include supporting crop yields with nutrient inputs from legume N fixation (Blesh, 2018), and increasing availability of other nutrients that can make crops more nutrient-rich, particularly soil phosphorus from solubilization by acidic and enzymatic root exudates (Hinsinger et al., 2011; Xue et al., 2016). Reduced soil erosion is also likely to improve crop yields and nutrient availability, especially if the system in question is a low-input smallholder farm on steep terrain (Vanek and Drinkwater, 2013). Increased yields in a resource-poor agricultural context could correspond to improved household food security or self-sufficiency, or to increased incomes, if crops are sold (Sibhatu et al., 2015b). Shifts in both ecological and nutritional functions of agroecosystems in this example illustrate the interactions resulting from farmer management decisions that can lead to agroecosystem resilience.

INDICATOR FRAMEWORK

Prior Frameworks for Agroecosystem Functioning and Resilience

Scholars have proposed frameworks for evaluating and enhancing food system resilience and sustainability (Tendall et al., 2015; Prosperi et al., 2016; Béné et al., 2019), rural landscape (Bailey and Buck, 2016) and livelihood resilience (Pelletier et al., 2016), working lands conservation and resilience (Kremen and Merenlender, 2018), and agricultural or agroecosystem resilience (van der Werf and Petit, 2002; Büchs, 2003; Cabell and Oelofse, 2012; Altieri et al., 2015; Blesh and Wittman, 2015; Prosperi et al., 2016), which include both environmental and food security indicators. These indicators have primarily considered the broader food system, including addressing questions of food supply, prices, and accessibility that drive food consumption, but have focused less on how these relationships play out in individual agroecosystems, and how management decisions at the farm level affect these broader food system dynamics.

Similarly, there is a robust literature on ecological functions that contribute to agroecosystem resilience, resulting in a number of indicator frameworks that identify or quantify ecological processes that stabilize farms faced with shocks (Büchs, 2003; Cabell and Oelofse, 2012; Callo-Concha and Ewert, 2014). In the most extensive agroecosystem-specific indicator framework for resilience to date (Cabell and Oelofse, 2012), the authors offer a suite of 13 social and ecological indicators derived from a review of the resilience literature. While this list is extensive, there are no nutrition-specific indicators, nor are there any direct measures of household food security, diet diversity, or diet quality incorporated into the framework (Cabell and Oelofse, 2012).

This trend holds across indicator frameworks for agricultural and livelihood resilience, which lack an explicit link between ecosystem nutrient availability and provisioning for human nutrition at the household level (Prosperi et al., 2016; Quinlan et al., 2016). Related work has addressed tradeoffs and synergies between agricultural yield and ecosystem services (Steffan-Dewenter et al., 2007; Bennett et al., 2009; Nelson et al., 2009); however, direct links to the underlying biophysical conditions and nutrient cycling regulated by farm management have not been made explicit in these frameworks. Additionally, more complex measures of nutritional provisioning that go beyond crop yield or market value are needed to fully capture resilience dynamics at the level of the agroecosystem (DeClerck et al., 2011; Remans et al., 2011; Wood, 2018).

A standard set of indicators for nutrition is the Food and Agriculture Organization's (FAO) four dimensions of food security: availability, access, utilization, and stability (FAO, 2008). As defined in the 1996 World Food Summit, food security is "physical and economic access to sufficient safe and nutritious food that meets [one's] dietary needs and food preferences for an active and healthy life" (FAO, 2008). The idea that adequate nutrition, and not only sufficient caloric intake, is required for long-term health is an important aspect of the FAO's definition and therefore critical to nutrient provisioning at the agroecosystem level (Jones et al., 2016). To relate ecological and nutritional functions of agroecosystems, then, food security and nutritional indicators should be integrated into frameworks for social-ecological resilience.

Indicators Linking Ecological and Nutritional Functions of Agroecosystems

We developed an indicator framework that pairs ecological and nutritional indicators of agroecosystem function (**Table 1**). Specifically, we conducted a literature review and selected ecological and nutritional indicators with demonstrated importance for agroecosystem functions and resilience in prior theoretical and empirical work. We then paired indicators based on known relationships between underlying ecological and nutritional functions. Finally, we linked the selected indicators to farm management strategies, coping, and adaptive capacity in smallholder agroecosystems. We tested potential metrics (i.e., measures of each indicator) for agroecosystem functions by applying the indicator framework to a case study, described below. Results from the case study demonstrate the utility of the framework and could be used to refine or adapt indicators and their measures in an iterative process of metric development and data analysis.

The literature review and paired indicator approach resulted in four pairs of ecological and nutritional indicators measurable at the agroecosystem level. Here, an agroecosystem is defined as the social and ecological components of a farm, including all land and species managed by a farm household. Our framework could apply to larger scales, such as landscapes or regions, but the conceptual framing and case study data presented in this paper are at the agroecosystem level. Researchers could use these indicators of agroecosystem function to support smallholders to adapt agricultural practices based not only on their ecological impacts but also their contributions to, or tradeoffs with, household nutritional needs.

Though they do not directly measure outcomes for human nutritional status, "nutritional" indicators in our framework relate conceptually to the FAO's four dimensions of food security: availability, access, utilization, and stability (FAO, 2008). These proxy measures enable relatively simple data collection and analysis compared to more complex, direct measures that may be less feasible in low-resourced contexts. Each of our indicator pairs aligns with one dimension of food security, offering proxies for food availability, access, and utilization, and tying them to ecological functions that may underlie their stability (**Table 1**). In line with the FAO dimensions, our framework integrates stability as a subcomponent of all other indicators rather than as its own indicator. We also incorporate both the quality and nutritional functional diversity of diets to offer a more comprehensive understanding of food "utilization."

Productivity. Total crop production per area (1E) and staple food availability (1N) are paired as *Productivity* indicators. Total crop production per area, or total yield, is broadly defined as the amount of food produced per harvested area. The ability of an agroecosystem to maintain productivity over time, even in the face of disturbance or environmental variability is called yield stability (Pimm, 1984; Raseduzzaman and Jensen, 2017). Sustained crop production indicates that soil fertility and associated nutrient cycling processes are functioning and able to produce staple crops for farm households. Many smallholders cultivate staples for direct household consumption, even when local markets are available (Isakson, 2009; Oyarzun et al., 2013). Therefore, this ecological indicator links to staple food availability, an indicator that represents the baseline nutritional (caloric) function of agroecosystems. Staple food availability can be defined as the capacity of a smallholder agroecosystem to provide sufficient quantities of staple crops to meet household caloric needs (FAO, 2008). This indicator is particularly relevant in smallholder systems, where a single crop (such as maize in Guatemala) can make up the majority of the diet (Fuentes Lopez, 2005). In the case of staple grains, *Productivity* generally supports caloric sufficiency of diets but may not guarantee nutrient sufficiency, necessitating indicators 2–4. *Productivity* indicators relate to the FAO dimension of food availability (FAO, 2008).

Diversity. Crop diversity (2E) and access to a diversified diet (2N) are paired as *Diversity* indicators. An agroecosystem with crop diversity contains species that fill distinct ecological niches. Crop species can vary over time, such as when cover crops are grown between harvested food crops in a rotation (Snapp et al., 2005), or species may overlap in space, through intercropping, for example. Diversified crop production in space and time contributes to long-term crop productivity (Vandermeer, 1989) and ensures household access to multiple types of crops at any given time of year, which is why it is paired with household access to a diversified diet. Access to a diversified diet is defined as on-farm availability of a diverse selection of edible crops whose nutritional complementarity increases diet quality (Remans et al., 2012; Jones, 2016) and is best measured using standardized methods of diet diversity (i.e., Minimum Dietary Diversity for Women, Dietary Diversity Score, or Food Variety Score). A diversified diet complements staple crop availability, as food availability is necessary but insufficient to complete a healthy diet. Households may gain access to a diversified diet by growing diverse crops, purchasing diverse crops, or through a combination of growing and purchasing foods (Jones, 2017). In smallholder contexts where markets remain inaccessible or unreliable, such as in our case study region, edible crop diversity is a robust indicator of access to a diverse diet. *Diversity* indicators relate to the FAO dimension of food access.

Quality. Beneficial species interactions (3E) and edible crop quality (3N) are paired as *Quality* indicators. Farmers can

TABLE 1 | Indicator framework for ecological and nutritional functions of agroecosystems.

Indicator pair	Ecological (E) or nutritional (N)	Indicator	Agroecosystem function
1 Productivity	E	Total crop production per area	Produce crops over time and under variable environmental conditions
	N	Staple food availability	Supply sufficient quantities of staple crops to meet household caloric needs
2 Diversity	E	Crop diversity	Fill distinct ecological niches and contribute to long-term productivity by varying crop species over time and in space
	N	Access to a diversified diet	Provide access to diverse food crops, potentially impacting diet quality
3 Quality	E	Beneficial species interactions	Facilitate crops' nutrient uptake, growth, and reproduction through beneficial interactions within and between trophic levels
	N	Edible crop quality	Increase crop nutrient content and elicit phytochemical responses through facilitative species interactions, improving crop nutritional quality for human diets
4 Functional Diversity	E	Functional diversity and redundancy	Enable a functional safety net by planting crops with diverse ecological functional traits and levels of associated non-crop species diversity
	N	Nutritional functional diversity	Fulfill nutritional needs for household diets by growing crop species that provide complementary and diverse nutrients

Indicators were adapted from prior frameworks, including Cabell and Oelofse (2012), the EC-FAO food security framework (FAO, 2008), and the sustainable diets literature (e.g., Allen et al., 2014).

foster beneficial species interactions within and between trophic levels by planting species known to facilitate other crops' growth and reproduction, such as growing leguminous crops alongside grasses or other non-legumes to stimulate nutrient uptake (Li et al., 2016), maintaining flowering species and natural vegetation on farms to attract pollinators (Garibaldi et al., 2013), or using ecological pest management (e.g., push-pull techniques) (Letourneau et al., 2011). As has been well-characterized in natural ecosystems in fields such as chemical ecology (Hunter, 2016b), these interspecific and inter-trophic interactions can affect the yield and nutrient content of harvested crops that contribute to household diets (Ahmed and Stepp, 2016; Dainese et al., 2019). Edible crop quality is a measure of the concentrations of crop nutrients important for human nutrition that vary based on environmental and management factors (Ahmed and Stepp, 2016). Positive species interactions enhance edible crop quality by increasing nutrient availability and uptake, such as through facilitation, niche partitioning, and increased nutrient use efficiency in multi-cropped systems (Zhang and Li, 2003; Brooker et al., 2015). They can also elicit phytochemical responses that may impact crop secondary metabolite concentrations relevant to human diets (Liu, 2003; Brandt et al., 2011; Hunter, 2016a). In agricultural landscapes with degraded or low-fertility soil, crop nutritional quality can decline sharply (Lal, 2009); it is therefore important, especially in regions with micronutrient deficiencies, including Guatemala, to consider management approaches that could improve the quality and not just the quantity of crops produced (Watson et al., 2012). In addition to management approaches, increasing protein or micronutrient concentrations in staple crops is a major goal of biofortification and other breeding initiatives aiming to improve diet quality beyond caloric sufficiency to reduce malnutrition (White and Broadley, 2009; Gunaratna et al., 2010). *Quality* indicators relate to the FAO dimension of food utilization.

Functional Diversity. Ecological functional diversity and redundancy (4E) and nutritional functional diversity (4N) are paired as *Functional Diversity* indicators. Ecological functional diversity and redundancy occur when an agroecosystem contains multiple crop types with differing functional traits, but with enough overlap in traits to provide an ecological safety net should one crop fail (Moonen and Bàrberi, 2008; Martin and Isaac, 2015; Wood et al., 2015). Increased crop diversity, the planned component of biodiversity, also increases associated (non-crop) biodiversity, which can further enhance agroecosystem functions, as well as buffering capacity and resilience (Altieri, 1999; Elmqvist et al., 2003). A mix of crops that encompasses distinct ecological functions (e.g., annual and perennial species) is also more likely to contribute to a broad range of nutrient requirements, represented here by the nutritional functional diversity indicator. Nutritional functional diversity is defined as the degree to which an agroecosystem fulfills nutritional functions for household diets by providing complementary and diverse nutrients across species on the farm (DeClerck et al., 2011; Luckett et al., 2015; Wood, 2018). Unlike the *Diversity* and *Quality* indicators, nutritional functional diversity accounts for the complete suite of nutrients present in different crop types, and evaluates the amounts, diversity, and evenness of nutrients across an agroecosystem, given human nutrient intake requirements (i.e., dietary reference intakes) (Remans et al., 2011). For example, out of two farms that have the same edible crop diversity (species richness = 3 for each farm), a farm that produces maize, beans (protein-rich), and sweet potato (high in vitamin A) offers greater nutritional functional diversity and can better meet nutritional requirements than does a farm that grows maize, cassava, and rice, all of which are carbohydrate-rich staple crops (DeClerck et al., 2011). *Functional Diversity* indicators relate to the FAO dimension of food utilization.

CASE STUDY APPLICATION

Case Study Selection

We tested our indicators for ecological and nutritional functions using data from a case study in a rural region of eastern Guatemala (**Figure 2**). The case study combined interview data with analysis of protein content of maize from farms to identify the status of ecological and nutritional functions of smallholder agroecosystems in the region, as well as interactions between functions. Our aim was to identify how interactions between ecological and nutritional indicators affect trends in agroecosystem function and resilience. While the metrics in the Guatemalan case study are site-specific, the indicator framework is designed to be both generalizable and adaptable to facilitate applications in other systems and regions.

Nationally, Guatemala suffers from the double burden of malnutrition, with the second highest global rate of childhood stunting (49%) and the highest rate in Latin America, along with growing presence of overweight and obesity (50% in women of reproductive age) (Black et al., 2008). These nutritional outcomes are closely correlated with poverty and ethnicity. Over 75 percent of Guatemala's indigenous population falls below the poverty line (Bygbjerg, 2012). According to national census data, indigenous peoples compose nearly 40 percent of the population (Instituto Nacional de Estadística, 2013). Of those, 7.6 percent are Q'eqchi' Maya, the primary sample population for this study.

The case study took place in the highly remote, eastern lowland region of Guatemala called Sarstún, in the Izabal department (**Figure 2**). Sarstún is an isolated and data-poor region with informal governance structures and few institutional resources. Tropical secondary forests, mangroves, subsistence farms, and small-scale fisheries characterize the hilly, coastal landscape. The forested areas bordering the Sarstún River were designated as a national protected area—the Sarstún River Multiple Use Area (SRMUA)—in 2005. However, due to its remoteness, few administrative resources have reached the majority-indigenous communities that reside there. The total population of the SRMUA is estimated to be slightly more than 4,000 people, distributed across 21 agricultural communities, the majority of whom (78%) are Q'eqchi' Maya (Coadministración, 2009).

Landscape trends in eastern Guatemala include local migration to the region, increased large-scale investments and acquisitions of land for cattle and palm oil plantations (i.e., "land grabbing"), and generalized degradation due to increasing pressure on forest resources (Alonso-Fradejas, 2012; Grandia, 2012). In SRMUA smallholder agroecosystems, as with approximately 65% of the Latin American farming population (Berdegué and Fuentealba, 2011), household nutrition is primarily dependent on local crop production and malnutrition is prevalent. With heightened pressures on land from both internal and external forces, Q'eqchi' smallholders and their agroecosystems are increasingly vulnerable to losses in both ecological and nutritional functions and decreased resilience in the face of shocks. This combination of contextual factors makes the SRMUA an appropriate test case for our ecological and nutritional indicator framework, as it can be used to identify and evaluate management practices that affect resilience on smallholder farms.

Data Collection and Analysis

Our sample included eleven villages (52% of total villages) and 60 households (∼10% of total population) in SRMUA, selected through a randomized sampling scheme in communities with ties to the local sustainable rural development non-profit APROSARSTUN (the Mayan Association for Rural Well-being in the Sarstún Region, Spanish acronym). We conducted semi-structured farmer interviews with 60 Q'eqchi' farmer households, from which we derived both categorical and continuous response variables for indicator analysis. We also analyzed maize samples from each household for nitrogen and protein concentrations, the case study metric (measure of the broader indicator) for edible crop quality in our framework application.

Interviews focused on maize production and household management of cornfields (*milpas*), along with discussion of crop outcomes and biophysical change over time (e.g., yield stability and soil fertility). Interviews were conducted either in Spanish (60%) or in Q'eqchi' Mayan through a translator. APROSARSTUN provided field assistants and translators for the study, which may have influenced farmer responses to questions regarding agroecological management practices, though precautions were taken to ensure unbiased responses. Four key informant interviews were also conducted with APROSARSTUN staff members to contextualize interview results and better define appropriate ecological and nutritional metrics for the case study. All interview guides and study materials were reviewed by the Tufts University Social, Behavioral, and Educational Research Institutional Review Board and were given exempt status under IRB study # 1403034. We received verbal consent from all participating members of each household prior to conducting interviews and collecting samples. The semi-structured interview questionnaire is available in **Appendix 2**.

Representative maize samples from the most recent harvest were collected from households following the interview ($n = 55$). Five households were unable to provide maize samples for the study, either because they only had access to purchased maize at home (not grown on their land) or because they did not have access to their grain storage at the time of the interview. Maize was then nixtamalized following a method first described by Bressani and Scrimshaw (1958); briefly, maize samples were heated in a 4% calcium hydroxide ($Ca(OH)_2$) solution at 94°C for 50 min, removed from the heat and left to stand for 14 h, washed, and transferred to a lyophilizer for 48 h to dry prior to grinding with a ball mill (Kleco). Ground samples were analyzed for % nitrogen by dry combustion in a CHNOS analyzer (vario MICRO cube, Elementar Americas, Mt. Laurel, NJ, USA) with L-Glutamic acid standards. Percent nitrogen was converted to maize % protein by multiplying by a conversion factor of 6.25 (Galicia et al., 2009).

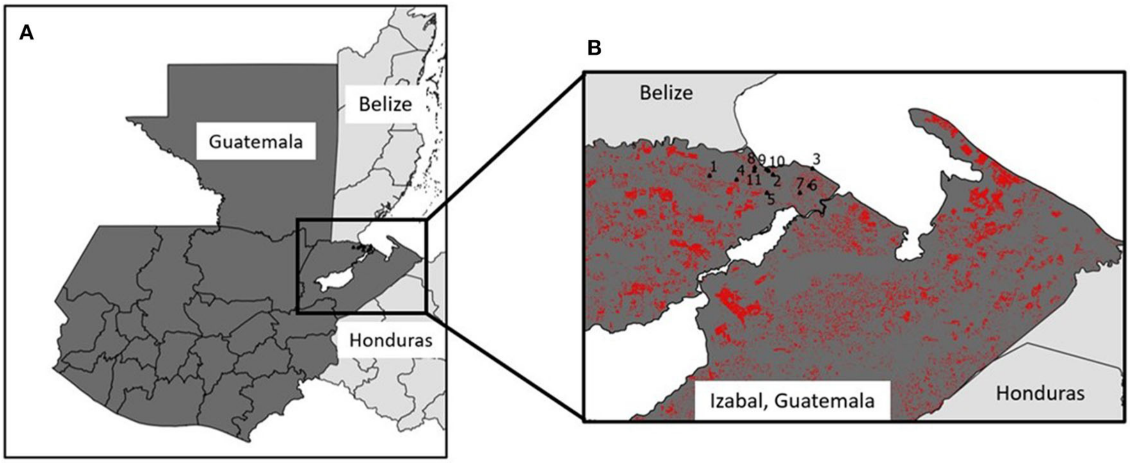

FIGURE 2 | Map of case study location and villages sampled in the Izabal Department of Guatemala. **(A)** Shows Guatemala alongside its neighboring countries of Belize and Honduras, both of which are adjacent to the study site shown in the inset map. **(B)** Shows a closer view of the eleven villages included in the case study, which are located in the Sarstún Region of Guatemala just south of the Sarstún River, which forms the border with Belize. Red pixels on the inset map represent forest loss from 2000–2014, the year that interviews were conducted. Source: Hansen/UMD/Google/USGS/NASA.

Indicators and Statistical Analysis

Table 2 provides an overview of how case study data were converted into metrics used in the indicator framework. Because we used an existing dataset to test the framework, we were unable to include diet diversity data, which was not a component of the original study questionnaire (**Appendix 2**). Based on available data, we tested three of the four paired indicators (*Productivity, Quality,* and *Functional Diversity*, but not *Diversity*) in our case study. Reflecting the difficulty of acquiring fine-scale quantitative data in the case study region (e.g., on crop yields), the majority of case study variables were categorical, with the exception of crop species richness data used to calculate ecological and nutritional functional diversity (*Functional Diversity* indicators), which were continuous. We therefore transformed all raw data into categorical variables with three levels (0, +1, −1) to give each metric equal weight within the framework.

Categorical values of 0, +1, and −1 were assigned to each farm household for each indicator, representing the neutral, positive, or negative status of a given indicator on farms. For each case study variable, we determined the range for a negative, neutral, or positive status using the following approach: (1) if there was a standard or mean value or range of values considered "sufficient" in the literature (e.g., mean protein concentration in maize from the FAO), we made this the 0 or "neutral" value; however, (2) if there was a clear scientific rationale for an indicator (e.g., high crop diversity is more ecologically beneficial than low crop diversity) but no way to quantify a "neutral" value, we scaled indicator values relative to the maximum value in the sample. This approach enabled us to determine the relative status of agroecosystem functioning on farms, given regional conditions, and to identify the most marginalized or vulnerable farms as well as those with relatively improved outcomes (and their associated management practices). Summary statistics for the raw data from the case study are presented in **Appendix 1** (**Table A1.1**), along with distributions of all raw data used to define case study metrics and their assigned categories for analysis (**Figure A1.1**).

To demonstrate the potential of the indicator framework to analyze data at a finer resolution—where such data is available for smallholder agroecosystems—we conducted a more detailed analysis of ecological and nutritional functional diversity (*Functional Diversity* indicators) using continuous rather than categorical values. We tested for a relationship between the *Functional Diversity* indicators using simple linear regression. To quantify ecological functional diversity of agroecosystems (i.e., crop functional diversity), we used the open software platform FDiversity (Casanoves et al., 2011) to calculate Rao's Quadratic Entropy (Q) based on crop species richness and area data for each farm (Botta-Dukat, 2005; Schleuter et al., 2010) (**Table A3.1**). Crops were evaluated for key plant functional traits by determining their binary (yes or no) associations with each of the following functional categories: perennial, C3 grass, C4 grass, forb or broadleaf, nitrogen-fixing, woody, and vining/groundcover (**Table A3.2**). Similarly, to calculate nutritional functional diversity, we used average concentrations and dietary reference intakes of 17 essential dietary nutrients in the 29 food crops cultivated by farmers in the sample, along with crop-specific area data for each farm (**Tables A3.3, A3.4**) (Remans et al., 2011). FDiversity software was also used to calculate Rao's Q values for nutritional functional diversity measures. Although agrobiodiversity data was an initial input for calculating both ecological and nutritional functional diversity indicators, this data was processed with functional group (for ecological functional diversity) and nutrient data (for nutritional functional diversity) for each crop prior to analysis and metric calculation for the case study. All functional traits were weighted on an equal basis in the analysis.

After quantifying all indicators for our sample through the above metrics, we evaluated relationships between farm

TABLE 2 | Indicators of ecological and nutritional functions applied to a case study of smallholder agroecosystems in eastern Guatemala.

Indicator pair	Ecological (E) or nutritional (N)	Indicator	Case study metric	Metric calculation	Interview question (translated)
1 Productivity	E	Total crop production per unit area	Crop yield over time	Neutral (0), increasing (+1), decreasing (−1)	Have you seen a difference in the productivity of your cornfields ("milpas") since you began farming here? If so, how has it changed (has it increased, decreased, or stayed the same), and why?
	N	Staple food availability	Deficit/surplus maize yield	Sufficient maize for household (0), sells maize (+1), buys maize (−1)	How much maize did you produce last year, and what percent of it went feed your household? Was it sufficient to feed your household? If there was maize remaining, how much of it did you sell or exchange?
2 Diversity	E	Crop diversity	Agrobiodiversity	4–6 species (0), 7 or more species (+1), 0–3 species (−1)	How many crops do you plant during the main growing season? What about the dry season ("matahambre")? (prompt with list of crops, if needed)
	N	Access to a diversified diet	(not measured in case study)	–	–
3 Quality	E	Beneficial species interactions	Multi-cropping	Multiple crops in monoculture (0), multiple crops in polyculture (+1), single crop in monoculture (−1)	Do you grow more than one crop? If so, do you plant your crops together in the same field, or in separate fields?
	N	Edible crop quality	Maize protein concentration	Average maize % protein range from FAO (8–11%) (0), >11% protein (+1), <8% protein (−1)	Do you have any white corn cobs from your last harvest? Would it be possible for me to take some grains from your corn as a sample to test its nutrients?
4 Functional Diversity	E	Functional diversity and redundancy	Crop functional diversity	Rao's Quadratic Entropy (Q) of 1.87–3.83 (0), Q of 3.84 and above (+1), Q of 0–1.86 (−1) (quantile cutoffs)	What is the total planted area of each crop you grow? (prompt with list of crops)*
	N	Nutritional functional diversity	Nutritional functional diversity	Rao's Quadratic Entropy (Q) of 0.0057–0.013 (0), Q of 0.014 and above (+1), Q of 0–0.0056 (−1) (quantile cutoffs)	What is the total planted area of each crop? (prompt with list of crops)**

*Raw data were subsequently transformed using crop functional trait data prior to analysis (**Table A3.2**).*
Raw data were subsequently transformed using crop nutrient data prior to analysis (Tables A3.3, A3.4**).*
*Case study metrics (measures of the broader indicators) are shown with categorical levels, in the order: neutral (0), positive (+1), negative (−1). All indicators were derived from interview data except edible crop quality, which was measured by analyzing maize grain samples for protein concentration. There were no available case study data on dietary diversity; therefore indicator 2N was not included in analyses. For distributions of the continuous and categorical variables used in metric calculations, see **Figure A1.1**. The complete interview questionnaire can be found in **Appendix 2**.*

management practices and ecological and nutritional functions of agroecosystems in two steps. First, we created an integrated Agroecosystem Function Index (AFI) for each farm using ecological and nutritional indicators in our framework. We quantified the overall agroecosystem functioning of each farm in the case study as a single number between −6 and +6 (the range would be −8 to +8 for all four pairs of indicators). AFIs were derived by summing all values (0, +1, and −1) for the three ecological and three nutritional indicators for a given farm household, resulting in a cumulative positive, negative, or zero (neutral) value. AFI values closer to +6 or −6 indicated stronger positive or negative functional states on farms, respectively. Then, to evaluate tradeoffs and synergies between pairs, we summed the coded values (0, +1, −1) for each ecological-nutritional indicator pair on each farm and used contingency analysis to test the null hypothesis that each pair of variables was independent across farms (**Table 4A**). Using 15 contingency tables, we then analyzed the level of co-occurrence (synergy) or existence of opposite trends (tradeoffs) between ecological and nutritional functions in pairs and non-pairs across all farms in the sample (**Tables 4B,C**).

Finally, we complemented our quantitative analysis with qualitative analysis of interview data, operationalizing our indicator framework to assess smallholder resilience. For this analysis, we used coded interview data to identify community-level and idiosyncratic shocks on farm households in our sample. Farmers' demonstrated abilities to respond to these shocks were categorized into coping and adaptive capacities. Coping capacities were defined as strategies that enabled farm households to persist in agriculture without qualitative changes to the structure of the agroecosystem. Adaptive capacities were defined as farmer changes to agroecosystem management meant to increase flexibility and improve outcomes in the face of shocks. Farms that showed both coping and adaptive capacities were considered adaptive. Adaptive farm management strategies were categorized into three groups: ecological, market-oriented, and hybrid strategies. Ecological strategies included maintaining high levels of soil cover, using nitrogen-fixing perennial and annual species to replenish soil fertility, growing a diversity of crops, and refraining from burning farm fields. Market-oriented strategies included growing hybrid maize varieties in monoculture, applying higher rates of herbicides, insecticides, and inorganic fertilizers, and focusing production on a smaller number of crops to bring to market. Hybrid approaches included increasing the diversity of perennial tree crops to sell to specialty markets, as well as other combinations of agrichemical application and use of nitrogen-fixing plant species. As a test of the framework's ability to assess agroecosystem function-resilience dynamics, we quantified relationships between farmer coping and adaptive capacities and ecological and nutritional indicator values on farms using analysis of variance (ANOVA) models, described below. Following previous definitions of resilience capacities (Béné et al., 2012), transformative capacities could also be assessed using the framework. However, there was not sufficient evidence of transformative capacity in the sample to include it in our analysis. Transformative management strategies could include conversion from traditional crops to a novel production system or migration to an urban environment, for example.

Data were analyzed using R software (version 3.6.2, "Dark and Stormy Night") (R Core Team, 2019). We used a Kendall's tau rank correlation to assess the association between ecological and nutritional components of the AFI across farms. McNemar's Chi-squared tests were run on contingency tables corresponding to each of the ecological and nutritional indicator pairs, with the exception of *Functional Diversity* indicators, which had a high number of neutral (0) values and required a Fisher's Exact test (**Tables 3**, **4**). We assessed the relationship between our ecological and nutritional *Functional Diversity* indicators using a simple linear regression model with the lm function in R (R Core Team, 2019). Also in R, we performed ANOVA followed by Tukey's Honestly Significant Difference *post-hoc* tests to assess statistical differences in ecological and nutritional indicator values on farms. One set of analyses focused on shocks and a second set analyzed coping and adaptive capacities. We used three separate mixed-effects models for each of these analyses, each of which included either the AFI (sum of ecological and nutritional indicators per farm), the nutritional component of the AFI (sum of nutritional indicators per farm), or the ecological component of the AFI (sum of ecological indicators per farm) as the response variable. Shock type and capacity type were the main effects in the two sets of analyses, respectively, with village as a random effect in all models. We explored the community-level shock of land-grabbing using a separate ANOVA model with no random effect, comparing farms in villages that did or did not have a land grab according to interview data. Ninety-five percent confidence intervals were used to assess statistical significance.

Case Study Results

Synergies and Tradeoffs Between Ecological and Nutritional Indicators

At the agroecosystem level, case study data showed a positive relationship between ecological and nutritional functions. There was a significant and positive rank correlation between the ecological and nutritional components of the AFI across farms [Kendall's tau $= 0.58$, $z = 5.7$, $p < 0.0001$ for sum(E) and sum(N)]. This means that farms with positive levels of functioning based on ecological indicators were significantly more likely to also have higher values for nutritional indicators, and vice-versa.

Analyses of individual case study indicator pairs, however, showed both significant positive (synergistic) and negative (tradeoff) relationships (**Table 3**). Across all farms, there were positive and neutral relationships between ecological and nutritional indicators for indicators 3 and 4 (*Quality and Function)*, but there were tradeoffs within pairs for indicator 1 (*Productivity*). Our more detailed analysis of *Functional Diversity* indicators showed a strong positive relationship between ecological and nutritional functional diversity across farms in the sample ($n = 60$ farms, adjusted $R^2 = 0.74$, $F = 171.6$ on 1 and 58 df, $p < 0.0001$). This result provides evidence that higher crop functional diversity on smallholder farms increases the likelihood that farms will also produce crops that offer a diverse array of essential nutrients in amounts relevant to human dietary adequacy (Wood, 2018).

Negatively correlated indicator pairs provided evidence that certain shocks and farmer responses to them led to tradeoffs between ecological and nutritional functions of agroecosystems in the Guatemalan case study (**Table 3**). Tradeoffs could signify a time lag between ecological degradation and negative nutritional functions (e.g., malnutrition), as well as the coping capacity of smallholder farm households, including by using off-farm labor to supplement income and purchase food when soil degradation leads to low agroecosystem yields (Vanek and Drinkwater, 2013). Case study results for *Productivity* indicators, for example, suggest that despite declining yields in the majority of study households, most farms still produced sufficient maize to sell some surplus to neighbors (**Tables 2**, **3**). Tradeoffs between positive nutritional indicators and negative ecological indicators may reflect management strategies that increase yields or food provisioning in the short term but degrade the natural resource base over time (**Table 5**).

In addition to testing the indicator pairs in the framework, we also tested relationships across the full set of indicators and found that 67% of indicator combinations (ecological-nutritional, ecological-ecological, and nutritional-nutritional) were non-independent (i.e., were related) according to McNemar's Chi-squared and Fisher's Exact Tests (**Table 4B**). Although many

TABLE 3 | Results of contingency analysis between individual pairs of ecological and nutritional indicators, representing tradeoffs or synergies, across all farms ($n = 60$) in a case study in eastern Guatemala.

Indicator	Ecological metric	Nutritional metric	Test statistic†	*df*	*p*-value	Synergy or tradeoff?
1 Productivity	Crop yield over time	Deficit/surplus maize yield	26.73	3	6.70×10^{-6}	**Tradeoff:** negative ecological; positive nutritional
2 Diversity	Agrobiodiversity	(data not available for case study)	–	–	–	–
3 Quality	Multi-cropping	Maize protein concentration	25.67	3	1.1×10^{-5}	**Synergy**
4 Functional Diversity	Ecological crop functional diversity	Nutritional crop functional diversity	N/A	3	2.2×10^{-16}	**Synergy**

†Test statistic for indicators 1 and 3 was McNemar's Chi-squared. For indicator 4, we used Fisher's exact test to evaluate the statistical likelihood that rows and columns in the contingency table were non-independent (alternative hypothesis).
*All indicators showed significant positive (indicators 3 and 4) or negative (indicator 1) within-pair relationships at a 95% confidence interval. See **Table 4** for a visual representation of the analysis.*

indicators were non-independent, only four non-paired indicator relationships had clear directionality in the contingency analysis (**Table 4C**). We found a weak but significant negative synergy between maize yield over time (1E; negative) and maize protein concentration (3N; neutral) ($\chi^2 = 26.3$, df $= 3$, $p = 9.7 * 10^{-6}$) (**Table 4**), indicating that both quantity and quality of maize may be affected by degradation in the case study region. There was also a significant tradeoff between maize yield over time (1E; negative) and multi-cropping (3E; positive) ($\chi^2 = 26.8$, df $=$ 3, $p = 6.5 * 10^{-6}$), which could be evidence that farmers are increasing their use of beneficial species interactions as their staple crops become less productive. Finally, multi-cropping (3E) had a significantly positive association with maize protein concentration (3N), and with both ecological (4E) and nutritional functional diversity (4N) (**Tables 4B,C**), meaning that farmers with higher crop functional diversity were also more likely to intercrop species, which was also positively related with crop quality. These results indicate that a holistic approach to assessing relationships across the framework could yield a more comprehensive understanding of ecological and nutritional functions of agroecosystems than indicators paired using theory and prior empirical understandings alone.

Farmer Capacities Mediate Agroecosystem Functioning and Resilience to Shocks

Agroecosystem Function Index (AFI; our proxy for the overall status and direction of combined ecological and nutritional indicators) values for the case study ranged from −6 at the lowest (1 farm) to 6 at the highest (3 farms). Most farms were characterized by a combination of positive, negative, and neutral levels of different agroecosystem functions, leading to a median AFI of 1 and a mean slightly above 0.

We operationalized our indicator framework to examine farmers' resilience, with a focus on coping and adaptive capacities, in the face of landscape-level, community-level, and idiosyncratic household shocks. Across all villages, farmers faced the landscape-level shock of deforestation. Farmers also identified one major community-level shock: acute loss of land due to large-scale land acquisitions (or "land grabs") that reduced or eliminated land ownership in five out of eleven villages in the sample (Alonso-Fradejas, 2012). Land grabs for rubber plantations, oil exploration, and cattle ranching were cited as reasons for farmers' loss of agricultural and common lands. Idiosyncratic, or household-level, shocks were also recorded for all farms and reflected the effects of broader biophysical patterns (e.g., climate changes), community-level shocks (e.g., direct household losses of land due to land grabs), as well as more localized problems (e.g., pest pressure, low soil fertility). Primary household shocks mentioned during interviews included climate shocks (shifts in rainfall patterns and rapid temperature changes that caused crop damage), degradation of cropland (generally related to deforestation) that led to yield losses, loss of land ownership or tenure from land grabs, increased pre- and post-harvest pest pressure, and combinations of these. Only three farmers (5% of the sample) stated that they had not experienced any changes that affected their livelihoods in the last decade, and all of these were younger farmers who had < 10 years of experience as heads of farming households.

We used the AFI to test household response to community-level and idiosyncratic shocks, thereby assessing their adaptive capacity and resilience to agroecosystem disturbance. Both community-level and household-level shocks led to significant differences in indicators of agroecosystem functioning (**Figure 3**). Relative to the generalized landscape-level shock of deforestation, the acute shock of land grabbing led to significantly lower nutritional indicators in affected villages, driving a lower AFI and resilience on farms that had experienced land grabs (**Figure 3A**). Households experienced inconsistent effects of community-level shocks. Farms that experienced direct losses of land due to coupled landscape degradation and land grabbing had significantly lower nutritional indicators in the AFI than those who did not experience land losses (**Figure 3B**). However, there were no significant effects of household-level shocks on the overall AFI, likely due to non-significant differences in farm ecological indicators by shock type.

TABLE 4 | Matrices showing relationships between ecological and nutritional indicators for 60 farms in the Guatemalan case study.

(A) Visual representation of contingency analysis for paired indicators

Indicator		Contingency table			Outcome
		−1	0	1	
1 Productivity	−1	10	8	19	**Tradeoff**
	0	4	3	9	
	1	0	1	5	
		−1	0	1	
3 Quality	−1	2	4	1	**Weak Synergy**
	0	2	11	0	
	1	3	24	6	
		−1	0	1	
4 Functional diversity	−1	18	2	0	**Synergy**
	0	2	16	2	
	1	0	2	18	

(B) Test statistic (McNemar's χ^2 or Fisher's Exact) and p-value matrix

	1E	3E	4E	1N	3N	4N
1E	NA	$\chi^2 = 26.8$ $p = 6.5*10^{-6}$	$\chi^2 = 12.7$ $p = 0.005$	**$\chi^2 = 26.7$ $p = 6.7*10^{-6}$**	$\chi^2 = 26.0$ $p = 9.7*10^{-6}$	$\chi^2 = 12.7$ $p = 0.0053$
3E	$\chi^2 = 26.8$ $p = 6.5*10^{-6}$	NA	$\chi^2 = 14.7$ $p = 0.002$	$\chi^2 = 3.7$ $p = 0.3$	**$\chi^2 = 25.7$ $p = 1.1*10^{-5}$**	$\chi^2 = 14.31$ $p = 0.0025$
4E	$\chi^2 = 12.7$ $p = 0.005$	$\chi^2 = 14.7$ $p = 0.002$	NA	$\chi^2 = 6.0$ $p = 0.1$	$\chi^2 = 14.8$ $p = 0.002$	**Fisher's exact $p = 2.2*20^{-16}$**
1N	**$\chi^2 = 26.7$ $p = 6.7*10^{-6}$**	$\chi^2 = 3.7$ $p = 0.3$	$\chi^2 = 6.0$ $p = 0.1$	NA	$\chi^2 = 26.3$ $p = 8.4*10^{-6}$	$\chi^2 = 5.0$ $p = 0.17$
3N	$\chi^2 = 26.0$ $p = 9.7*10^{-6}$	**$\chi^2 = 25.7$ $p = 1.1*10^{-5}$**	$\chi^2 = 14.8$ $p = 0.002$	$\chi^2 = 26.3$ $p = 8.4*10^{-6}$	NA	$\chi^2 = 14.7$ $p = 0.002$
4N	$\chi^2 = 12.7$ $p = 0.0053$	$\chi^2 = 14.3$ $p = 0.0025$	**Fisher's exact $p = 2.2*20^{-16}$**	$\chi^2 = 5.0$ $p = 0.17$	$\chi^2 = 14.7$ $p = 0.002$	NA

(C) Direction and outcome of indicator relationships from contingency analysis

	1E	3E	4E	1N	3N	4N
1E	NA	Tradeoff	No relationship	**Tradeoff**	(–) Synergy	No relationship
3E	Tradeoff	NA	(+) Synergy	No relationship	**(+) Synergy**	(+) Synergy
4E	No relationship	(+) Synergy	NA	No relationship	No relationship	**(+/–) Synergy**
1N	**Tradeoff**	No relationship	No relationship	NA	(+) Synergy	No relationship
3N	(–) Synergy	**(+) Synergy**	No relationship	(+) Synergy	NA	No relationship
4N	No relationship	(+) Synergy	**(+/–) Synergy**	No relationship	No relationship	NA

(A) Contingency tables showing co-occurrence of values of −1, 0, and +1 for ecological and nutritional indicators within each pair. Ecological indicators are the rows and nutritional indicators are the columns. Greater co-occurrence appears in green, with smaller co-occurring values appearing in yellow and orange (0). The greatest co-occurring value for each indicator pair (in bold) was labeled as either a synergy (+1 and +1, −1 and −1, 0 and 0, 0 and +1) or tradeoff (−1 and +1 or vice-versa). (B) Values for McNemar's χ^2 test statistics and p-values are presented for relationships between all indicators in the case study, including cross-tabulations between non-paired indicators. Values were considered significant at a 95% confidence interval. Only indicator pair 4 required a different test statistic due to a high number of neutral (0) values, and Fisher's Exact Test was used. Relationships between paired indicators are shown in bold; all paired indicators were significantly related. Greener cells represent stronger relationships. The strongest relationship was between the Functional Diversity indicators, 4E, and 4N. (C) Relationships between all ecological and nutritional indicators. Relationships between paired indicators are shown in bold, with positive synergies in green, negative synergies in red, and tradeoffs in yellow. Non-paired indicators had positive relationships (synergies), with the exception of the negative relationship (tradeoff) between ecological Productivity (1E) and Quality (3E) indicators.

The qualitative interview analysis revealed how coping and adaptive capacities can drive differentiated outcomes in smallholder agroecosystems (**Figure 4**). Farms relying on coping capacities ($n = 20$) by working for or renting land from plantation owners had significantly lower ecological and nutritional indicators than farms with higher adaptive capacities ($n = 40$). While there were many distinct adaptation strategies farmers used to respond to shocks, they tended to follow either an ecological ($n = 23$), a market-oriented ($n = 8$), or a hybrid (both ecological and market-oriented; $n = 9$) approach, as described in section Indicators and Statistical Analysis. **Table 5** highlights specific management characteristics on farms that

TABLE 5 | Coping and adaptative strategies used by farmers in the Guatemalan case study to respond to community-level and idiosyncratic shocks.

Farmer resilience capacity	Management approach	Household shock type	Practices used to recover from shock
Adaptive	Ecological ($N = 23$)	**Climate change ($N = 13$)** Land degradation ($N = 8$) Pest and weed pressure ($N = 8$) Land dispossession ($N = 6$) None ($N = 0$)	**Legume cover cropping ($N = 10$)** Agroforestry ($N = 8$) Crop diversification ($N = 4$) Maize variety diversification ($N = 4$) Manual pest control ($N = 3$) Seed saving and exchange ($N = 3$) Polyculture ($N = 2$) Sustainable tourism ($N = 1$)
Adaptive	Market-oriented ($N = 8$)	Climate change ($N = 1$) **Land degradation ($N = 4$)** Pest and weed pressure ($N = 3$) Land dispossession ($N = 1$) None ($N = 1$)	Hybrid and transgenic seed varieties ($N = 2$) Fertilizer use ($N = 2$) Pesticide use ($N = 1$) **Grows some crops only for market (not home consumption) ($N = 5$)**
Adaptive	Hybrid ($N = 9$) (ecological + market-oriented)	Climate change ($N = 1$) **Land degradation ($N = 4$)** Pest and weed pressure ($N = 1$) Land dispossession ($N = 3$) None ($N = 1$)	**Legume cover cropping ($N = 6$)** Agroforestry ($N = 2$) Tree crops for market ($N = 2$) Seed saving and exchange ($N = 2$) Hybrid seed varieties ($N = 3$) Fertilizer use ($N = 3$) Pesticide use ($N = 3$) Grows some crops only for market (not home consumption) ($N = 4$)
Coping	Coping ($N = 20$)	Climate change ($N = 6$) Land degradation ($N = 4$) Pest and weed pressure ($N = 6$) **Land dispossession ($N = 12$)** None ($N = 1$)	**Rent land from plantation owner ($N = 12$)** Off-farm work on plantation ($N = 7$) Off-farm traditional work* ($N = 3$) Increase farm workload of female head of household ($N = 2$) Fertilizer use ($N = 1$) Increase pesticide use ($N = 6$) Rely on communal land ($N = 1$) Purchase all maize for household consumption ($N = 2$)

**Off-farm traditional work includes tasks such as fishing and practicing traditional medicine (as a "curandero") as an alternative to farming.*
The number of farmers that mentioned each shock type and management practice is listed in parentheses (N). Farmers that described more than one shock or practice in response to a shock are counted multiple times. The most common shocks and practices used by farmers in each group are shown in bold.

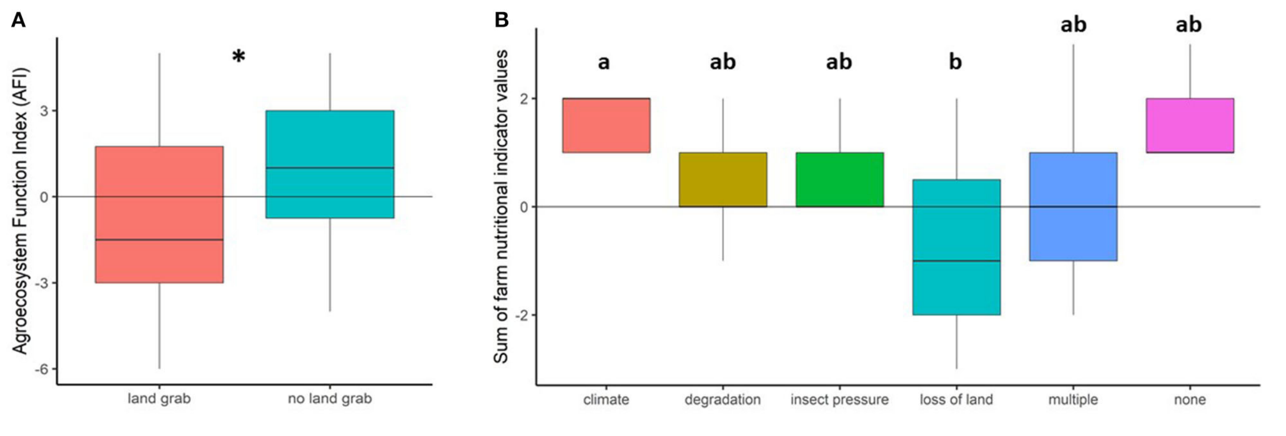

FIGURE 3 | Shocks in smallholder agroecosystems in Sarstún, Guatemala. **(A)** Shows the Agroecosystem Function Index (AFI) for farms experiencing a major community-level shock, land grabbing, and those subject to only generalized deforestation as a landscape-level shock (no land grab). **(B)** Represents the suite of household-level shocks experienced by smallholders in Sarstún and their relationships with the nutritional component of the AFI (sum of nutritional indicator values by farm). There were no significant differences by shock type for the ecological component or the overall sum of indicator values, the AFI. $N = 60$ farms. In **(A)**, * Indicates $p < 0.05$. In **(B)**, different letters indicate significant differences in indicator values for distinct household shocks ($p < 0.05$; Tukey's HSD).

used coping and adaptive capacities to respond to shocks. Nearly all farms, regardless of their AFI, used some combination of these management approaches.

Ecological and hybrid adaptation strategies were associated with high AFIs, whereas market-oriented and coping strategies resulted in lower ecological and nutritional indicators (**Figure 4**). Ecological and hybrid approaches made use of velvet bean (*Mucuna pruriens* (L.) DC.), a leguminous cover crop, to improve soil fertility and ecosystem functioning for long-term crop production. In an interview, when asked his preferred way to improve soil fertility, one farmer asserted, "The most beneficial way is with velvet bean ("frijol abono," which means "fertilizer bean") because it is the most common and economical in the region, but it is growing more difficult to find the seeds because the practice is fading." Interview data confirmed that while 50% of the sample planted velvet bean at the time of the study, an additional 18% of the sample (68% overall) had previously used velvet bean but abandoned the practice because they now rent land or find it more difficult to save seeds to re-plant when using herbicides. Collectively, these findings suggest that adaptive capacity may be declining in the region.

Negative AFI values commonly resulted from community-level shocks that reduced households' adaptive capacity and ability to manage farms for improved agroecosystem functioning (**Figure 4**). A common example was loss of land to plantation owners ($n = 26$ farms), which led farmers to engage in coping strategies, such as decreasing their crop diversity and use of cover crops, along with shortening cycles of swidden management. These new practices reduced forest cover, which many farmers noted in interviews as a principal reason for soil fertility declines that decreased yields. One older farmer mentioned that his yields had fallen by more than 50% in his lifetime, because "before there were large areas of secondary forest that helped the soil and led to good crop yields. There are no longer mature forested areas and the soil is poor, which affects the productivity of the soil, when we burn areas with little forest cover."

Lack of land tenure also shifted farmers' management strategies toward coping when they began renting land to grow their milpas (cornfields) ($n = 19$ farms). Renting led households to reduce the number of crops and their investment in soil conservation, as farmers had little incentive to use agroecological management practices on land they did not own. One farmer commented, "Yes, I use velvet bean, but there is no true guarantee of soil conservation because my land is rented and I don't have a specific plot; I don't know which plot I'll get next year." Insecure land tenure particularly affected households that had recently immigrated to the region and those in close proximity to plantations that now own what was once community land. Farms at a greater distance from plantations also shifted their management as farmers went to work as day laborers. One such farmer explained his monoculture milpa by saying, "Now we don't have any other crops [in addition to maize] because we have to work on the plantation, which is a 3 h walk from here."

Market-oriented households adapted their management to shocks by growing some crops, such as hybrid or transgenic maize varieties, solely for sale on the commercial market to increase their purchasing power (**Table 5**). While market-oriented farms had on average higher AFI values than farmers relying on coping capacity alone, both coping and market-oriented households' AFIs were significantly lower than those from households using ecological or hybrid approaches (**Figure 4**). Despite evidence of coping and adaptive capacities in the face of changing landscape conditions, the capacity of smallholders to transform their agroecosystems as they underwent community-level and idiosyncratic shocks appeared limited in the case study.

DISCUSSION

Sustaining or enhancing ecological and nutritional functions of agroecosystems is necessary to foster the adaptive capacity and resilience of smallholders. To this end, our study paired ecological and nutritional indicators of agroecosystem function in a novel indicator framework, elucidating previously neglected relationships between management practices, ecological functions, and provisioning of nutrients in harvested crops. We then applied this indicator framework to a case study in a remote region of Guatemala to test its ability to identify (1) synergies and tradeoffs between ecological and nutritional functions in a smallholder context, and (2) farmer capacities and management practices that shape agroecosystem resilience in the region.

Agroecosystem Functioning, Adaptive Capacity, and Resilience in Eastern Guatemala

Overall, our analysis of smallholder farms in eastern Guatemala illustrates a suite of synergistic and tradeoff dynamics between ecological and nutritional functions and resilience at the agroecosystem level. We tested three indicator pairs, *Productivity, Quality,* and *Functional Diversity,* in the case study, which revealed two synergistic (*Quality, Functional Diversity)* and one tradeoff (*Productivity)* relationship within a context of community and household-level shocks. Overall, six of the 15 unique combinations of indicators were significantly positively related, and two had significant tradeoffs. Positive relationships between indicators indicated that farmers tended to use multiple ecological practices simultaneously, or not at all, and that ecological and nutritional outcomes tended to be synergistic (either both negative or both positive). Results from our regression analysis of ecological and nutritional functional diversity (*Functional Diversity* indicators in the framework) further supported the positive relationship between ecological and nutritional functions and resilience of agroecosystems using continuous data.

In some cases, there were tradeoffs between ecological and nutritional indicators in the Guatemalan case, particularly between maize yield over time (1E; a *Productivity* indicator) and both staple food availability (1N) and multi-cropping (3E; a *Quality* indicator). We interpreted these tradeoffs as illustrations of farmers' coping capacity when facing environmental degradation and loss of land. Farmers were likely increasing their use of multi-cropping and relying on markets to purchase food in the near-term, enabling

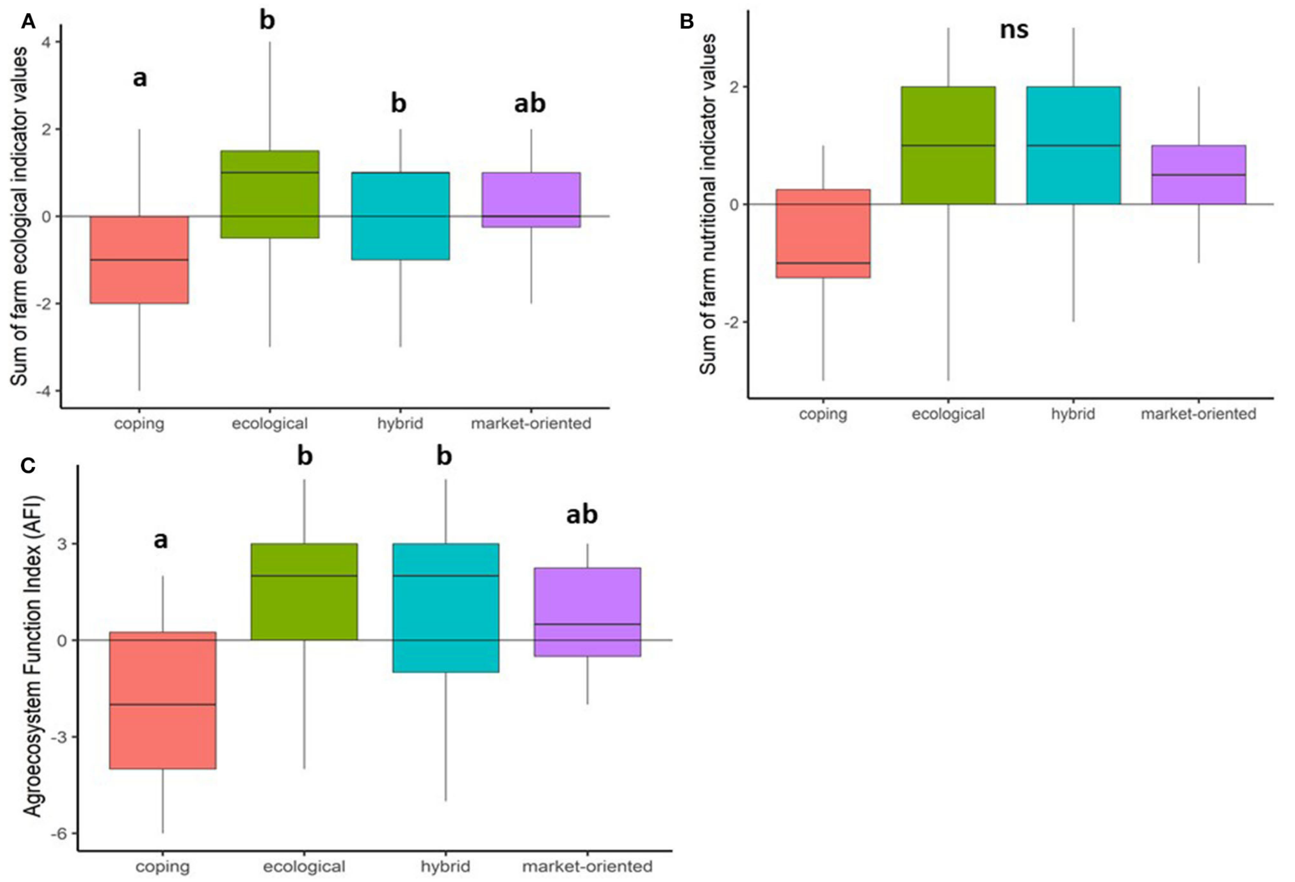

FIGURE 4 | Coping and adaptation strategies in smallholder agroecosystems in Sarstun, Guatemala. Farmer strategies to cope or adapt to shocks are shown in relation to the ecological **(A)**, nutritional **(B)**, and overall sums of indicators represented by the Agroecosystem Function Index (AFI) **(C)**. Primary adaptation strategies were ecological, market-oriented, or a hybrid approach using both ecological and market-oriented practices. Coping strategies included renting land, off-farm work on plantations, and increasing pesticide use. Ecological strategies included incorporating agroforestry techniques, use of leguminous cover crops, and integrating multiple crop diversification practices. Market-oriented strategies included increasing crop sales, growing hybrid maize varieties, and higher fertilizer and pesticide application rates. Different letters indicate significant differences by household coping or adaptation strategy ($p < 0.05$; Tukey's HSD). Village was included as a random effect in all models.

positive nutritional indicators even as ecological indicators were negative.

These tradeoffs may have resulted from temporal disparities, with nutritional indicators responding to change on short timescales and ecological indicators operating over longer timescales. For instance, renting agricultural land is a common practice in Sarstún (32% of our sample), particularly on farms that have lost land through the community-level shock of land grabbing. Renting enables farmers to access relatively high-yielding land on a seasonal basis to produce a staple crop for family consumption, rather than working to improve degraded land with agroecological management for long-term food production. This practice decouples ecological and nutritional functions beyond a single growing season. As has been observed in other contexts, farmers in the sample openly commented that they chose not to use any fertilization methods, such as cover cropping, on rented lands, even if they actively incorporated them into fertility management on communally or privately-owned land in their possession (e.g., Fraser, 2004). Renting has the potential to underwrite extensive degradation of natural resources at the landscape level. Thus, even with rented land, Sarstún smallholders our sample still had mean maize yields that were five times lower than the Guatemalan national average.

Using ecological and nutritional indicators for resilience to community-level and idiosyncratic shocks, we identified the most adaptive and most vulnerable households across the sample. We found that farmers using ecological or hybrid (ecological and market-oriented) adaptation strategies had significantly higher levels of agroecosystem functioning (AFI) than farmers who were coping with losses of land by working on or renting from plantations (**Figure 4**, **Table 5**). Farmers relying on coping capacities such as renting or off-farm work were both ecologically and nutritionally more vulnerable than farmers using adaptive management practices such as cover cropping, agroforestry, or increased production of cash crops in a hybrid approach (**Figure 4**). Purely market-oriented farmers who did not integrate ecological practices had similarly low levels of agroecosystem functioning as coping farmers. Results from the interviews showcased farmers' capacity to adapt to shocks by using ecological and market-oriented strategies to promote agroecosystem functioning and resilience in the face of landscape degradation, land grabs, and climate change. While we found no evidence of transformative capacity in the Guatemalan case study (Béné et al., 2012), the indicator framework and index developed here could also be used to identify cases in which management practices are transformative.

Agroecosystem Functioning, Adaptive Capacity, and Resilience in Other Contexts

Previous work has identified similar synergies and tradeoffs between ecological and nutritional functions of smallholder agroecosystems, with marked impacts for adaptive capacity and resilience. One study in Northern Potosi, Bolivia, found that cropping system intensification on sloping mountain rangelands increased soil erosion and reduced soil organic matter, ultimately undermining productivity, food security, and farmer livelihoods (Vanek and Drinkwater, 2013). However, adapting management to apply phosphorus fertilization in concert with mixed legume-grass cover crops increased soil cover, biological nitrogen fixation, and nutrient availability and assimilation, with feedbacks that reduced erosion and increased crop productivity. Upon surpassing a soil fertility threshold through agroecological management, ecological and nutritional tradeoffs can become synergies that result in long-term positive states (also see Bennett et al., 2009, for related discussion using an ecosystem services framework).

The indicator framework presented in this paper could be applied in a wide range of contexts through the use of case-specific metrics for each indicator. The metrics in our case study were selected based on themes that farmers and key stakeholders identified in interviews, including agroecosystem functions affected by landscape degradation, loss of land tenure, and related livelihood changes. We defined ranges of values and thresholds to either match an established mean value and range (e.g., maize protein concentration from the FAO) or a relative range based on the distribution of values in the case study (e.g., nutritional functional diversity). Because the framework is designed to understand agroecosystem functioning and resilience to shocks in a particular context, this relative valuation approach is appropriate, as it enables comparison across farms and identification of the most vulnerable farms for targeted interventions.

Metrics for agroecosystems in less data-scarce regions could be developed to more closely represent the mechanistic links between specific ecological processes and their nutritional outcomes. For example, other possible pairs of metrics could include crop rotation complexity (2E) and diet diversity (measured for example, with Minimum Dietary Diversity for Women from the FAO) (FAO and FHI 360, 2016) (2N) for *Diversity*, which we were unable to capture in the Guatemalan case study. Community-level metrics relating farm management to broader ecological and nutritional outcomes could also be useful additions to the framework. Indicators may benefit from re-assignment or broader groupings depending on the context in which the framework is applied, as well as the levels of expected interaction between the specific variables selected for the indicators.

Extending and Scaling the Framework: Structural Enablers and Constraints

Social factors that shape farmer capacities for resilience, such as knowledge and skills, participation in social networks, and cultural and institutional influences, affect and interact with ecological and nutritional indicators at both agroecosystem and food system scales (**Figure 1**). Indirect relationships between this broader social context and indicators of agroecosystem function are not currently accounted for in our framework. Therefore, a first extension of this study could be to adopt a food systems resilience perspective and include socio-cultural determinants of adaptive capacity, human health, and food security and nutrition in the framework (Schipanski et al., 2016). Indicators could be sourced from existing frameworks (e.g., Cabell and Oelofse, 2012), which include complementary social indicators related to reflection and learning, and community-based, grassroots organization, among others. Integrating ecological and nutritional indicators with key sociocultural influences would be a logical next step to improve the indicators' ability to accurately represent the resilience of rural livelihoods (Chowdhury and Turner, 2006; Laney and Turner, 2015; Sterling et al., 2017).

Importantly, both environmental conditions and institutional structures can shape and constrain farmer capacities, resulting decision-making, and management systems, even when they lie outside of the agroecosystem's spatial boundary (Hendrickson and James, 2005; Currie, 2011; Brown, 2014). This is especially true in a Global South context, in which power imbalances and landscape-scale environmental degradation frequently go hand-in-hand (DeClerck et al., 2011). We found evidence in our case study, for example, that acute losses of land due to land grabbing at the community level were associated with significantly lower indicators of agroecosystem functioning relative to longer-term landscape-level shocks, including deforestation. At a higher level of social organization, dynamics of the agricultural governance system, particularly power-holding institutions such as governmental agricultural agencies, extension services, seed and chemical companies, and non-profit organizations shape and constrain the options available to small-scale producers (Stuart, 2009). Such organizations, as well as cultural norms, local knowledge and practices, and community expectations, influence smallholder resilience capacities (Scherr, 2000; Guerra et al., 2017). Because these power structures can determine land use and agricultural management practices, they impact ecological and nutritional functions, their interactions, and ultimately the resilience of agroecosystems.

Smallholders may also shift their ecological and nutritional outcomes by engaging with social networks and adaptive capacity at a community scale. Recent research has highlighted that community-level and regional crop diversity can often lead to stronger improvements in diet diversity and nutritional security than diversity at an agroecosystem level (Remans et al., 2015; Tobin et al., 2019). Complementing production diversity, access to markets (Jones, 2016; Koppmair et al., 2016) and the diversity of foods purchased at markets (Bellon et al., 2016) are key contributors to diet diversity and nutrition at a household level. Market-orientation, one form of adaptive capacity explored in our case study, can provide an additional pathway to diet quality through income generation for food purchases (Sibhatu et al., 2015a; Sibhatu and Qaim, 2018), although prior studies have noted that high-calorie, high shelf-life purchased foods that contribute to diet diversity may be supplanting more nutritious traditional foods even in rural contexts (Oyarzun et al., 2013). In remote settings (Koppmair et al., 2016) or the off-season for cash crop production (Some and Jones, 2018), however, the diversity of crops available on the farm gains relative importance. While our indicator framework does not explicitly include community-scale measures, we have demonstrated through our test case that it has potential to identify the effects of community-level shocks on households. By analyzing farmer market-orientation as a form of adaptive capacity, we were also able to examine the role of markets on agroecosystem functions and resilience, even the case study's remote context. As our findings demonstrate, however, market strategies do not necessarily increase smallholder resilience. Our findings aligned with prior work showing that market-orientation as an adaptation strategy shows potential to contribute to smallholder resilience but exposes smallholders to new risks that must be managed (Kuhl, 2018); we found that without a hybrid approach including ecological management strategies in addition to market-oriented strategies, resilient outcomes were not realized on farms. Given the growing literature on agrobiodiversity, diet diversity, and nutrition at larger scales, future work could extend our results by using the framework to study ecological and nutritional indicators at the community or regional scale. Similarly, a fifth indicator representing the social determinants of food security and nutrition, such market access or diversity, could be added to increase the framework's robustness and applicability in less remote regions.

Relatedly, including multiple sources and types of data (e.g., spatial, biophysical, and survey-based) could also improve the predictive ability of the indicator framework (e.g., Geoghegan et al., 2001). Due to Sarstún's remoteness, there is little up-to-date agricultural and demographic data available, and so our case study relied primarily on observational data. Lack of data is a common issue in research on smallholder agriculture, and our framework offers a tool to analyze agroecosystem functioning and its relationship to adaptive capacity and resilience in data-scarce contexts. By scaling indicator values to the maximum in a sample of smallholders, our relative approach to indicator quantification enables researchers and practitioners to identify the most adaptive and most vulnerable households. This approach could be used to target development resources to the households most in need following shocks that can precipitate both ecological degradation and food insecurity, such as land grabs (Alonso-Fradejas, 2012; D'Odorico et al., 2017). Similarly, the framework could also allow the identification of positive farm management strategies worth scaling up. Future frameworks developed for locations where fine-scale data is more freely available would benefit from empirical tests to better understand and incorporate the role of institutional and landscape-level factors on agroecosystem-level social, ecological, and nutritional processes. Interactions between landscape context (e.g., Smith et al., 2020), governance, and farm management decisions affect the livelihoods and resilience of millions of smallholders.

Feedbacks, Transitions, and Transformation in Agroecosystems

The body of work on adaptation and resilience emphasizes the capacity of social-ecological systems to not just maintain stability in the face of shocks but also to adapt or transform—defined as a shift to novel system states or components—as their context changes (Walker et al., 2004; Cote and Nightingale, 2012). In smallholder agroecosystems, social-ecological resilience offers a framework to critically examine not only ecological and nutritional functions, but also their interactions and feedbacks over time (Darnhofer et al., 2010; Béné et al., 2016). Feedbacks can be adaptive or maladaptive. These feedbacks contribute to system functions and act as drivers of agricultural transformations, affecting ecosystem stability and human health.

Our analysis, which combined quantitative and qualitative methods, focused on interactions between ecological, and nutritional functions and their relationships to the adaptive capacity of smallholder agroecosystems. In this framing, tradeoffs and negative feedbacks can lead to ecological degradation and human malnourishment over time, whereas synergies and positive feedbacks result in ecological sustainability and human nutrition (**Figure 1**). There was strong evidence of both coping and adaptive capacities among smallholders in our case study. Data suggest that managing for short-term nutritional functions (e.g., by renting land for a single growing season to produce higher yields; coping) over longer-term ecological functions (e.g., through agroecological management of landholdings; ecological adaptation) could result in negative trajectories for both environmental and human well-being over time.

Adaptive or transformative management at the farm-scale may contribute to agroecosystem resilience by reinforcing ecological and nutritional functions, creating adaptive feedbacks that lead to greater system resilience (Jones et al., 2013; Vanek et al., 2016) (**Figure 1**). Alternatively, management may set off a chain reaction of destabilizing ecological and nutritional functions that lead to agroecosystem degradation via maladaptive feedbacks (Scherr, 2000; Birge et al., 2016). Each of these cycles could result in agroecosystem transformation. However, the former adaptive feedback model would work to the advantage of smallholder households through ecosystem regeneration and sustainable diets (Allen et al., 2014), whereas the latter could result in an unsustainable system, and, over time, household or community-scale malnutrition (Bezner-Kerr et al., 2010; Snapp et al., 2010; Schipanski et al., 2016). If agricultural products are sold or traded, these feedbacks could broadcast beyond the level of the agroecosystem to affect communities or the wider region.

In response to landscape degradation, smallholder farmers often adopt coping strategies that allow their households to maintain their nutritional provisioning despite widespread erosion of the natural resource base. However, these same strategies may prevent *deliberate* and positive long-term resilience or transformation of the agroecosystem (O'Brien, 2012; Béné et al., 2016). With reduced ecological functioning at the landscape level, agroecosystem transformation is likely to occur regardless of temporary coping behaviors to bolster household food security (Kates et al., 2012; O'Brien, 2012). In the case study, this could include farmer emigration to Guatemala City, or families transitioning out of agriculture to work in coastal fisheries or the nascent ecotourism industry (e.g., Katz, 2015); alternatively, it could include farm transitions to agroecological management. Notably, the quality of this transformation will look very different depending on whether additional households adopt ecological management strategies—such as those on 53% of farms in our study—that demonstrate adaptive capacity and contribute to resilience in spite of the impaired environmental status of the overall landscape. Observed tradeoffs, such as between the ecological and nutritional indicators representing *Productivity* in the case study, suggest the need for targeted policies or interventions to support longer-term synergies between ecological and nutritional functions of smallholder agroecosystems (Béné et al., 2016).

A temporal extension of our framework could parse out these short- and long-term dynamics of agroecosystem resilience. Future quantitative analyses could discern changes over time, causality, and interactions between indicators using continuous metrics and time series data, uncovering feedbacks and potential pathways to system transformation. Expanding the indicator framework to account for temporal dynamics and transformation could improve its predictive ability and utility for agroecosystem management over longer time-scales or under changing environmental conditions. Additional analyses of ways that farmers' well-being and nutrition, in turn, influences their capacity to engage in adaptive management would also be of interest, particularly related to practices that are labor-intensive or physically demanding.

CONCLUSIONS

With escalating human and environmental pressure on global agricultural landscapes, adaptive capacity is an increasingly essential tool for smallholder farmers to maintain agroecosystem functioning, and through it, their livelihoods. We created a novel indicator framework to demonstrate the importance of linking ecological and nutritional functions of agroecosystems to leverage their synergies. Using a case study of smallholder farms in a remote region of eastern Guatemala, we found that adaptive management practices tended to produce synergistic ecological and nutritional relationships, whereas coping and market-oriented strategies prioritized basic nutritional functions while undermining ecological ones. Practices that leveraged ecological and nutritional synergies to improve agroecosystem functioning demonstrated smallholders' capacity for resilience in degraded environments.

To foster resilient agroecosystems, we must meet the dual goals of bolstering ecological functions while producing sufficient quantities of high-quality food to ensure food security and nutrition for all people. Our framework establishes that these two goals can be synergistic in smallholder agroecosystems and that farmers can adopt management strategies in line with both ecological and nutritional goals. This adaptable indicator framework can help identify best practices that lead to ecological and nutritional synergies in diverse agroecosystems and contexts and could support decision-makers in targeting supportive resources to the most vulnerable households. The ecological and nutritional indicators proposed here enable nuanced analyses of adaptive capacity and resilience in data-scarce agricultural regions. Future work could relate ecological and nutritional indicators at larger spatial and temporal scales to incorporate the community, landscape, and governance conditions that enable farmers to manage agroecosystems for resilience.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Tufts University Social, Behavioral, and Educational Research Institutional Review Board and determined to be of exempt status: IRB study # 1403034. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

AES designed the study, conducted case study fieldwork and lab work, and led data analysis, with support from LK. AES, JB, and LK constructed the indicator framework and envisioned the paper. AES wrote the manuscript, with substantial contributions from JB and LK. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.543914/full#supplementary-material

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MINI REVIEW
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A Mini-Review on Overcoming a Calorie-Centric World of Monolithic Annual Crops

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There is widespread recognition that a narrow crop base has inherent vulnerabilities. Crop diversification is one strategy that can help enhance human health, environmental sustainability and resilience of farming communities—yet lock-in mechanisms have mediated against such diversification. This mini-review considers inadvertent negative impacts on crop diversity of policies that favor a few, highly annual crops. Priorities of agricultural research and government institutions such as Public Distribution Systems promote production of a few determinant cereal species, and do not consider the ecosystem service functions associated with diverse growth types (e.g., long duration, indeterminant, and perennial crops). Genetic improvement of fields crops has prioritized short maturity cycles, calorie production, and inadvertently, this may lead to high consumption of water and nutrients. Such crops are highly productive; however, they "lock-in" dependence on fossil fuels and chemical pest regulation. Further, early duration crops have modest root systems, and are short-statured. This limits generation of co-products, such as fodder, fuel wood, leafy vegetables, and soil amelioration. Research gaps and next steps are proposed to address this challenge, including: (1) investigation of adoption barriers and opportunities in order to foster diverse crop growth types and "bright spots" of agroecosystem diversity, (2) changing metrics for assessing system performance, to consider nutrient-enrichment, multipurpose properties and ecosystem services in agricultural policy, and (3) investment in developing perennial and multipurpose grain crops, and plant-facilitated nutrient accessing mechanisms. Enhanced resilience in agriculture requires greater attention to promotion of crop diversity, including functional diversity and socio-economic innovations.

Keywords: multipurpose, lock-in trap, growth habit, ecosystem services, green revolution, crop diversification

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INTRODUCTION

Modern, intensified agriculture is highly productive of calories, and provides essential services for many economies. At the same time, environmental costs include the loss of 70% of insect biodiversity in one recent study (Sánchez-Bayo and Wyckhuys, 2019), the wide-spread challenge of nutrient loss from agricultural systems (Bowles et al., 2018), and a sizeable role for this sector in greenhouse gas emissions (Robertson et al., 2000). Agricultural sustainability challenges such as these have risen in tandem with what are increasingly simplified forms of production systems in many parts of the world (Steffen et al., 2015; O'Brien et al., 2019). A handful of crops have come to dominate (Ramankutty et al., 2018). This constricted genetic base is vulnerable to epidemics, and has been shown to reduce biocontrol pest regulation (Landis et al., 2008; Hufford et al., 2019). There are a number of socio-economic factors that reinforce this narrow range of crops, in what has become a "lock-in trap," as explored here (Oliver et al., 2018). A lock-in trap is a societal situation that is resistant to change, due to high connectivity among factors that reinforce each other through feedback loops, often to the determinant of many stakeholders and environmental sustainability. Examples in the literature include unsustainable use of fisheries and management of resources on agricultural lands in Australia (Allison and Hobbs, 2004; Laborde et al., 2016). I consider here how investment in subsidized markets, and research, may have inadvertently reinforced large-scale production of highly annual, calorie producing crops to the determinant of diversity in crop types.

This mini-review focuses on an overlooked attribute of this lock-in trap, that the small number of crop species that dominate agricultural landscapes today have similar, highly annual growth traits. That is, a short-statured and determinant growth habit dominates modern crop varieties. This is at the expense of maintaining a diversity of multifunctional traits and indeterminant growth types. Modern varieties of row crop species are overwhelming determinant in terms of growth type. For example, tomato [*Solanum lycopersicum* (*L.*)] has many perennial and indeterminant traits, as seen in wild relatives; yet over the last 80 years varieties for field crop production have been bred for annual traits (Barrios-Masias and Jackson, 2014). Another example is soybean [*Glycine max* (*L.*)], where there is a body of literature documenting through retrospective analysis how plant traits have changed over a century of crop improvement. Plant breeders, from Canada to China, have developed short-statured and high-oil content varieties from multi-use varieties that were grown historically for forage and seed (Bruce et al., 2019). Today modern varieties are highly annual and short in stature (Wang et al., 2016).

It is understandable why plant breeders have entered into the pursuit of maximizing calorie production through developing highly annual traits including high allocation toward reproduction. Yet, there are tradeoffs associated with this headlong pursuit of one plant type, and this includes the neglect of long duration, multipurpose and semi-perennial growth types which have increasingly become marginalized (Snapp et al., 2019b). This is important, because there may well be an increased vulnerability that is an inadvertent consequence of an agricultural food system that relies on a few plant growth types (Lin, 2011). The limited production of co-products such as vegetation that can be used as a forage, and roots for soil enhancement; these are additional inadvertent consequences of privileging annual crop traits. The final section of this paper considers research gaps that could help diversify crop growth types and promote multi-functional agriculture, for enhanced resilience in a rapidly changing world.

LOCK-IN TRAP IN INDIA

India provides an instructive example of a socio-ecological system that inadvertently promotes a few crop types, reviewed here to provide context to the on-going controversary associated with simplified production systems, which persist despite their critics. This has a genesis in the Green Revolution (Pingali, 2012). There have been a wide range of policy and technological interventions in India to improve access to high-calorie foods, from public distribution institutional interventions to crop improvement. These will be explored here. A key government policy that impacts crop production patterns is the India Public Distribution System (PDS), and related subsidies (Chakraborty and Sarmah, 2019). The India PDS is a distribution system that moves wheat and rice grain throughout the country, and subsidies access to this food in poverty stricken areas (Saini and Ahlawat, 2019). This has provided a large and consistent market for wheat-rice and rice-rice systems (two cereals per year). One unintended consequence of the privileging of these high calorie producing crops appears to be the decline in production area associated with numerous alternate crops (**Figure 1**). Causal attribution is not possible here, and many other factors may be important. For example, in India and many countries, production of wheat and rice is highly supported through investment in genetics, agronomic advice and subsidized fertilizer and irrigation.

There are a wide range of policy and institutional innovations associated with the India Public Distribution System. The India PDS redistributes the large volumes of high-calorie grain produced in the Indo-Gangetic plain of Northern India to reach poverty stricken households (Ahluwalia, 1993). The PDS has functioned for over three decades to deliver food, wheat and rice primarily, at highly subsidized prices, to food insecure populations. It has been critiqued as ineffective in achieving that goal and there is a large literature on how to improve the food safety net goals of PDS interventions (Ramaswami and Balakrishnan, 2002; Chakraborty and Sarmah, 2019). At the same time, there has been little attention to the impact of PSD on agricultural production systems and farmer crop choice. One study found that PDS had an impact on the production side of the equation, as this state institution was shown to consistently buy and distribute poor quality grain, relative to wheat grain quality in the private sector (Ramaswami and Balakrishnan, 2002).

Consumer demand also plays a role in the dominance of a few crops. This is in addition to the role of market distortions, as many crops can't compete in terms of being a consumer-favored source of highly inexpensive calories (Davis et al., 2018). That is, many crops that are modest producers of calories have been relegated to "minor" status, such as millet or sorghum. Yet, such crops are nutrient dense, providing important sources of micro-nutrients, and provide other ecosystem services as well, such as conservation of water (deFries et al., 2016). If appreciation of the unique nutritional advantages of a diverse diet were more widespread, this could potentially lead to greater consumer demand. Nutritional education, and appreciation of traditional diets, have been proposed as means to promote minor crops, and

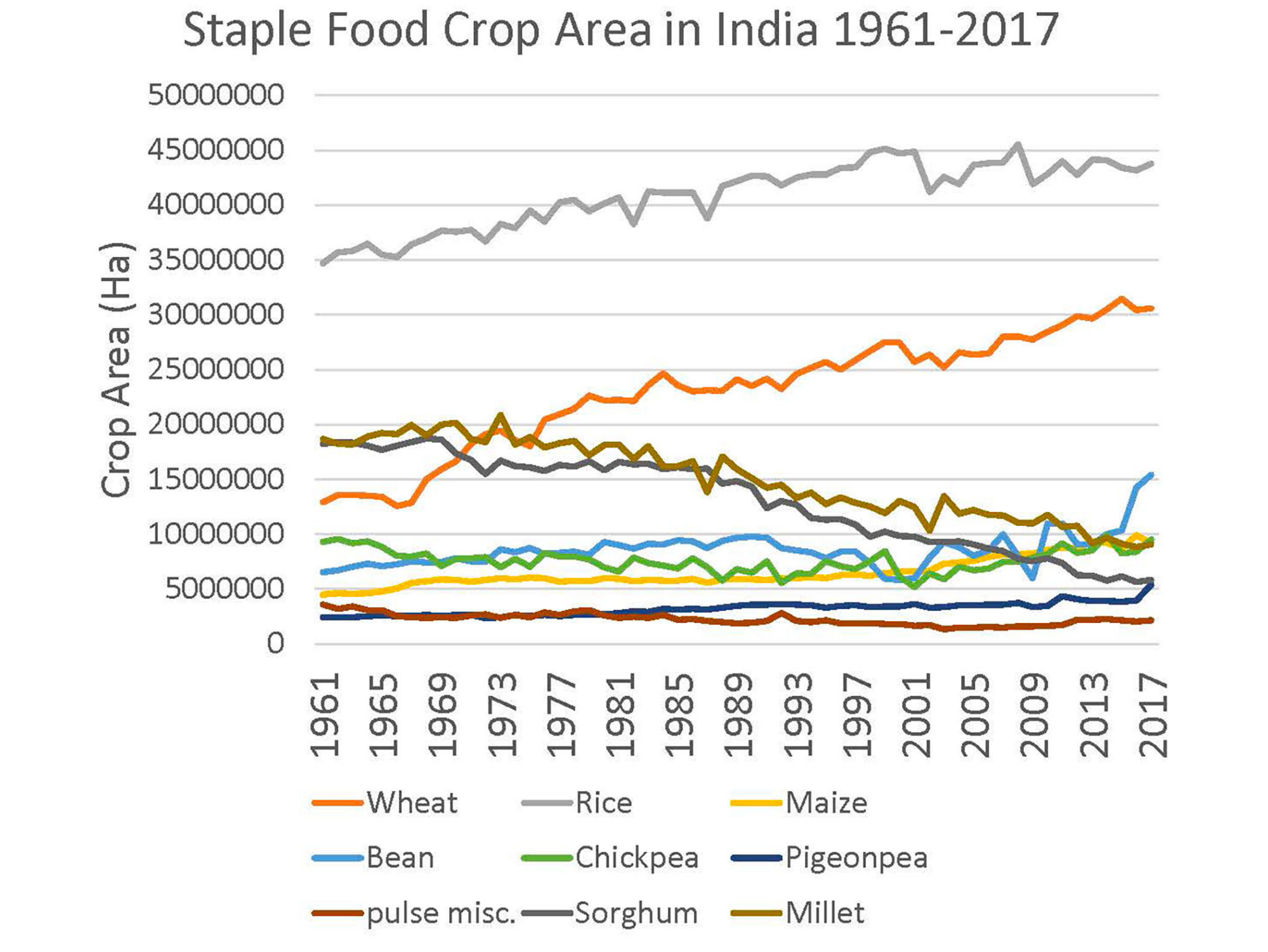

FIGURE 1 | India production area associated with nine major food crop categories (FAOSTAT, 2020).

indeed has led to resurgence in demand for sorghum in some urban markets (Minnaar et al., 2017; Rao et al., 2017).

The privileging calorie production through a focus on rice and wheat has commendable food security intentions. There is broad consensus of an increasing worldwide demand for calorie, which is associated with ecosystem disservices such as carbon loss (Johnson et al., 2014); however, there are many inefficiencies of current food systems that raise the apparent demand for calories, including that cereals are widely fed to livestock (Ritchie et al., 2018). Further, the calorie production aspect of food security has recently been moderated through attention to the "hidden hunger" associated with requirements for protein and micronutrients (Gödecke et al., 2018).

The net effect of promoting high-calorie producing crops appears to have been the prevention of other crops being sown. As shown in **Figure 1**, agricultural statistics reflect a steady expansion in wheat and rice production areas across India over the last six decades. At the same time, the area sown in cereals such as millet and sorghum have decreased dramatically. It is instructive to consider the production area of maize, which was once a minor crop in India. This crop is a champion at translating nitrogenous fertilizers into calorie-rich grain, if given sufficient nutrients and water (Sinclair and Horie, 1989). Maize production has grown markedly in recent decades (**Figure 1**). This is suggestive that it is not just wheat and rice that are favored by policies in India, there may be an overall privileging of crops with the highly-annual growth type that goes with the ability to translate fertilizer into grain, for high calorie production. This has an environmental cost as production of maize is often associated with high use of fertilizers, irrigation, and pesticides (Maggi et al., 2019).

Another case in point is that of pulse production, which has generally stagnated in India. In many markets the price of pulses has climbed, leading to reduced availability and consumption declines among poorer households in India (Rajuladevi, 2001). Pulses have historically been important source of protein. Recent analyses have highlighted that legume crops can play a key role in environmental security as well as promoting human health (Foyer et al., 2016). Yet pulses have often been under-invested in, including limited agricultural research dollars relative to cereals (Pachico, 2014). Pulses such as common bean remain with moderate yield potential and poor grain quality traits such as cooking times and digestibility (Cichy et al., 2015), and a recent review found that investment in genetic improvement of pulses has been almost nil in African agricultural development (Snapp et al., 2019a).

One way out of this lock-in trap in India has been proposed in the literature. It involves an update to the metrics for agricultural system performance (deFries et al., 2015). That is explored by Davis et al. (2018), regarding the impact of a change in agricultural policy to consider nutrition and other ecosystem services, in addition to focusing on calories. Sorghum and millet diversification of Indian rice-wheat systems is explored by these studies, which are suggestive that calories could be maintained, while substantial conservation of water achieved. All of this could be combined with enhanced production of micronutrients if diversified cropping system patterns were supported by the Indian government. These are modeling based estimates, yet they take into account that crop nutritional quality varies greatly with species, and that valuing zinc, iron, calcium, and protein would in turn support high crop diversity. Crop species vary in their impact on natural resources, thus diversification could lessen the “fossil” water withdraws that threaten the sustainability of agriculture in India today (Davis et al., 2018). Environmental services may be related in large part to the genetic variation associated with minor crops, which often include a tremendous diversity of growth types, such as early maturation, as well as long duration (Bezançon et al., 2009). Sorghum genotypes for example include land races which can be grown for 2 or 3 years, through cutting back the stems after the initial harvest, so as to produce deep root systems, soil conservation, and multiple harvests of vegetation for livestock feed, and construction purposes (Rogé et al., 2016).

Lock-in Trap of Simplification in Crop Growth Types

The narrowing of crop species diversity has been accompanied by a reduction in the diversity of functional plant types. Consider for example the grain crops cowpea, pigeonpea, rice, sorghum, and soybean, all of which at one time had tremendous variety in growth type and stature. Cultivars of these species historically included long duration, indeterminant growth habits [e.g., tall-statured types among cereals, and viney, prostrate types among legumes (Rogé et al., 2016; Snapp et al., 2019b)]. Semi-perennial management of sorghum and rice is still occasionally performed, as these crops can be grown as ratoons (Larkin et al., 2014). Yet, crop improvement efforts overwhelming focus on maximizing yield, and selection for plant traits such as determinant, annual growth habit, and a high harvest index (Sinclair, 2019). Harvest index refers to the ratio of commodity biomass (usually grain) that is produced per total plant biomass. A synchronous, determinant growth habit is also favored, which is compatible with mechanized harvest. In sum, traits prioritized in improved, modern varieties of food crops include rapid, early shoot growth, a modest root system, and annualized, highly determinant reproduction.

Grass species are suited to producing large amounts of grain, through early, rapid growth and a starchy endosperm. This has led to cereal production being prioritized by many agricultural policy and crop improvement efforts. Other species produce seed with nutritionally high-quality biochemical traits, which can be metabolically expensive to produce (Tian and Bekkering, 2019). This constrains the yield potential of such crops, as is notable for legume crops which in addition support a symbiotic association with bacteria, at some metabolic expense. A study in Europe found evidence for about 30% higher biomass production in systems that had minimal vs high legume presence (Iannetta et al., 2016). At the same time, nitrogen fertilizer requirements were high for the low legume systems. There may be additional tradeoffs associated with prioritizing plant traits of rapid growth and early maturation, as these can constrain tissue quality due the limited time for uptake and integration of nutrients (**Figure 2**). Tradeoff expression, however, is expected to vary with crop species, and plant breeding programs. A historical analysis of soybean varieties, for example, found that nitrogen concentration of the seed has remained stable whereas phosphorus concentration has declined (Balboa et al., 2018). The yield potential enhancement of modern maize varieties, on the other hand, has been associated with clear declines in nitrogen concentration (Scott et al., 2006).

Consequences of the Dominance of Rapid Maturation Crops

The emphasis on short-stature, and determinant growth types, continues today. This is indicated by priorities of plant breeders in Africa and South Asia (Snapp et al., 2019b). Modern varieties of pigeopea and sorghum for example have been bred to be as much as a meter shorter than many land races. An annual-centric focus in crop improvement is not consistent with the environmental and market context faced by many small-scale farmers. Indeed, smallholder farms are rarely chemical-intensive and often rely on indeterminacy to tolerate pests, as multiple flowering periods can reduce the negative impact of pest predation (Dube and Fanadzo, 2013). There is need for diversity in crop types. For example, many modern varieties provide traits such as early maturation and high grain yield. Yet, at the same time, smallholder farmers also require tall-statured varieties to escape to some extent grazing by free roaming livestock, and to produce in addition to grain, stems and stalks for fuel, and construction purposes (Orr et al., 2015; Rogé et al., 2017).

Consideration of the impact of a variety grown on the entire farming system is important in the context of smallholder farm livelihoods (McDonald et al., 2019). That is, the impact of a crop species on soil resources, and the stability of associated crops that are frequently grown in mixed production systems. For example, semi-perennial crops in rotation with cereals has been shown to increase the stability of crop production (Snapp et al., 2010; Chimonyo et al., 2019). Further, long duration, tall-statured and indeterminant types of crops produce copious amounts of vegetation which can be used for forage,

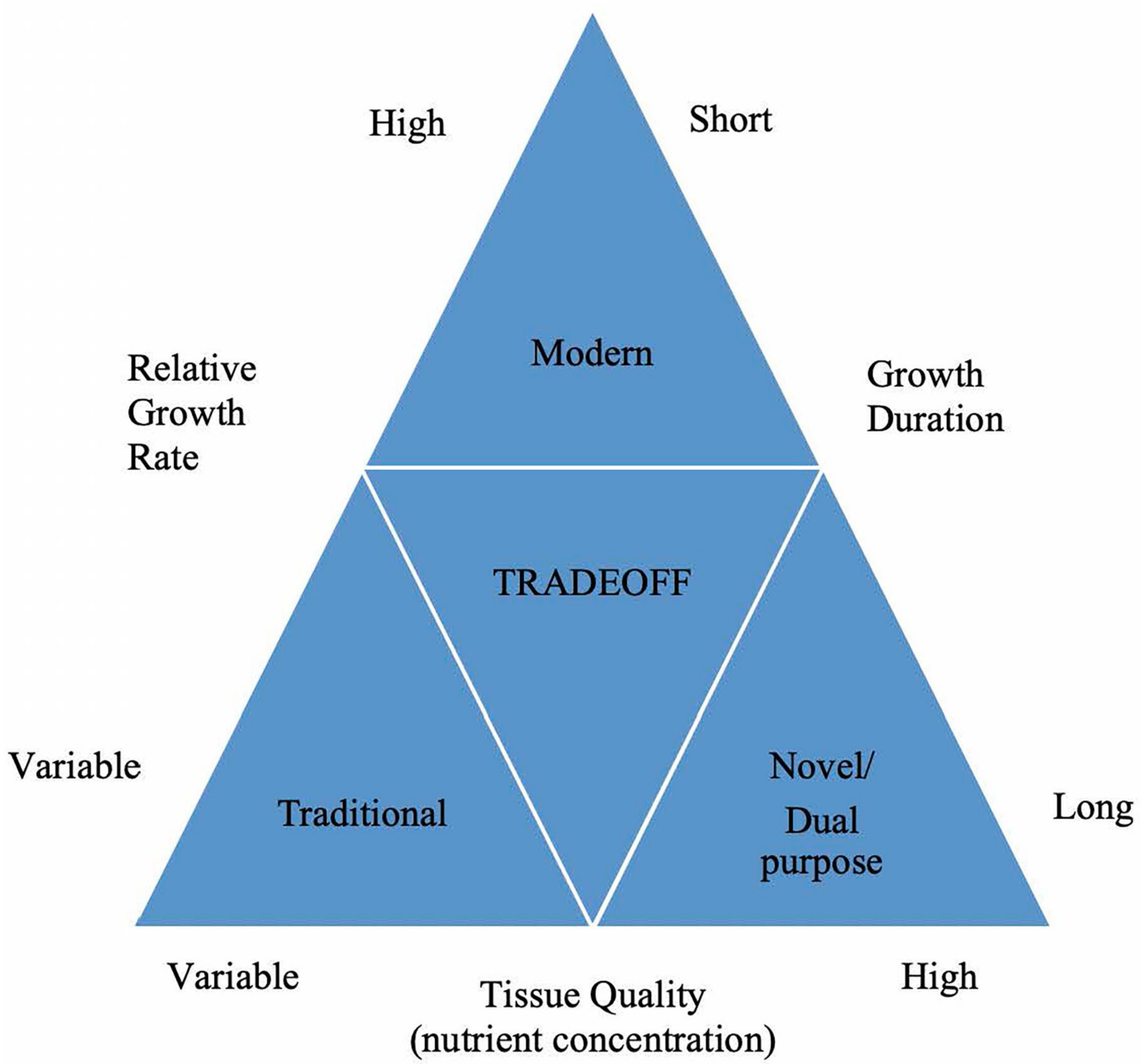

FIGURE 2 | A conceptual diagram of tradeoffs associated with plant traits associated with the rate and duration of growth, and tissue quality. Modern, improved cultivars of food crops are generally associated with relatively rapid growth rates and short duration, to achieve early vigor, and a high crop yield. Traditional varieties vary markedly in relationship to these traits, and novel plant breeding efforts have emphasized long duration of growth for dual purpose use, or high nutrient concentration for biofortification.

or soil building purposes, and are valued co-products on many farms (Singh et al., 2003). Farmer-preferred varieties of pigeonpea in Malawi, for example, are associated with production of livestock feed, fuel wood, and soil fertility, as well as grain (Orr et al., 2015; Grabowski et al., 2019). Yet, modern varieties of pigeonpea recently released in Southern and East Africa emphasize short duration and short-stature, with few co-products (Snapp et al., 2019b).

Long duration, and semi-perennial properties have been associated with many traditional varieties of food crops, which often include a diversity of plant growth habits. There is little consideration in the literature of the opportunity costs of prioritizing instead a narrow range of crop growth types in modern crop improvement, that of rapid growth and early maturity. There may be steep tradeoffs with nutrient concentration, as illustrated in **Figure 2**. Farmer interest in diversity of crop growth types, and functions, is indicated by the persistence around the globe of traditional varieties, and agronomic practices such as polycultures and ratooning (which support multiple harvests from the same plant, and high biomass production). Perennial, indeterminant traits support multiple flushes of flowers and vegetation that can play a key role in resilience to stress, and are associated with deep root systems (Kell, 2012). Root system carbon inputs are an important soil carbon sequestration mechanism. Indeed, perennial and long duration crop traits have been highlighted as the chief means to support soil biological processes, and derived ecosystem services including crop health, nutrient provisioning and water regulation (Rasche et al., 2017).

DISCUSSION

Evidence is growing of the inadvertent consequences of policy, agronomy and modern genetics that privilege short-duration field crops. A recent study in the Great Plains of the USA illustrates this (O'Brien et al., 2019). In tandem with simplification of crops grown to primarily corn and soybean, and associated loss of winter cereals that provided semi-perennial cover, the authors observed marked declines in peak river flows, and substantial increases in chemical inputs for crop production. Further studies of landscape consequences are needed, but it is reasonable to infer that crops that grow fast and have a high harvest index will not have been selected for simultaneous investment in root systems and biochemical properties that repel pests—thus might be expected to require large amounts of nutrients and water, and may be pest susceptible. Maize is clearly a case in point, being globally associated with nitrogenous fertilizer and pesticide use (Maggi et al., 2019).

To bring back ecosystem services in agricultural landscapes there are many efforts to diversify annual field crops through use of cover crops (Snapp et al., 2005). There are also strategies being investigated add diversity to arable fields through agroforestry and plantings of prairie strips (Lin, 2011; Leakey, 2017; Schulte et al., 2017). These are important research areas. However, such approaches focus on diversification in the off-season, or at the margins of fields. The primary land use remains in simplified rotations and monocultures of annual field crops, which constrains opportunities for diversification and for remediation of environmental problems.

A radical approach to expanding the range of growth types cultivated for food has been proposed, that of developing perennial grain crops (Glover et al., 2010; Kane et al., 2016). This strategy remains overall a theoretical concept, in that there are very few examples of perennial grain varieties being released or adopted, and studies have raised concerns about the economics within the current agricultural policy context (Bell et al., 2008; Snapp et al., 2019b). Yet a perennial growth type could potentially deliver a broad range of environmental services and should not be overlooked (Larkin et al., 2014; Sprunger et al., 2019). There are a number of traditional African farming systems that rely to varying degrees on semi-perennial grain crops such as sorghum and pigeonpea that can be ratooned (Rogé et al., 2016). Insufficient calorie production potential has been raised as a critique of perennial grain crops. Yet, there is evidence that photosynthetic capacity can be upregulated, and that this is associated with perennial traits, thus there may be unexplored genetic potential to expand calories produced in perennial crop types (Jaikumar et al., 2013). Maximizing grain yield is not the only goal in agricultural production, as stability of yield and environmental long-term resource conservation are also widely valued, and thus there is growing evidence that mixture of growth types would have value among modern crop varieties, including perennial and semi-perennial traits (Snapp et al., 2019b).

Ways Forward

Three research gaps are identified here, as initial steps to address the lock-in trap of monocultural production. It will also be important to support an enabling environment for crop diversity, which may involve scholars engaged in activism, and private-public partnerships (Jordan et al., 2020). An inspirational example is the Green Lands Blue Waters network in the US Midwest that promotes continuous living cover for a more diverse, sustainable agriculture (https://greenlandsbluewaters.org/). Prairie strip research partnerships and collaborations with farmers is another such example (Atwell et al., 2010). As is legume-based farming, promoted in Europe through participatory networks and attention to the agricultural policy framework (Mawois et al., 2019).

A major research gap is the lack of an evidence-base that documents barriers and opportunities associated with growing diverse crops, and how these operate at different scales, from farm, community, region to national. There are marked declines in production of legume crops, suggestive of steep barriers to production. For example, several recent studies have reviewed the multifunctional roles that legume crops play in Europe, and the societal price paid in the form of fertilizer dependency and associated environmental disservices (Iannetta et al., 2016; Mawois et al., 2019). In an agricultural market context where stable, high grain yields and certainty in access to markets are all important to economic viability of farm enterprises, legume production is often perceived as not meeting any of these goals (Mawois et al., 2019).

There has been limited studies of adoption among species that are categorized as alternative or minor crops. For example, international agricultural centers have invested in a handful of studies on legume variety adoption, in contrast to hundreds on cereal variety adoption (Snapp et al., 2019a). One methodological approach would be to study the "bright spots" where crop diversity has flourished, as well as "dark spots" where diversity is highly limited (Frei et al., 2018).

Research is needed on how to incentivize diversity in agriculture, through performance criteria. Change in the criteria for assessing performance of cropping systems could markedly alter how genetic and agronomic success is judged, and how agricultural policies are framed. The example presented earlier for India is a case in point, diversification of wheat and rice with millet and sorghum crops would be promoted if the political metrics included water conservation or nutrient enriched grain (Davis et al., 2018). The global case for inclusion of legume crops has also been based largely on metrics that consider human health and environmental conservation (Foyer et al., 2016). Valuation of ecosystem services, and consideration of sustainable development goals are major subjects of major research inquiry (Wood et al., 2018), yet there are gaps in terms of specifics and attention to how these could be operationalized to inform agricultual development and policies. Sustainability intensification assessment is one approach that provides a practical example of how to visualize tradeoffs among indicators and assess overall performance and potential contributions of agricultural technologies (Smith et al., 2017).

Research gaps include projects that will require a long-term perspective and a massive investment of resources. This is the nature of efforts to breed crops with new plant traits including perennial growth habits to support ecosystem services (Glover et al., 2010). Diversity of growth types would be maintained if crop breeding efforts also included intermediate multipurpose types [e.g., semi-perennial shrubs and vines (Snapp et al., 2019b)]. Enhanced ratoon ability could for example provide copious amounts of fodder and grain under drought stress, yet this has rarely been researched as part of genetic improvement or agronomic programs. A related area of research is that of developing crops that facilitate associated microbial symbioses that enhance availability of sparingly soluble phosphorus through biologically-enhanced mobilization, that fix substantial amounts of nitrogen, and that promote soil carbon accrual. There are well-documented examples of germplasm that supports all three processes, including in pigeonpea (*Cajanus cajan*) and lupin (*Lupinus albus*) (Garland et al., 2018).

In conclusion, crop diversity documentation requires a major effort as an evidence base for all three research gaps. This to characterize phenotypes and utilization in traditional agriculture, to help move forward efforts on multiple fronts. In addition, genetic throughput mechanisms need to be developed for the variety of crop traits discussed here. This requires attention to farmer-valued traits and how to characterize them through participatory breeding, as well as traditional phenotyping. Addressing these and related research gaps should not be overlooked in the search to enhance agricultural ecosystem services and sustainability.

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SS conceptualized, conducted, and wrote the review.

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Confronting Barriers to Cropping System Diversification

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There is no shortage of data demonstrating that diversified cropping systems can sustain high levels of productivity with fewer external inputs and lower externalities compared to more simplified systems. Similarly, data exist indicating diverse cropping systems have greater capacity to buffer against and adapt to weather extremes associated with climate change. Yet, agriculture in the US Corn Belt and other major crop production regions around the world continues to move toward simplified rotations grown over increasingly large acreages. If our goal is to see more of the agricultural landscape made up of diverse agricultural systems and the ecosystem services they provide, it is critical we understand and creatively address the factors that both give rise to monocultures and reinforce their entrenchment at the exclusion of more diversified alternatives. Using the current state of farming and agriculture policy in the US as a case study, we argue that a pernicious feedback exists in which economic and policy forces incentivize low diversity cropping systems which then become entrenched due, in part, to a lack of research and policy aimed at enabling farming practices that support the diversification of cropping systems at larger spatial scales. We use the recent example of dicamba-resistant crops to illustrate the nature of this pernicious feedback and offer suggestions for creating "virtuous feedbacks" aimed at achieving a more diversified agriculture.

Keywords: agrichemical industry, cover crops, crop rotation, ecosystem services, herbicide resistance, policy, glyphosate

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INTRODUCTION

Farmers have known for millennia that planting the same crop in the same field year after year quickly leads to impoverished soil and unmanageable populations of disease organisms, weeds, and insect pests (Bullock, 1992; Howieson et al., 2000; Karlen et al., 2006). Similarly, recent studies have demonstrated that increasing the diversity of a simple crop rotation by even a few crops can result in not only similar or greater overall crop productivity and economic returns compared to the conventional rotation, but also improved soil fertility and lower pest populations and lower requirements for fertilizer and pesticide inputs (Smith et al., 2008, 2018; Davis et al., 2012; Weisberger et al., 2019; Archer et al., 2020). What is more, numerous studies have shown that the ecosystem services that arise from diversifying crop rotations, such as soil quality and fertility enhancements, can also help buffer these systems against weather variability associated with climate change (Bommarco et al., 2013; Gaudin et al., 2015; Williams et al., 2016; Bowles et al., 2020). And recent socio-ecological research suggests that some farmers practicing monoculture acknowledge the role that cropping system diversification can play in adapting to climate change (Roesch-McNally et al., 2018).

Given the evidence for improvements in productivity, economic outcomes, climate resilience, and other ecosystem services associated with diversified crop rotations, how is it that farmers continue to maintain vast near monocultures of corn and soybean in the US Midwest or wheat in southern Canada and Western Australia, for example? And why would they seemingly choose to farm this way when the benefits of maintaining diverse crop rotations are so well-understood, even among farmers practicing monoculture production?

The answer to the first question is relatively simple. Farmers can successfully grow crops in monocultures and simple crop rotations because they have access to synthetic fertilizers and pesticides. Indeed, without inputs of synthetic fertilizer and pesticides, continuous crop monocultures could not exist. This reliance on fertilizer and pesticides also explains, in large part, why monoculture farming is so fraught from an environmental pollution perspective (Robertson and Swinton, 2005). Often, the synthetic fertilizer that is necessary to drive monoculture production is not taken up by the crop or stored in soil organic matter, meaning some portion of the fertilizer remains in the soil and is therefore susceptible to loss and movement to other ecosystems (Robertson and Vitousek, 2009). Similarly, the pesticides which are required to control otherwise untenable pest levels rarely remain confined to their intended targets (Kolpin et al., 1998; Humann-Guilleminot et al., 2019).

Of course, this simple proximate explanation for the existence of monocultures does not fully acknowledge the critical role the agrichemical industry plays in supporting monoculture production. Indeed, the easy access to fertilizers and pesticides farmers require in order to maintain simple crop rotations would be impossible without an agrichemical industry eager to supply these inputs. Conversely, the agrichemical industry's business model (true of most business models that require keeping shareholders appeased) depends on the production and sale of as much product as possible, while at the same time defending against threats to its market share (Magdoff et al., 2000; Mascarenhas and Busch, 2006; Hendrickson, 2015; Harker et al., 2017). Hence, it might be just as accurate to say monocultures exist because there is an agrichemical industry that profits from their existence.

Now that we've established how it is possible for farmers to farm in monoculture, the second question is why do they choose to do so given the issues described above? The answer to this question is more complex. Many potentially interacting factors contribute to a farmer's decision to specialize in just one or a few crops. We term these factors "simplification forces" because the net result is often a simplified crop rotation. While much of the initial decision process is under farmer control—meaning that in some sense farmers do indeed *choose* to implement simplified cropping systems—at some point farmers can become locked-in to the simplified system due to factors that act to reinforce the continued existence of the simplified system (e.g., Geels, 2011). These reinforcing factors make it extremely difficult for farmers, once locked-in to a simplified system, to change their practices and/or integrate additional types of crops or cropping practices into their systems. We more fully describe these factors below, with special attention to the reinforcing factors which make cropping system re-diversification such a challenge.

SIMPLIFICATION FORCES

Simplification forces incentivize reducing the diversity of a cropping system. At their foundation, most of these simplification forces have an economic, and hence, political basis (MacDonald et al., 2013). A dearth of competitive markets for alternative crops and a lack of infrastructure for processing and/or product storage limit the types of crops that farmers consider profitable in a given region (Bradshaw et al., 2004; Meynard et al., 2018; Roesch-McNally et al., 2018). Simplification is accelerated when volume discounts are offered for the prevailing commodity crop and associated inputs (Magdoff et al., 2000). In addition, the decoupling of crop and livestock production in many regions has led to farmers specializing in annual row crops, forgoing the perennial pasture crops that were once more common components of their rotations (Howieson et al., 2000; Karlen et al., 2006; Davis et al., 2012; Roesch-McNally et al., 2018). The climate, through its effects on the economics of farming, can also act as a simplification force, simply by limiting the types and scale of crops that can be profitably grown, when they can be planted and harvested, and their yield potentials (Bradshaw et al., 2004). National agriculture policies that have commoditized certain crops or that externalize risks associated with simplified cropping systems provide additional financial incentives, through subsidies and federal crop insurance programs, to grow only those crops (O'Donoghue et al., 2009; Iles and Marsh, 2012; Roesch-McNally et al., 2018). For example, the majority of commodity payments in the US go to just seven crops, while farmers growing certain other types of crops, such as vegetables, nuts, and fruits, often receive little in the way of federal subsidies or other incentives (Iles and Marsh, 2012). Emergence of new policy and subsidiary markets for the products (or byproducts) of the dominant monocrop can also incentivize cropping system simplification, even in systems that are already highly simplified (Karlen et al., 2006). The increase in corn production and concomitant reduction in landscape scale crop diversity that occurred in the US Corn Belt as a consequence of the 2007 Energy Independence and Security Act (i.e., "ethanol mandate"; Rahall, 2007) is a good example of this phenomenon (Landis et al., 2008). Additionally, the fact that externalities associated with monoculture cropping, such as environmental pollution and loss of biocontrol ecosystem services, are typically not borne by the farmer, and therefore not passed on to the consumer, further disincentivizes adoption of more complex cropping systems (Robertson and Swinton, 2005; Landis et al., 2008).

Concomitant with the simplification of cropping systems, many agricultural regions have seen tremendous increases in the size of individual farm operations (MacDonald et al., 2013). In the US for example, while small and medium-sized farms (i.e., farms with 1–404 ha of harvested cropland) make up the overwhelming majority of farms on a number basis, large-scale farms (i.e., those with >405 ha of harvested cropland) account for over 67% of the total harvested cropland (**Figure 1**; USDA NASS, 2019). Many of the same broad simplification forces described above—including disassociation of livestock and crop farming, federal policy decisions, and innovations in labor-saving practices and technology—have, along with economies of size, contributed to the increasing scale and profitability of simple farming systems (Duffy, 2009; Geels, 2011; MacDonald et al., 2013; Hoppe, 2014). At the same time, investments in technology and specialized equipment, including large-scale machinery such as tractors, planters, and sprayers which enable large-acreage farming, can also act to lock farmers into their existing cropping systems.

REINFORCING FACTORS, SOCIO-TECHNOLOGICAL LOCK-IN, AND DEFENSIVE SIMPLIFICATION

As we've alluded to, some of the simplification forces described above may also become reinforcing factors if they also strongly inhibit farmers' abilities to subsequently change their farming practices—a situation known as "lock-in." Lock-in is the inability of a farmer to deviate from or change the existing farming practice or system due to social, political, economic, or technological reasons (Arthur, 1989; Wilson and Tisdell, 2001; Geels, 2011; Magrini et al., 2016; Wigboldus et al., 2016). The large-scale and specialized equipment purchased to facilitate the growing of crops over extremely large farm acreages represents one type of reinforcing factor because it also makes it difficult or impossible to integrate additional crops into a rotation if those crops would require different equipment for planting or harvesting (Geels, 2011). The erosion of specialized knowledge, skills, and experience necessary to grow and manage a diversity of crops, as well as a lack of farm workers able and willing to do the work, can all be significant reinforcing factors (Iles and Marsh, 2012; Carlisle et al., 2019). Finally, reliance on fertilizers and pesticides can also be reinforcing factors leading to lock-in (e.g., pesticide and technology "treadmills"; Wilson and Tisdell, 2001; Mortensen et al., 2012; Wigboldus et al., 2016).

Beyond locking farmers into specific cropping systems, some especially pernicious reinforcing factors can threaten the ability of other farming systems to coexist within the landscape. These factors tend to be byproducts of the enablers of cropping system simplification, specifically the crop protection products that are relied upon to control weeds and insect pests in simplified cropping systems. These types of reinforcing factors are particularly problematic because they have the potential to reduce landscape-scale crop diversity either by threatening the coexistence of certain crops or cropping systems or by forcing farmers to further simplify their cropping systems as a defensive counter-measure. As an illustrative example, the remainder

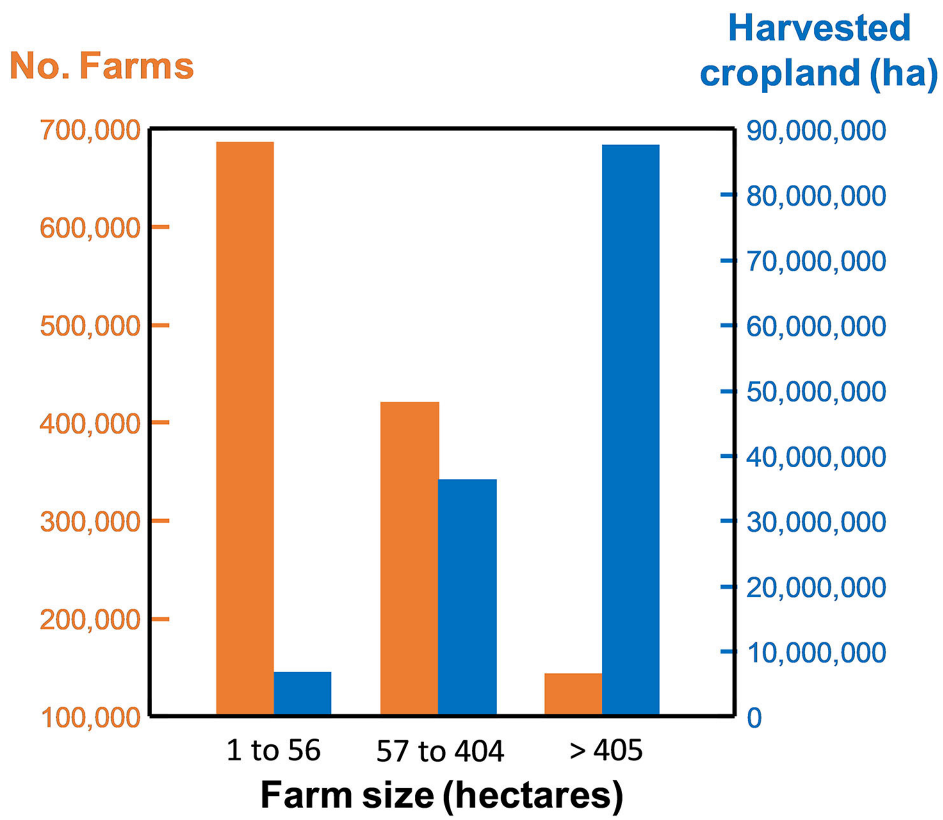

FIGURE 1 | Number of farms with harvested cropland (orange bars) and total harvested cropland (blue bars) by farm size class in the US in 2017. Data are from Table 9 of the 2017 US Census of Agriculture, Census Volume 1, Chapter 1: U.S. National Level Data (USDA NASS, 2019).

of this section describes one such reinforcing factor: the use of herbicides with high potential to cause drift damage and the recently developed genetically-modified herbicide-resistant crops that have greatly expanded their use.

The Case of Dicamba-Resistant Crops

Herbicide-resistant crops, crop plants genetically engineered to be resistant to herbicides that would normally kill them, first came to the US market in the mid-1990s. These crops were engineered to be resistant to the herbicide glyphosate, a highly effective broad-spectrum herbicide that is phytotoxically active on a large number of weed and crop species across a wide range of taxa (Duke and Powles, 2009). Modified to express enzymes that are insensitive to or can metabolize glyphosate, glyphosate resistant (GR) crops allowed farmers to easily apply glyphosate in soybean, corn, cotton, canola, sugarbeet, and alfalfa to control weeds without harming the crop (Behrens et al., 2007). With Monsanto (now Bayer) holding the patent on the resistant genes and who first developed glyphosate herbicide, the resulting seed and pesticide package was heavily marketed and then adopted at an unprecedented rate by growers who were attracted to the flexibility and simplicity of the glyphosate/glyphosate resistant crop technology package (Mascarenhas and Busch, 2006; Mortensen et al., 2012).

This commodity seed and pesticide package became a strong reinforcing factor soon after emerging on the market in 1996, and by 2000, GR soybeans accounted for 54% of US soybean hectares (Duke and Powles, 2009). By 2018, GR crops were grown on 90, 91, and 94% of the US corn, cotton, and soybean hectares, respectively (Wechsler, 2018). The technology is effective and easy-to-use and farmers have often responded to these benefits by exclusively planting GR cultivars and applying glyphosate herbicide in the same fields, year after year (Duke and Powles, 2009; National Research Council, 2010).

This single-tactic approach to weed management practiced on over 90% of the principle commodity crops in the US has resulted in an unintended, but not unexpected, problem: a dramatic rise in the number and extent of weed species resistant to glyphosate (Heap, 2020) and a concomitant decline in the effectiveness of glyphosate as a weed management tool (Duke and Powles, 2009; National Research Council, 2010). As the area planted to GR crops increased, the total amount of glyphosate applied also kept pace, creating intense selection pressure for the evolution of weeds resistant to glyphosate. To be clear, this dramatic increase in glyphosate use would not have been possible without GR crop biotechnology. The number and extent of weed species resistant to glyphosate has increased rapidly since 1996, with 48 species now confirmed globally (Heap, 2020). While several of these species first appeared in cropping systems where glyphosate was being used without a resistant cultivar, the most severe outbreaks have occurred in regions where GR crops have facilitated continued overreliance on this herbicide (Evans et al., 2016).

To address the problem of GR weeds, the seed and agrichemical industries aggressively developed and marketed new genetically engineered cultivars of soybean, cotton and corn, where in addition to glyphosate resistance, additional herbicide resistance traits were added. For reasons that make little sense other than they control a number of the weeds that have evolved resistance, these "next generation" herbicide resistant crops were engineered to also be resistant to older and more environmentally problematic herbicides dicamba (BASF, Monsanto/Bayer) and 2,4-D (Dow AgroSciences) (Behrens et al., 2007; Wright et al., 2010). Unfortunately, while these herbicides can be used to provide some level of control to the weeds that evolved resistance to glyphosate, they are also highly active and strongly phytotoxic to most other broadleaf plants, including crops, and, most problematically, are well-known to be highly drift prone (Egan et al., 2014). Together, these two properties of the new herbicide-resistant crops mean there is a high likelihood of severe damage to broadleaf crop and non-crop plants in nearby fields (Mortensen et al., 2012). In other words, the extent and impact of the non-target effects of using herbicides like dicamba and 2,4-D are large, with equally large potential to result in defensive simplification.

The adverse effects of dicamba use that had been predicted by models and extrapolations from experimental work (Egan et al., 2011; Egan and Mortensen, 2012) have played out in the field. In 2016, the year of their commercial release, dicamba-resistant crops were planted on 25 million acres. Acreage doubled to 50 million acres in the 2018 field season and an estimated 60 million acres were planted to these crops in 2019 (Unglesbee, 2019; Wechsler et al., 2019). As a consequence of the problematic properties of dicamba, in the 2017 field season, 3.6 million acres of non-transformed soybean were injured by dicamba drift (Nandula, 2019). This extent of crop injury in response to this crop and herbicide use practice is unprecedented. It is also a significant under representation of the total plant injury that occurred. Importantly, the 3.6 million acres only accounts for injury to soybean and doesn't include other susceptible broadleaf crops nor does it include non-crop broadleaf flowering plants. The dicamba drift issue has been so bad, in fact, that some farmers have been forced to purchase and plant the dicamba-resistant crops defensively, in order to minimize the potential of dicamba drift from neighboring farms injuring their own crops (Wechsler et al., 2019; Fletcher, 2020).

We attempt to capture the spatial and temporal dynamics of such a cropping system practice on landscape level crop diversity in **Figure 2**. The simple simulation represents an agricultural landscape comprised of 30 individual fields where early in the time course, one field uses the newly transformed crop/herbicide package. The use of the package results in neighboring farmers adopting the practice because retailers shift to limiting seed choices and because package developers provide incentives for "defensive planting," a practice where farmers reduce their risk of crop damage by planting transformed crops that are not susceptible to the non-target effects. Over the course of the 5 year time period, the Shannon diversity index quantifying landscape scale crop diversity falls as does the proportion of the landscape planted to the non-transformed crop.

We contend that the dicamba-resistant cropping system example highlights a central crux of the cropping system diversity problem (**Figure 3**). In this case, private sector interests (i.e., the biotech seed and agrichemical industry) profit from the

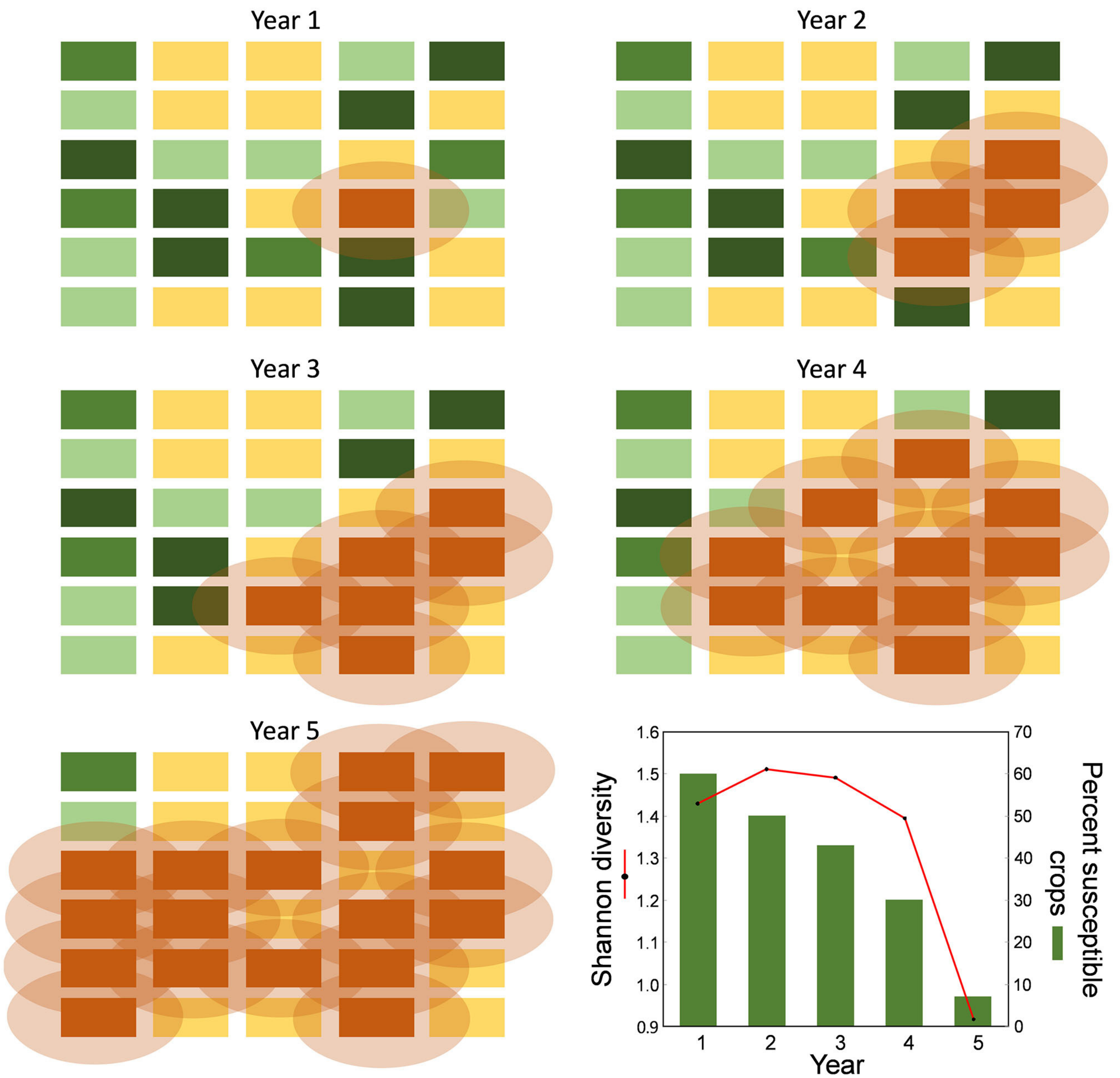

FIGURE 2 | Change in crop diversity across a hypothetical agricultural landscape over 5 years following the introduction of a reinforcing factor that drives defensive simplification of neighboring cropping systems. Each rectangle represents a field or farm. In this example, the orange field/farm in year 1 adopts a new crop or cropping practice associated with off-target effects that manifest on neighboring fields (i.e., dicamba herbicide vapor drift that travels beyond the confines of the field to which it is applied). Here we assume that off-target movement of the herbicide extends some distance from the field on which it is applied, represented by the "drift cloud" (shaded oval), and causes damage to susceptible crops (shades of green) but not resistant (orange) or tolerant (yellow) crops in farms/fields that are located immediately adjacent to the offending farm. As a defensive measure, farms that experience an unacceptable level of damage in a given year, represented by at least some portion of the field covered by the drift cloud, switch to the new cropping practice the following year. Farms/fields adjacent to the new adopters of the practice are subsequently impacted and then adopt defensive planting of resistant cultivars in the following year. The result of this dynamic over time is an overall reduction in crop diversity at the landscape level (bottom right panel).

agrichemical inputs and the associated "enabling technologies" that are required to maintain a specific farming system—i.e., large-scale, simplified crop rotations. The larger the scale, and the simpler and more widespread the farming system becomes, the more the industry's product packages are required, and the more the industry profits (Hendrickson, 2015; Clapp, 2018). And as we've stated above, the larger and more simplified the farming system becomes the more farmers become locked-in to the simplified system (Levins and Cochrane, 1996). At the same time, federal agriculture agencies, such as the USDA, invest in

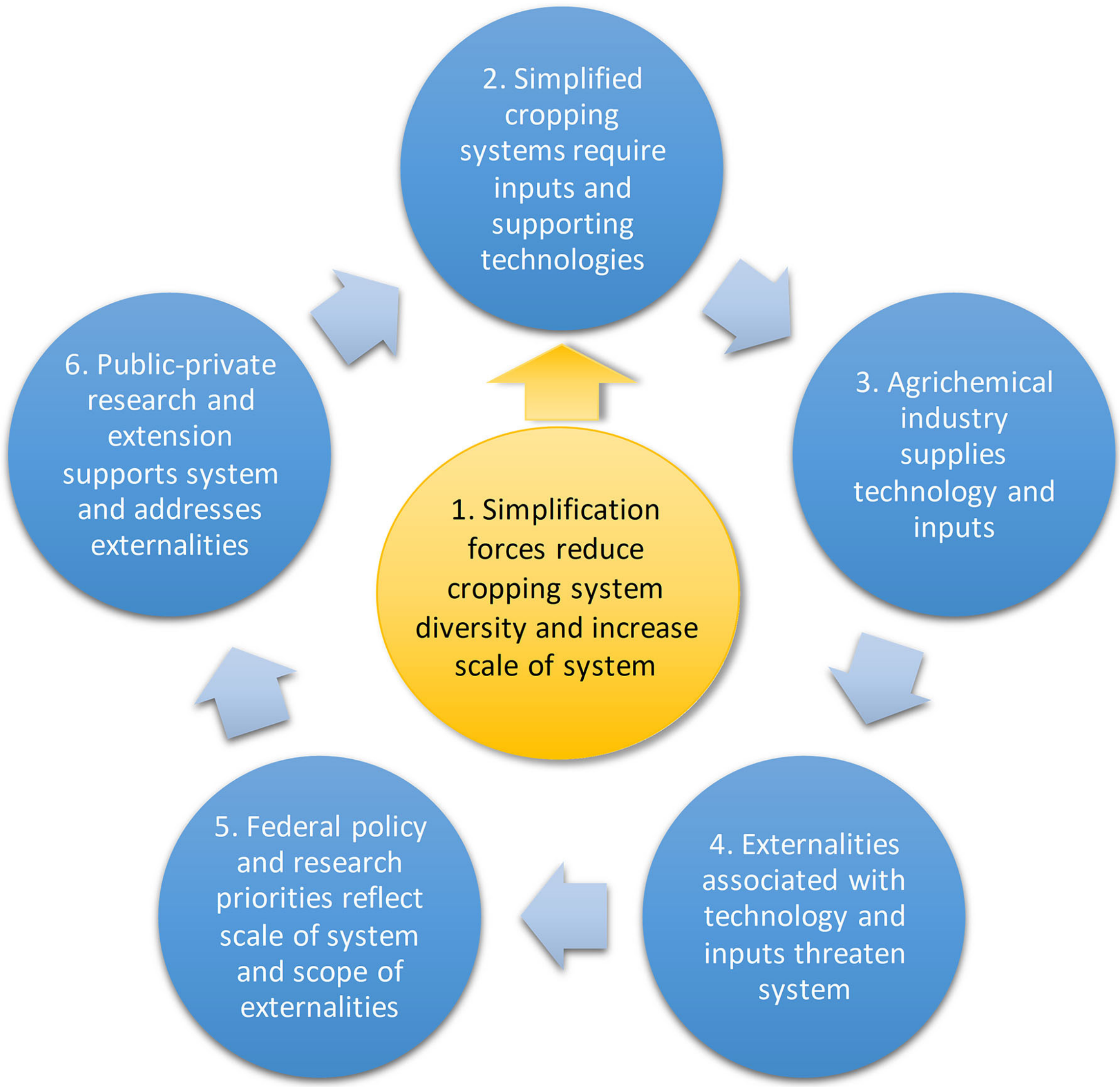

FIGURE 3 | The crux of the cropping system diversification problem: feedbacks that reinforce cropping system simplification. We argue that steps 5 and 6 are where efforts could be made to break the cycle of simplification and encourage cropping system diversification at larger scales.

the system because it is so large-scale. Consequently, agency-funded research is aimed primarily at addressing inefficiencies of the system, along with the problems that arise because of the system, rather than seeking viable alternative systems (DeLonge et al., 2016; Miles et al., 2017). In so doing, the agencies' research funding priorities support the maintenance of this system. This focus on solving the problems that arise from the system also helps to facilitate partnerships between the same private sector industry that profits from the maintenance of the system and the public sector researchers and extension educators who understandably want to serve their farmer-stakeholders, many of whom are locked-in to the system. These private-public partnerships ultimately benefit the private sector by helping to reinforce the simplified system (Hendrickson, 2015). They can also reinforce the system by locking out competing innovations that would serve to diversify the system if those innovations do not have the support of private interests (Vanloqueren and Baret, 2009).

An assumption implicit in all of this is there must necessarily be a tradeoff between the scale of a farming system and its diversity. In other words, we tend to assume that large scale cropping systems must inherently be simple/low diversity and therefore heavily reliant on external agrichemical inputs with their concomitant externalities, ecosystem disservices, and sustainability challenges. The validity of this assumption is critical given the enormous footprint, both spatially and environmentally, that large scale farming systems have across the agricultural landscape (**Figure 1**). But is there a biological or ecological principle that specifically supports this assumption? The only principles that seem to support the tradeoff that we are aware of appear largely economic (Magdoff et al., 2000). If this is indeed the case, it suggests that current barriers to cropping system diversification might be overcome through actions at the federal policy level; actions that would prioritize cropping system diversification at large scales and catalyze research aimed at scaling up diversified farming systems and the practices and enabling technologies that would support them (Wigboldus et al., 2016; Miles et al., 2017). Our final section outlines a possible roadmap for achieving such actions.

A ROADMAP FOR OVERCOMING BARRIERS TO CROPPING SYSTEM DIVERSIFICATION

Our aim with this final section of the paper is to convey what it is we see as most needed by policy makers, agriculture scientists, food systems advocates, and other agriculture professionals in order to move agriculture writ large toward greater diversification specifically, and more broadly, an agriculture informed by our expanding knowledge of ecological functions and services. To that end, we see three broad strategies for achieving this objective and discuss each below. Our primary intent is not to provide a step-by-step plan for enabling each strategy, but rather shine a light on each with the hope that others will take the torch and illuminate a clearer and more productive way forward.

Agriculture Scientists Must Enhance the Scope and Scale of Systems-Level Research

We recognize that systems research is needed to elucidate field and landscape scale properties that underpin long-term delivery of ecological function and services (Kleijn et al., 2019). However, ecologically informed systems research alone will be insufficient to bring about enhancements in cropping system diversification at scales large enough to be beneficially impactful if the products of that research are perceived as only being relevant for smaller-scale farming systems (Wigboldus et al., 2016). Conversely, agroecologists and other agriculture scientists who are unwilling to engage in research that addresses the dominant large-scale agricultural systems are likely missing opportunities to make potentially impactful contributions to meaningfully address agriculture's larger environmental footprint. For example, there has been much agroecological research conducted on cover crops for use in smaller-scale and organic cropping systems. Consequently, in organic agriculture cover crops are required on all annually cropped acres, resulting in very high levels of adoption of the practice on certified organic farms. However, the fact that only 1–2% of US farmland is managed organically means the aggregate effects of this particular cropping system diversification practice on ecosystem function and services at larger landscape scales is limited. What is more, because cover crop adoption on the other 98% of US farmland managed conventionally is low, ranging from 1 to 3% (Hamilton et al., 2017), the greater benefits to society that this practice could generate are, at present, largely unrealized.

Given that the majority of harvested cropland is in large-scale farms (**Figure 1**), research aimed at increasing cropping system diversification must involve identifying diversification strategies that can be demonstrated to work at these larger scales. What are the barriers to cover crop adoption on the other 98% of conventionally managed US farmland and how can these be overcome? What other strategies to cropping system diversification could apply to, and likely be adopted by, large farm systems and how could these be implemented? What federal economic policy mechanisms could be enacted or changed to facilitate cropping system diversification on large farming systems rather than incentivizing these same farms to maintain simplified systems? How do we close yield gaps in organic cropping systems and scale these systems up without simplification? These are some of the types of questions more agroecologists, in collaboration with rural sociologists and economists, should be asking. In order to enable more agricultural scientists to ask these types of questions, federal agricultural research funding priorities could place greater emphasis on research aimed at identifying holistic production systems where the systems and components of those systems are scalable to the extent possible (Miles et al., 2017). This leads to our second strategy.

Publicly Funded Research Should Address the Common Good, Rather Than Prop Up Unsustainable Systems

Many of the simplification forces described in the previous sections are the direct or indirect result of policies established by our federal agriculture agencies. Paradoxically, these same federal agencies also direct our public investment in agricultural research. The result is a national agriculture policy agenda that not only incentivizes, but often promotes, large-scale, low-diversity cropping systems and a national research agenda where much of the funded research addresses problems associated with large-scale, low-diversity cropping systems (Davis et al., 2009; Miles et al., 2017). This does little to serve the common good. We need look no further than the case study detailed in this paper to support this argument. Hence, problems associated with large scale, simplified cropping systems, like herbicide resistance and environmental pollution, will continue to drive the research agendas of agriculture scientists until there are changes to either federal policy or the national research agenda (Davis et al., 2009; Harker et al., 2017; Miles et al., 2017).

We argue that the federal research agenda needs to prioritize cropping system diversification at all scales. Similarly, while productivity of *all* farms will need to continue to rise over the coming decades, there will also need to be a disproportionate focus placed on improving the long-term sustainability and environmental footprint of farming, and this clearly must include large-scale farms (Hunter et al., 2017). Further, we recognize the capacity for such research to bring about changes in practices on the ground will be much greater with farmers engaged in the research process (Rosmann, 1994; Hassanein, 1999; van de Fliert and Braun, 2002). Such participatory methods are proving to be particularly effective at identifying suites of context-dependent agroecological practices that are effective and manageable at farm scales (Blesh and Wolf, 2014; White et al., 2017).

Land-grant institutions will also need to play a role in better serving the common good. When funding comes directly from agrichemical companies to public sector agricultural scientists or extension personnel, there is a strong incentive to focus the research on the issues that the company's products address (Davis et al., 2009; Harker et al., 2017). If these issues are the result of cropping system simplification itself, such as the rise in glyphosate-resistant weeds owing the overuse of glyphosate in GR cropping systems, the resulting research and extension is likely to further entrench the simplified cropping system rather than lead to alternatives to the system (**Figure 3**). A disproportionately large amount of land grant research has been aimed at propping up unsustainable cropping systems, and for too long these institutions have served as the research and marketing arms of the seed and agrichemical companies (Magdoff et al., 2000). Reducing the influence agrichemical companies have on land grant research and extension programs would likely open a much larger range of potential solutions, beyond just those in which private firms have an interest (Vanloqueren and Baret, 2009).

Agricultural Scientists Must Engage in Policy and Rule-Setting

Scientists need to engage in the process of rule-setting. Rule-setting that regulates agricultural practices or that incentivizes practices through a variety of regulatory or economic mechanisms (MacDonald et al., 2013) is fraught with economically motivated conflicts of interest biased to support a larger, vertically integrated and more simplified agriculture. It should be the role of the scientist to translate their work in a way that informs ecologically based decision-making. In the absence of this type of decision-making, private sector interests will offer a production-oriented justification that argues for simplification and vertical integration and where little attention is paid to the externalities of production (Harker et al., 2017). The voice of the scientist in this process is largely absent, or when it is invoked, is usually that of a neutral arbiter. Scientists are often called on to present or interpret data when policy makers advance to rule-setting. What is needed is something more than a presentation of the data; what is needed are interpretations of the root cause of problems of an overly simplified agriculture and a decision process that empowers scientists to measure or project the externalities of production practices that enable a more ecologically-weighted policy making. Much has been written about this potential role for scientists (Hoppe, 1999; Crouzat et al., 2018; Sarkki et al., 2020). Are they neutral arbiters, honest brokers, or issue advocates? We argue that agroecologists and other agricultural scientists have largely limited their role to that of the neutral arbiter and therefore have blunted their ability to shift agriculture toward one that places greater value on lessening its environmental impact through such practices as increased crop diversification.

CONCLUSIONS

The agroecology literature is replete with papers expounding the benefits of cropping system diversification and the necessary role that crop diversity plays in facilitating a more sustainable system of agriculture (e.g., Altieri, 1999; Lin, 2011; Bommarco et al., 2013; and many more), including broad brush calls for "agroecological transformation of monocultures" as "a strategy that represents a robust path to increasing the productivity, sustainability, and resilience of agricultural production" (Altieri et al., 2015). Similarly, recent research indicates many monoculture farmers do recognize the potential agroecological value that diversifying their cropping systems would hold for them in terms of their ability to deal with climate change and that they would like to implement diversification strategies if possible (Roesch-McNally et al., 2018). However, being willing to do something and being able to do something are two very different things. As we hope we have made clear, many farmers are effectively locked-in to their simplified cropping systems due to a variety of factors, some of which are in their control but many of which are largely outside of their control. Addressing these factors head on is likely the only way that cropping system diversification will occur on scales large enough to have meaningful beneficial environmental impacts. We have outlined several strategies, aimed primarily at agricultural scientists and their research funders, for addressing these issues, including expanding the agroecology research agenda to include large-scale farming systems and the search for scalable practices that can actually be integrated into large farms, as well as encouraging agricultural scientists to take a more participatory approach to policy and rule-setting; however, many other strategies likely exist. In that vein, a major potential mechanism for overcoming diversification barriers that we have not touched upon is the power of consumers to directly influence how crops are grown. There are a range of potential actions within this realm, from certification of biodiversity-friendly agriculture to radical reorganization of supply chains, highlighting the need for greater collaboration between the natural and social sciences (e.g., Robertson and Swinton, 2005; Salliou et al., 2019; Valencia et al., 2019). Lastly, federal policy and research funding should be redirected toward incentivizing cropping system diversification and away from initiatives that support unsustainable cropping systems. While our lens on these strategies is from the perspective of the state of farming and policy in the US, it is important to note that these same issues are being addressed and debated elsewhere, with varying degrees of success (e.g., Pe'er et al., 2019). It is likely that the outcomes of these efforts, many of which are occurring within governance systems that differ from those in the US, will provide additional examples for how limited public resources can be invested into agricultural practices that more effectively support the public good.

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Evaluating the Untapped Potential of U.S. Conservation Investments to Improve Soil and Environmental Health

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There is increasing enthusiasm around the concept of soil health, and as a result, new public and private initiatives are being developed to increase soil health-related practices on working lands in the United States. In addition, billions of U.S. public dollars are dedicated annually toward soil conservation programs, and yet, it is not well quantified how investment in conservation programs improve soil health and, more broadly, environmental health. The Environmental Quality Incentives Program (EQIP) is one of the major U.S. public conservation programs administered on privately managed lands for which public data are available. In this research, we developed a multi-dimensional classification system to evaluate over 300 EQIP practices to identify to what extent practices have the potential to improve different aspects of soil and environmental health. Using available descriptions and expert opinion, these practices were evaluated with a classification system based on the practice's potential to exhibit the following environmental health outcomes: (i) principles of soil health to reduce soil disturbance and increase agrobiodiversity; (ii) a transition to ecologically-based management to conserve soil, water, energy and biological resources; and (iii) adaptive strategy to confer agroecosystem resilience. Further, we analyzed nearly $7 billion U.S. dollars of financial assistance dedicated to these practices from 2009 through 2018 to explore the potential of these investments to generate environmental health outcomes. We identified nine practices that fit the highest level of potential environmental health outcomes in our classification systems. These included wetlands and agroforestry related practices, demonstrating that ecologically complex practices can provide the broadest benefits to environmental health. Practices with the greatest potential to improve environmental health in our classification system represent 2–27% of annual EQIP funding between 2009 and 2018. In fiscal year 2018, these practices represented between $13 and 121 million, which represented ∼0.08% of total annual USDA expenditures. These classifications and the subsequent funding analysis provide evidence that there is tremendous untapped potential for conservation programs to confer greater environmental health in U.S. agriculture. This analysis provides a new framework for assessing conservation investments as a driver for transformative agricultural change.

Keywords: Environmental Quality Incentives Program (EQIP), soil health, agroecology, adaptive capacity, environmental health, resilience

INTRODUCTION

Soil health has emerged in recent years as a unique and powerful solution to many of the 21st century's most wicked problems: the degradation of natural resources, food security, and climate change. Despite this more recent emergence, soil health is an ancient and ubiquitous agricultural concept. Described in 2,000 year-old Greek and Roman treatises on agricultural productivity (Karlen, 2012), soil health remained a fundamental principle through more than 200 years of modern agricultural philosophical development (Kuepper, 2010). Following the Green Revolutions of the early 20th century, industrialized agricultural systems focused more on external inputs and efficiency while soil health principles were deprioritized (Gliessman, 2014). The growing focus on soil health is a necessary shift in order to address generations of unsustainable agricultural practices in the U.S. that have led to soil degradation through processes such as erosion, salinization, compaction and decreased soil organic matter (Hatfield et al., 2017). The Food and Agriculture Organization of the United Nations estimates that 25% of the world's agricultural lands are highly degraded (FAO, 2011). Degraded soil resources are increasingly recognized as reducing the capacity of agricultural systems to respond to climate change; and further, improving soil health is a "multi win" approach to generate many valuable social and environmental co-benefits in addition to reducing climate risks (Webb et al., 2017).

Soil health is broadly recognized as the capacity of a soil to function as a vital living ecosystem that sustains plants, animals, and humans (USDA-NRCS, 2020a) although different definitions and conceptualizations of soil health exist, and are evolving (Karlen et al., 2017). Agricultural practices that promote soil health emphasize principles of reduced soil disturbance, increased crop diversity, continuous soil cover and living roots, and the integration of livestock. The intent of these principles are to promote soil health by preventing erosion, increasing soil organic matter, and supporting more resilient agricultural production systems over time. Examples of some common soil health practices include cover cropping, using organic amendments, reducing tillage, and rotating crops (Tully and McAskill, 2020).

As awareness has grown of the many social and ecological benefits associated with implementing soil health building practices, these practices are being promoted by an increasingly diverse chorus of voices. For example, soil health is emerging as an effective natural climate solution (Griscom et al., 2017) that is supported by agribusiness such as Indigo Ag (2020)[1], Danone (2020)[2], and General Mills (2020)[3], federal policymakers introducing legislation such as the 2019 Climate Stewardship Act (U.S. Congress, 2019) as well as the 2020 Agriculture Resilience Act (U.S. Congress, 2020), and documents such as the USDA's 2020 Agriculture Innovation Agenda (USDA, 2020), and by soil health proponents. All of these stakeholders focus their support on one or more of the many diverse co-benefits of healthy soils, such as improved water quality (Bodell et al., 2019), food quality and human health (Soil Health Institute, 2018), and reduced flood and drought impacts (Basche, 2017). Increasingly, the concept of improving soil health is at the nexus of conversations focused on reducing negative environmental impacts of agriculture, increasing carbon sequestration, and expanding climate resilience.

While there are no federally supported programs with a singular goal of addressing soil and environmental health on working lands, there are a number of programs that address different aspects of conservation, including land retirement programs such as the Conservation Reserve Program (CRP) and working lands programs such as Environmental Quality Incentives Program (EQIP) and Conservation Stewardship Program (CSP) (CRS, 2019). At the core of these programs, there are more than 300 practice standards that guide conservation planning and implementation by Federal and State agencies (USDA-NRCS, 2020b). The Natural Resources Conservation Service (NRCS) of the U.S. Department of Agriculture is responsible for developing and updating conservation practice standards for working and non-working farmlands, ranchlands, and forests. Each conservation practice standard contains information on why, when, and where a specific practice can be implemented, and sets forth the minimum criteria to be met for the practice to achieve its intended purpose. The standards are also evaluated according to their contribution to different program goals, such as soil or water quality improvement, pesticide use reduction, and more recently, greenhouse gas emission reduction and carbon sequestration (USDA-NRCS, 2020c).

The goal of this research is to explore how recent investments in EQIP have the potential to impact soil and environmental health in U.S. working lands through a practice-based assessment

[1]Terraton Initiative (2019). *Indigo Ag*. Available online at: https://www.indigoag.com/the-terraton-initiative (accessed March 24, 2020).

[2]Danone (2020). *Regenerative Agriculture*. Available online at: https://www.danone.com/impact/planet/regenerative-agriculture.html (accessed March 24, 2020).

[3]General Mills (2019). *Regenerative Agriculture Program*. Available online at: https://www.generalmills.com/Responsibility/Sustainability/Regenerative-agriculture (accessed March 24, 2020).

that builds upon multiple pre-existing principles. Specifically, we developed outcomes-based classifications that describe the target and potential outcome of practice implementation in the EQIP conservation program. The first of these classifications is based on the ability of a practice to address the principles of soil health, specifically, reducing disturbance or erosion and increasing plant and/or livestock diversity. The second classification is based on agroecological principles articulated by Gliessman (2014), representing the increased potential of the practice to transition an agroecosystem to ecologically-based management, with a goal of conserving soil, water, energy and biological resources. The third classification was based on adaptive strategy criteria following the framework of Walthall et al. (2013), describing the strategy of a practice to change the ecological structure and function of an agroecosystem in response to change. In aggregate, these classifications represent complimentary frameworks for evaluating the potential of U.S. conservation investments to address the increasing interest and demand for improving soil health as well as the environmental co-benefits soil can provide, such as resource conservation, water quality, protecting and improving rural livelihoods, and enhancing adaptation and resilience to the effects of climate change.

This research focuses on the impact of federal investments in soil health and broader environmental health administered by the USDA-NRCS through EQIP. Historically, a major emphasis of the EQIP program has been on livestock and wildlife related practices, which are meant to receive 55% of overall funding (CRS, 2019; USDA-ERS, 2019). Therefore, we examine the potential for EQIP practices to provide broad co-benefits to soil and environmental health. Specifically in this research, we (1) evaluate all current practices with EQIP through a multidimensional classification system to understand their potential to improve soil and environmental health; (2) quantify the number of federal dollars allocated to these practices between 2009 and 2018. This research critically evaluates how conservation dollars are being spent with respect to soil and environmental health. It is our aim that this and subsequent research can help inform future investment in those practices that provide the broadest benefits.

METHODS

EQIP Program Data Selection

Our analysis focused on EQIP for several reasons: First, EQIP is one of the major U.S. public conservation programs administered on private managed lands, and county-level data by conservation practice from the Resource Economics and Analysis Division of the NRCS were readily available. This enabled detailed analysis of the number of acres and dollars allocated to practices that improve soil health and broader environmental health within EQIP. Second, the data on individual and specific conservation practices at the county-level are not available for other conservation programs such as the Conservation Stewardship Program (CSP) or the Conservation Reserve Program (CRP). Finally, we focus on EQIP investments because the number of acres enrolled in EQIP practices that are estimated to improve soil health are several times larger than CRP (USDA-NRCS, 2020d).

Data Acquisition

Data for all EQIP practices from 2009 to 2018 was acquired by public request from the Strategic Information Team in the Resource Economics and Analysis Division of the Natural Resources Conservation Service (USDA-NRCS, 2020f). County-level data were provided for acreage and dollars from years 2009 to 2018 on all available 304 approved conservation practices across all states.

Conservation Practice Outcomes-Based Classifications

Practice standard information was obtained from the USDA-NRCS National Conservation Practice Standards catalog to assist in categorizing each practice according to the multi-dimensional outcomes-based classification criteria (described below). This catalog is available online, and describes technical information for the various approved conservation practices (USDA-NRCS, 2020b). State and local NRCS offices select conservation practices from these standards to develop conservation programs that are better suited to local needs. The practice standards are the basic organizing tool for agricultural conservation in the U.S. and are widely used in agricultural conservation program development, planning, and assessment at the local-, state-, and national-level.

We categorized the conservation practices according to criteria specific to each of the following classifications (**Figure 1**):

1. Practices were categorized by **type of** practice, indicating the primary asset or land use targeted by the practice, including the following categories (**Figure 1A**): (1) *data monitoring*, collection and evaluation; (2) a physical *structure or facility* or an equipment-related practice; (3) edge of field or *boundary practice* occurring on the perimeter of a managed land; (4) *in-situ* practice occurring directly in the *managed land*; and (5) *plan* or management description. Facilities included such practices as waste storage facility, compost facility, and sediment basin. Edge-of-field or *boundary practices* could include ditches or diversions that require soil movement and are utilized on the edge of an agricultural or pasture area. *In-situ* practices include tillage, cover cropping, alley cropping, etc. that occur within the field or pasture. For our classification system, wetland related practices (creation or restoration, for example) were included in the *in-situ* category as they are often located on previously cultivated fields, pastures, or forested areas, and therefore represent active management decisions where farmable land is taken out of production in order to optimize use.
2. Practices were then categorized by their **soil health outcome** on two axes describing whether or not the practice has the potential to reduce erosion and soil disturbance and/or increase agrobiodiversity (plants and/or livestock). In general, this classification was created to incorporate the widely accepted soil health principles from the NRCS (USDA-NRCS, 2020a). We created a matrix of four quadrants and a binary scheme (yes or no) for the two categories: reducing disturbance or erosion (x axis) and increasing agrobiodiversity (y axis) (**Figure 1B**). *Quadrant 1* represented practices that would not reduce disturbance or erosion nor increase plant or livestock diversity. *Quadrant 2* represented practices with

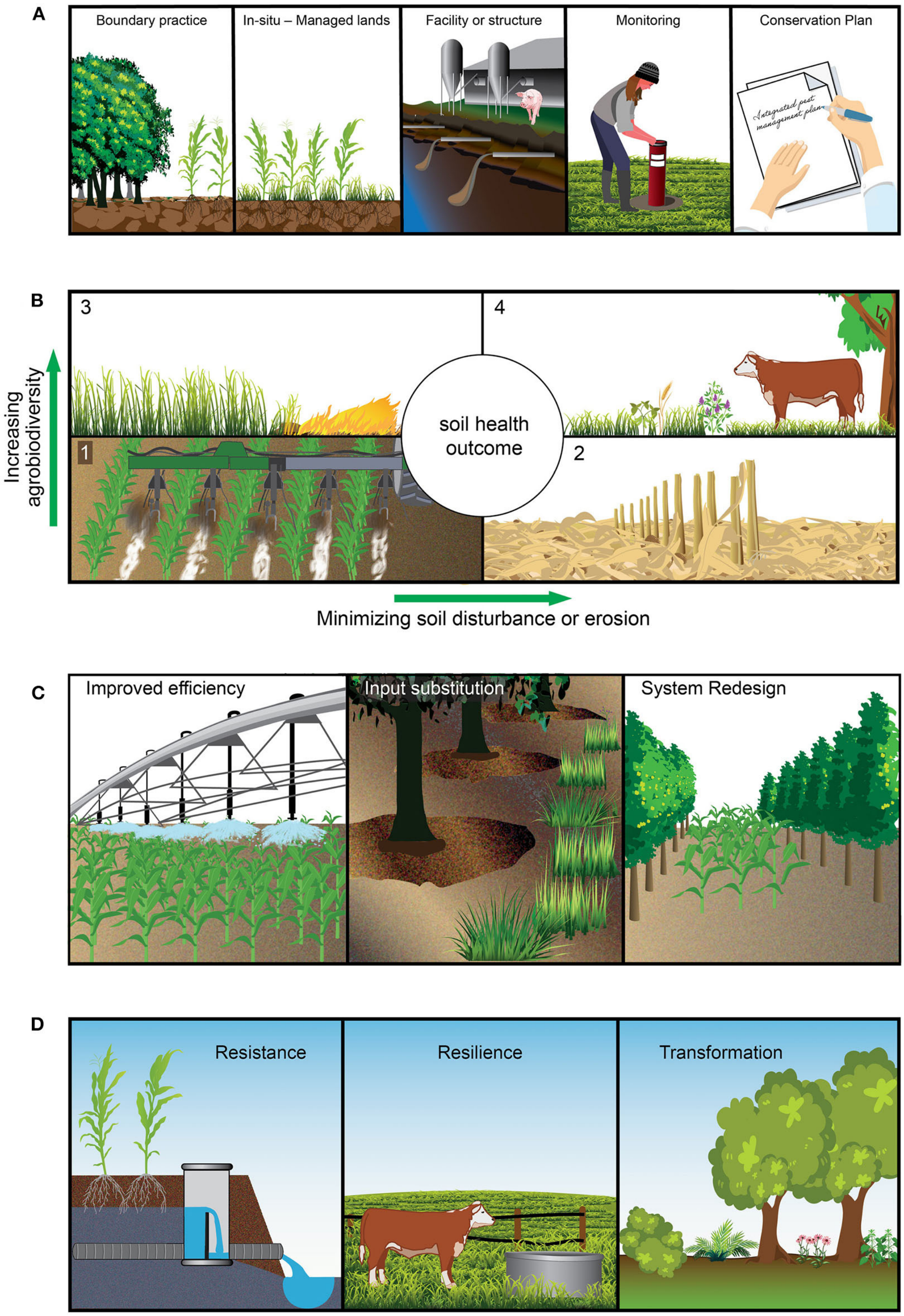

FIGURE 1 | Representations of practices and the various categories included in this analysis. USDA NRCS National Conservation Practice Standards code numbers are included below in parenthesis and can be found at the Standards Catalog Website (USDA-NRCS, 2020b). **(A)** illustrates the categories within the "type" classification, including: boundary practices, represented by Windbreak/Shelterbelt Establishment (380); managed lands, represented by Cover Crop (340); facility or structure, represented by Waste Storage Facility (313); monitoring, represented by Groundwater Testing (355); and conservation plan, represented by IPM Herbicide Resistance Weed Conservation Plan - Written (154). **(B)** illustrates the categories within the "soil health outcome" classification, including: *Quadrant 1*, no change or

(Continued)

FIGURE 1 | improvement to soil health, represented by Nutrient Management (590); *Quadrant 2*, reducing soil disturbance or erosion, represented by Residue and Tillage Management, No-Till (329); *Quadrant 3*, increasing agrobiodiversity, represented by Prescribed Burning (338); and *Quadrant 4*, both reducing soil disturbance or erosion and increasing agrobiodiversity, represented by Silvopasture (381). **(C)** illustrates the categories within the transition to ecologically-based management classification, including: *Level 1*, increasing efficiency, represented by Irrigation Water Management (449); *Level 2*, input substitution, represented by Mulching (484); and *Level 3*, system redesign, represented by Alley Cropping (311). **(D)** illustrates the categories within the adaptive strategy classification, including: *Resistance* strategies, represented by Drainage Water Management (554); *Resilience* strategies, represented by Prescribed Grazing (528); and *Transformation* strategies, represented by Multi-story Cropping (379).

the potential to reduce disturbance or erosion but not increase plant or livestock diversity. *Quadrant 3* represented practices that would not reduce disturbance or erosion, but with the potential to increase plant or livestock diversity. *Quadrant 4* represented practices with the potential to both reduce disturbance or erosion and to increase plant or livestock diversity.

Each practice was categorized based on the purpose of the practice as described in the practice standard. Specifically, we focused on those descriptions which were clearly relevant to reducing erosion or enhancing agrobiodiversity. We also considered the conservation practice's physical effects (CPPE) score (USDA-NRCS, 2020c). This is a score created by NRCS on a −5 (detrimental effects) to +5 (positive effects) scale, representing the potential of a practice to either increase or decrease particular environmental effects. We only considered scores for soil erosion impacts (e.g., sheet and rill, wind, ephemeral gully, classic gully) and used the CPPE scores as a guideline to support the final determination of the practice's efficacy in improving soil health. We determine that a practice would not change disturbance or erosion if the score was =<2 and it would reduce disturbance and erosion if the score => 3. We found that a few practices had the potential to degrade rather than improve soil health and created a category of "negative" to note practices where this was likely to occur.

To validate the soil health outcome classification, we used the categorization of all the practices listed to improve "soil quality" in the 2019 Soil and Water Resources Conservation Act (RCA) report (USDA-NRCS, 2020d) to ensure that these practices were similarly coded as having an effect on soil health. In summary, for this classification we used three sources: expert knowledge, the CPPE scores, and the 2019 RCA report.

3. Practices were also categorized based on their representation of a **transition to ecologically-based management**, using the principles of agroecology as defined in Gliessman's (2014) framework. This framework describes a continuum of practices in a transition or conversion to more ecologically-based management, illustrated by their emphasis on principles such as reestablishing "the biological relationships that can occur naturally on the farm instead of reducing and simplifying them" and emphasizing "conservation of soil, water, energy and biological resources" (Gliessman, 2014). Three category levels (i.e., levels of agroecology) were included in this classification: *Level 1* refers to practices that are primarily focused on improving efficiency of inputs (i.e., irrigation system improvements, reductions in energy use, waste management), where practice descriptions often included the language "efficiency"; *Level 2* refers to practices that are primarily focused on substitution of inputs that are generally understood to be less harmful (i.e., substitute inorganic fertilizer for compost); *Level 3* refers to practices that are primarily focused on systemic redesign at the farm-level (i.e., increase agrobiodiversity through hedgerows, intercropping, integrating crops and livestock) (**Figure 1C**). The ecologically-based management framework and classification levels was used to evaluate investment in sustainable agriculture research by the United States Department of Agriculture (USDA) Research, Extension & Economics (REE) Mission Area (DeLonge et al., 2016).
4. Finally, we categorized practices using **adaptive strategy** principles, which consider management strategies along a continuum of change to the ecological structure and function of the agroecosystem (Walthall et al., 2013). *Resistance* strategies require the least change in agroecosystem form and function and are typically reactive interventions that target specific threats with technological tools, for example the addition of irrigation in areas experiencing more frequent and intense drought or increased use of pesticides in an area experiencing higher pest pressures. Financial tools such as subsidized insurance and disaster relief are also included in *Resistance* strategies. *Resilience* strategies change the form and function of the agroecosystem in order to buffer the effects of disturbances and support a rapid return to a healthy condition after a disturbance, with no or minimal management intervention. *Resilience* strategies are typically proactive interventions that increase the functional and response diversity of the agroecosystem and reduce risks associated with multiple threats. For example, the adoption of soil health practices like cover crops and more diversified crop rotations buffer the effects of more variable rainfall, spread production risk across a variety of crops, reduce year-to-year yield variability, and can enhance profitability (Walthall et al., 2013). *Transformation* strategies facilitate the transition to a new agroecosystem with substantially different structure and function better suited to sustained production under current or projected conditions. *Transformation* strategies are typically proactive interventions designed to better position the agroecosystem to sustain production over the long term and reduce risks associated with multiple threats. These strategies also tend to produce multiple benefits to the producer and the local community (**Figure 1D**). For example, transitioning a conventional row crop operation in a floodplain to a managed grazing operation reduces the risk of flood damage to the operation, enhances regional water quality, and can also reduce the risk of downstream flooding (Basche, 2017). This resistance-resilience-transformation framework

was recommended for use in an ecosystem-based approach to agricultural adaptation (Easterling, 2009). It has been applied to climate risk management planning in U.S. National Forests (Spies et al., 2010; USDA Forest Service, 2010) and the National Fish, Wildlife, and Plants Climate Adaptation Strategy (NFWPCAP, 2012), and is used to categorize adaptation options in the USDA Climate Hub's Adaptation Workbook (Janowiak et al., 2016).

Where **information was limited or not applicable** we categorized practices as the following: (1) None: no info was available on the NRCS practice website and the practice name did not provide enough context to categorize; (2) NEI: not enough information from available resources to categorize the practice for a given classification; (3) NA: information was available to suggest that the practice was not applicable to a particular aspect/classification system.

While we recognize how critical the context is under which a practice is implemented, we considered the most optimal *potential* environmental outcomes, from a soil health, transition to ecologically-based management, and adaptive capacity perspective could occur from *in-situ* practices occurring directly on managed lands and that fit our highest ranking categories for each classifications (Soil health outcome *Quadrant 4*; Transition to ecologically-based management *Level 3*; Adaptive strategy *Transformation*). The implementation of multiple evaluative frameworks allows us to identify those practices that are most likely to provide numerous co-benefits to soil and the environment health.

A complete version of the classification system and dollars associated with practice codes are available in **Supplementary Table 1**.

Categorization of Practices

Based on the expertise of the research team, which was composed of agronomists, soil scientists, ecologists, and social scientists, three team members were selected to lead the classification of the 304 EQIP conservation practices using a coding system that was developed (see full list in section Conservation Practice Outcomes-Based Classifications).

First, ∼50% of practices were categorized independently by three team members. Intercoder agreement was evaluated by comparing the results from the three independent coders. Discrepancies were discussed in detail among the coders and the team. As needed, discrepancies were resolved using appropriate team expertise on USDA-NRCS practice planning and implementation. All practices were then coded two additional times among the experts to ensure consistency of categories among the different outcomes-based classifications.

Of the 304 practices examined, many could not be categorized according to all of the classifications. The full list included 88 "interim" practices, defined as those that are undergoing a 3-year trial period (USDA-NRCS, 2007). Based on the time lag of the introduction of these practices, some were duplicative and others no longer have description information available in the online catalog. We were unable to classify these practices because practice standards were not available in the catalog. Further, we found in our preliminary analysis that the interim practices comprised <1% of total funding distributed in the last 5 years, so we decided not to seek the additional information required to classify interim practices in this analysis. Further, detailed standards information was not available for 36 additional practices on the national list provided to us. However, we categorized some of these practices for "type" without more detailed descriptions, because the title of the practice made classification possible. For example, we could categorize some practices if the title included, "plan" or "monitor." As a result, a total of 203 total practices were categorized in the type classification. Following the classification of practice "type," there remained 180 practices that contained descriptions and were coded for the three remaining classifications.

EQIP Program Analysis

Based on the above classifications and the EQIP data procured via public records request, we determined the distribution of EQIP practices within each classification criteria for their potential to improve soil health, transition to ecologically-based management, and expand adaptive strategy. We also calculated the acres and dollars allocated to different practices at a county-level over the 10 year period of 2009 to 2018. The data provided included a column for contract status. We only included "partial certified" and "certified" status contracts because other categories did not have any dollars or acres associated with them. We worked with data based on the "contract year" which would be when the practice payout occurred rather than when the contract was initiated. Data analysis was completed using the R software platform (R Core Team, 2020) as well as ArcGIS (ESRI, 2010) and is summarized as percentages and totals.

RESULTS

EQIP Spending Overview From 2009 to 2018

The U.S. Department of Agriculture's budget for 2018 was estimated to be $144 billion dollars, with conservation programs administered through the most recent Farm Bill to be ∼$6.7 billion (CRS, 2018; USDA, 2019). EQIP is estimated to invest nearly $2 billion annually in structural, vegetative and management practices on eligible lands (USDA-NRCS 2020d; CRS 2019) (**Figure 2**). From this annual investment, between 2009 and 2018, it is estimated that ∼$9 billion dollars went specifically to financial assistance supporting practices (USDA-NRCS, 2020d), after subtracting dollars allocated to reimbursables and technical assistance. The contract data provided for the years of 2009–2018 totaled $6.95 billion, which represents ∼76% of the total dollars spent for financial assistance in EQIP over this time period. We understand that the data provided to us represents actual dollars spent on practice contracts, whereas other reporting (i.e., USDA-NRCS, 2020d) notes total allocated dollars over a period of time that may not yet have been spent (NRCS Data Team, personal communication).

We first analyzed the database to determine which practices received the most funding overall during this 10 year period. The ten practices receiving the greatest amount of EQIP dollars

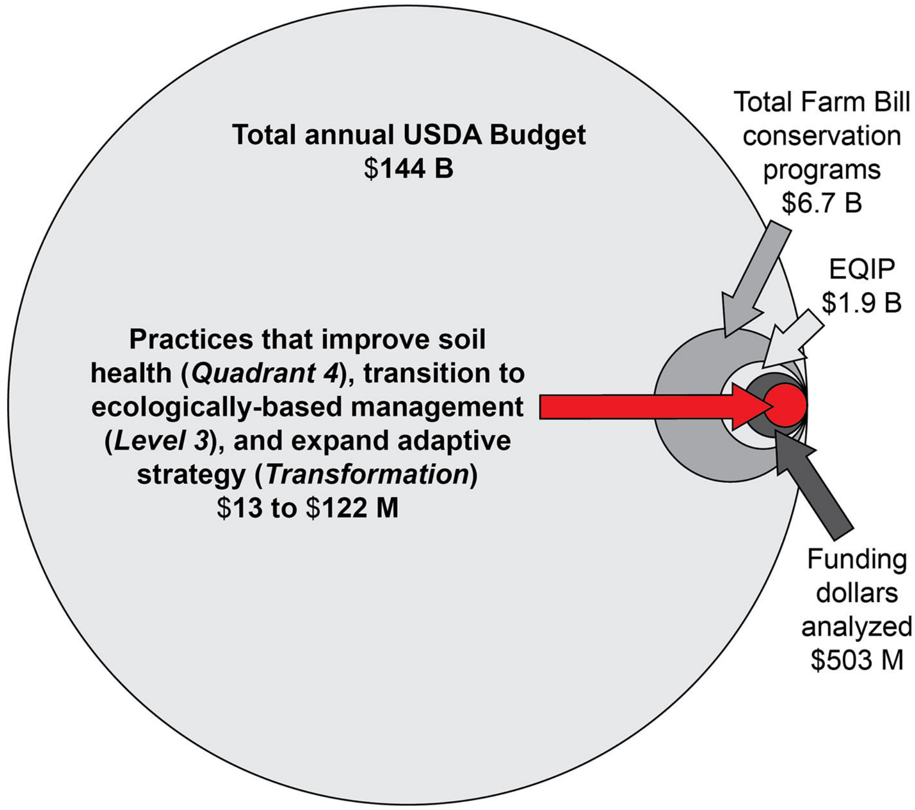

FIGURE 2 | Summary of funding dollars analyzed in this research. We found that investments in conservation practices that improve soil health, transition to ecologically-based management and expand adaptive strategy within EQIP comprise between 13 and 122 million dollars, which represents less than one-tenth of 1% of the U.S. Department of Agriculture's annual budget which was 144 billion in 2018 (USDA, 2019).

from 2009 to 2018 comprised ~46% of total spending, and the majority were not *in-situ* practices occurring on managed lands (**Supplementary Table 1**); six of the top ten were categorized as *facility* (Sprinkler System, Waste Storage Facility, Fence, Irrigation System, Irrigation Pipeline, Livestock Pipeline), three were categorized as *in-situ* practices occurring on managed lands (Cover Crop, Brush Management, Forage and Biomass Planting) and one was categorized as a *boundary practice* (Heavy Use Area Protection). Only two of the top ten funded practices met our criteria for the top category in any of our classification systems: cover crops and forage and biomass planting (soil health outcome *Quadrant 4* and transition to ecologically-based management *Level 3*). Over this time period, cover crops received ~$407 million (6% of the total dollars analyzed) and forage and biomass planting received ~$182 million (4% of total dollars analyzed).

EQIP Funding Primarily Focused on Structural, Efficiency and Non-transformative Practices

In order to understand the potential for EQIP practices to enhance environmental health, specifically to improve soil health, transition to ecologically-based management and expand adaptive strategy, we calculated the dollars allocated to the different categories (**Tables 1**, **2**). Beginning with the type of practice, we found that the largest percent of EQIP dollars were dedicated to the *facility* category (51%). *In-situ* practices occurring on managed lands comprised 39% of funding while *boundary* practices comprised 8%. Practices representing *plans* and/or monitoring systems represented a much smaller percentage of funding, at 2 and >1%, respectively (**Table 1**).

The soil health outcome classification was created to merge the five widely accepted principles of soil health into four quadrants form an x and y axis based on the practice's potential to reduce soil disturbance or minimizing erosion or increase agrobiodiversity (non-crop plants, crops, and livestock) (**Figure 1B**). We found that 9% of funding was allocated to practices representing no changes to soil health, or no reductions in soil disturbance, reducing erosion or increasing agrobiodiversity (*Quadrant 1*) (**Table 2**). We found that 16% of funding either minimized soil disturbance, reduced erosion, or increased biodiversity (*Quadrants 2 or 3*). Further, 22% of funding went to practices that achieved both goals (*Quadrant 4*). A small percent of funding went to practices that did not contain enough information to be classified or were determined to have a negative impact on soil health (3 and 2%, respectively). There were four practices that we felt could be categorized as having a negative impact on soil health and included irrigation

TABLE 1 | Dollars allocated to practices that fell within different categories in the classification of type of practice, describing its targeted land use.

Target land use (type classification)	Total payments 2009–2018 ($$)	% of total payments	Total acres	# of practice codes	# of contracts
Overall	6,948,067,609	100%	92,621,405	216	1,306,410
Boundary practice	521,949,147	8%	2,418,679	33	144,548
Structure or facility	3,573,944,969	51%	6,860,244	76	404,616
In situ practice on managed land	2,686,773,562	39%	79,572,871	68	688,529
Monitoring	995,829	0.01%	3	2	753
Plan	151,909,415	2%	3,306,748	24	65,226
No information	12,494,687	0.18%	293,122	13	2,738

A full list of practices, total dollars allocated, and categorizations used in this analysis can be found in the ***Supplementary Material****.*

TABLE 2 | Dollars allocated to practices that within each of the different classifications and categories included in our analysis.

Outcome classification	Category	Total payments 2009–2018 ($$)	% of total payments	Total acres	# of practice codes	# of contracts
	Overall	6,948,067,609	100%	92,621,405	216	1,306,410
Soil health outcome	No change to soil health (Quadrant 1)	641,491,414	9%	16,031,603	11	142,661
	Reduced disturbance or erosion (Quadrant 2)	1,023,796,334	15%	11,263,641	30	291,184
	Increased biodveristy (Quadrant 3)	63,296,375	1%	2,860,570	2	35,212
	Reduced disturbance or erosion and increased biodiversity (Quadrant 4)	1,507,645,577	22%	51,555,717	40	384,951
	Negative impact	113,633,641	2%	618,546	4	12,535
	Not enough information	53,302,645	1%	10,508	1	6,101
	No information	107,469,290	2%	469,176	36	34,914
	Not applicable to category	3,437,432,333	49%	9,641,907	92	398,852
Transition to ecologically-based management	Improved efficiency (Level 1)	3,495,166,193	50%	26,197,115	82	494,933
	Input substitution (Level 2)	264,395,194	4%	12,896,714	12	80,084
	System redesign (Level 3)	1,609,928,569	23%	48,109,916	43	420,217
	Not enough information	509,552,978	7%	3,041,311	4	148,431
	No information	107,469,290	2%	469,176	36	34,914
	Not applicable to category	961,555,384	14%	1,737,435	39	127,831
Adaptive strategy	Resistance	5,320,989,959	77%	36,211,210	138	859,540
	Resilience	1,378,187,989	20%	54,568,715	30	375,759
	Transformation	141,420,370	2%	1,372,304	12	36,197
	No information	107,469,290	2%	469,176	36	34,914

A full list of practices, total dollars allocated, and categorizations used in this analysis can be found in the ***Supplementary Material****.*

land leveling, land smoothing, land clearing and grazing land mechanical treatment. We found that about half, or 49% funding in the program, were not directly applicable to any of the soil health outcomes in this classification system.

In terms of the various practices representing a transition to ecologically-based management, we found that 50% of funding went to the *Level 1* category, representing practices that are aimed toward efficiency improvements (**Table 2**). We found that 4% of funding went to practices that could be classified as *Level 2* (e.g., input substitution) and 23% were classified as *Level 3* (e.g., system redesign). The remaining funding (23%) went to practices that either did not contain enough information to code in this classification, or were not applicable.

The adaptive strategy framework classifies practices according to successively greater change in the form and function of an agroecosystem in order to increase adaptive capacity (Walthall et al., 2013). The majority of funding went to practices in the *Resistance* category (77%) or those that avoid altering the existing structure and function of the production system, while 20% of funding went to practices classified under the *Resilience* class category (e.g., moderate adjustments to the structure and function of the existing production system to enhance functional

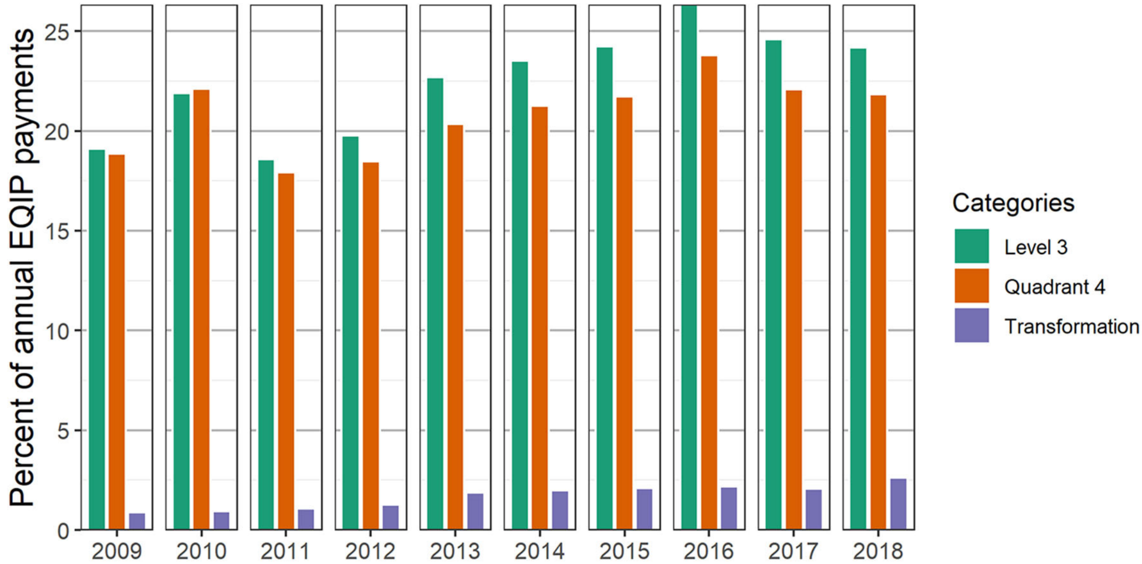

FIGURE 3 | Annual percent of total EQIP conservation practice investments from 2009 to 2018 dedicated to practices that were categorized as having the greatest potential to improve soil health (green), transition to ecologically-based management (orange), and expand adaptive strategy that were also executed *in-situ* on managed lands (purple). Total investments analyzed annually represented between $503 million (2018) up to $792 million (2013) (**Supplementary Table 2**). Over this time period, *Level 3* investments ranged from 19 to 26%, *Quadrant 4* investments ranged from 18 to 24%, and *Transformation* investments ranged from 1 to 3% (**Supplementary Table 2**).

and response diversity) (**Table 2**). We found that ∼2% of funding went to practices categorized as *Transformation* (e.g., major changes to the structure and function of the existing production system to enhance functional and response diversity) and another 2% did not contain enough information to categorize in this classification.

Investment in practices with the greatest potential to improve soil health outcomes (*Quadrant 4*, minimizing erosion and disturbance, maximizing biodiversity) and representing a transition to ecologically-based management (*Level 3*, system redesign) have fluctuated between ∼17–27% of total spending from 2009 to 2018 (**Figure 3**). In general, these practices increased from 2008 to 2010, decreased in 2011 and then increased again until 2016; however they have declined in the most recent years. Investment in the most optimal adaptive strategy practices (*Transformation*) have increased slightly over the last 10 years, but were never more than ∼3% of EQIP expenditures analyzed (**Figure 3**). The recent decline in *Level 3* and *Quadrant 4* funding may be partially explained by an increase in funding for the Conservation Stewardship Program (USDA-NRCS, 2020e), and potentially reflect farmer enrollment in different conservation programs.

Our analysis suggests that practices with the most potential to improve soil health, transition to ecologically-based management, and expand adaptive strategy executed *in-situ* on managed lands received ∼2–27% of EQIP funding, depending on the year (**Figure 3**). In fiscal year 2018, the total dollars spent on these practices was $13 million (*Transformation*), $110 million (*Quadrant 4*), and $121 million (*Level 3*), which represents at most 0.08% of the total $144 billion USDA annual budget (**Figure 2**).

Classification Systems Identify Overlapping, Complex Practices With the Largest Potential for Improving Environmental Health

Through our multi-dimensional classification system, we found a total of nine practices that fit categories with the greatest potential to improve environmental health (soil health outcome *Quadrant 4*, minimizing soil disturbance or erosion and increasing agrobiodiversity and; transition to ecologically based management, *Level 3*; Adaptive strategy *Transformation*) that were also executed *in-situ* on managed lands (**Figure 4**). These represent diversified crop, livestock, and forestry management including silvopasture and other agroforestry practices, wetland creation and restoration, and wildlife habitat management. The overlap of certain practices in all top categories demonstrate that our classification systems valued ecologically-complex practices that enhance biological diversification.

There was particularly high overlap in practices executed *in-situ* that were also categorized as both soil health outcome *Quadrant 4* and transition to ecologically-based management *Level 3*, where 28 of the 29 and 30 practices were the same, respectively (**Figure 4**). Some of the practices that comprised the largest percentages of funding within both of those categories included cover crops, forage and biomass planting, forest stand improvement, prescribed grazing, and watering facility (**Table 3**). For practices categorized as *Transformation*,

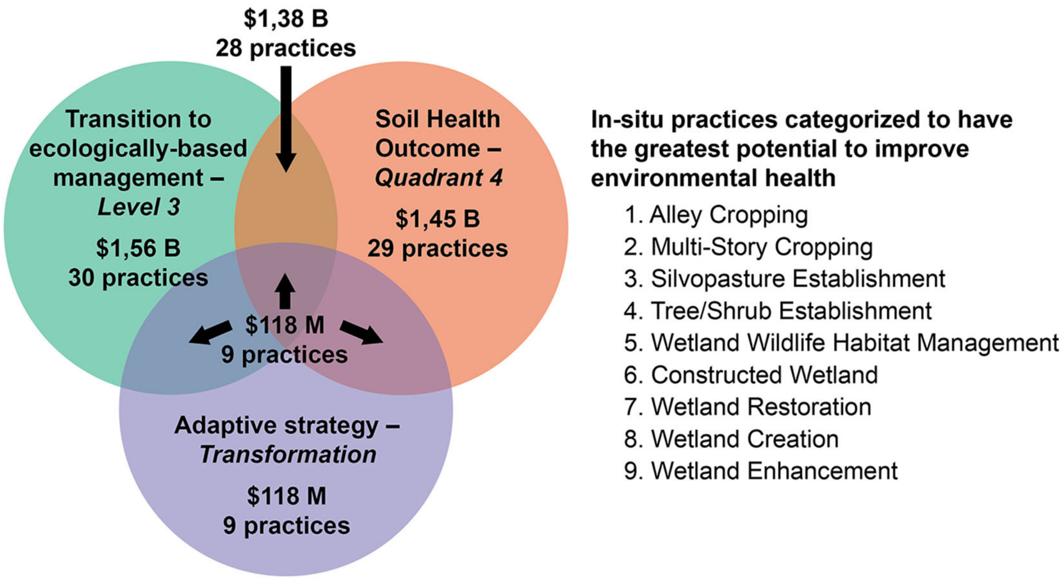

FIGURE 4 | Total invested dollars in the EQIP program from 2009 to 2018 in the classifications and categories having the greatest potential to improve soil health (green), transition to ecologically-based management (orange), and expand adaptive strategy (purple) that were also executed *in-situ* on managed lands. Overlapping practices meeting multiple top categories and invested dollars are identified in the Venn diagram. Our analysis identified nine overlapping *in-situ* practices on managed lands in the top categories for our classifications.

funding was dominated by tree/shrub establishment (*in-situ*) with a lesser amount dedicated to windbreak/shelterbelt establishment (*boundary practice*). A full list of practices, associated classifications, and total funding levels for the last 10 years can be found in **Supplementary Table 1**.

Top Practices Are Clustered Geographically

We evaluated the distribution of EQIP funds geographically to understand if there were regions where a greater percentage of the program dollars were dedicated to the most optimal practices (**Figure 5**). Regions with a greater percent of support for most optimal practices in soil health outcomes and ecological intensification were similar; we found regional clusters of high percentages (>40%) in the eastern Corn Belt, Northern and Southern Great Plains, as well as parts of the mid-Atlantic, Gulf Coast, Pacific Northwest, Alaska and Puerto Rico. In general, we found lower percentages (0–20%) of funding allocated for these practices in the Mid-Atlantic and Mountain West regions. In contrast, optimal transformation practices were funded at lower levels (0–20%) in West coast states, the Northeast, Great Lakes, Gulf Coast, Great Plains and Mountain West as well as Alaska, Hawaii and Puerto. In only a few scattered locations across the US did funding levels for these practices reach >40%.

Cover Crops Dominate Acreage for Conservation Spending on Optimal Practices

We investigated how EQIP acreage had shifted over the last 10 years for select practices that we considered to have the largest potential to improve environmental health. We focused on cover crops, forage and biomass planting and tree/shrub establishment which represented large percentages of investments across categories in our classification systems (section Classification Systems Identify Overlapping, Complex Practices With the Largest Potential for Improving Environmental Health) and were practices that were also consistently reported in acres. From 2009 to 2018 we analyzed funding representing ∼92.6 million acres of cropland (**Table 1**). Cover crops represented the largest number of acres of these three practices, then forage and biomass planting followed by tree/shrub establishment (**Figure 6**). Cover crop acreage supported by EQIP reached a maximum of ∼1.8 million acres in 2016; both forage and biomass planting and tree/shrub establishment were consistently under 250,000 acres combined. For context, there are ∼320 million harvested cropland acres in the U.S. (USDA-NASS, 2020). Although limited data exists to track national use of conservation practices, the 2017 Census of Agriculture estimated that there were ∼15 million acres of cover crops on U.S. cropland (USDA-NASS, 2020); by these estimates EQIP cover crop contracts would represent ∼10% of all cover crops utilized nationally.

DISCUSSION

Reprioritizing Conservation Investments to Generate Greater Soil and Environmental Health

In this research, we utilized a multi-dimensional classification system to evaluate conservation practices for their potential to improve soil health, transition to ecologically-based

TABLE 3 | Practices that comprised 70–90% of funding dollars analyzed in the EQIP program from 2009 to 2018 within the categories considered in our analysis to have the greatest potential to improve environmental health.

Outcome classification and category	Practice name	Total payments 2009–2018 ($$)	% of payments within Category	Target land use (type classification)	NRCS practice code	# of practice codes
	Overall	6,948,067,609	100%		-	216
Soil health outcome: Reduced disturbance or erosion and increased biodiversity (Quadrant 4)	*Quadrant 4 total**	*1,507,645,577*	-		-	*40*
	Cover Crop	406,855,053	27%	*in-situ* practice	340	-
	Forage and Biomass Planting	192,483,749	13%	*in-situ* practice	512	-
	Forest Stand Improvement	166,560,468	11%	*in-situ* practice	666	-
	Prescribed Grazing	115,964,156	8%	*in-situ* practice	528	-
	Tree/Shrub Establishment	107,485,940	7%	*in-situ* practice	600	-
	Terrace	94,930,987	6%	*in-situ* practice	612	-
Transition to ecologically-based management (Level 3)	*System redesign (Level 3) total**	*1,609,928,569*	-		-	*43*
	Cover Crop	406,855,053	25%	*in-situ* practice	340	-
	Forage and Biomass Planting	192,483,749	12%	*in-situ* practice	512	-
	Forest Stand Improvement	166,560,468	10%	*in-situ* practice	666	-
	Watering Facility	150,427,064	9%	*in-situ* practice	614	-
	Prescribed Grazing	115,964,156	7%	*in-situ* practice	528	-
	Tree/Shrub Establishment	107,485,940	7%	*in-situ* practice	612	-
Adaptive strategy (Transformation)	*Transformation total**	*141,420,370*	-		-	*12*
	Tree/Shrub Establishment	107,485,940	76%	*in-situ* practice	612	-
	Windbreak/Shelterbelt Establishment	13,363,897	9%	boundary practice	380	-
	Windbreak/Shelterbelt Renovation	9,539,717	7%	boundary practice	650	-
	Wetland Wildlife Habitat Management	6,032,925	4%	*in-situ* practice	644	-
	Wetland Restoration	1,459,018	1%	*in-situ* practice	657	-
	Silvopasture Establishment	1,176,973	1%	*in-situ* practice	381	-

Italics and starred rows (*) represent the total dollars that fell within the larger categories. Practice code numbers come from the NRCS Conservation Practice Standards catalog. A full list of practices, total dollars allocated, and categorizations used in this analysis can be found in ***Supplementary Table 1****.*

management, and expand adaptive capacity. Given increasing interest and public investments being dedicated to soil health and related environmental and social outcomes, we argue that there is a need to consider multiple criteria that can address potential co-benefits and scale of outcomes of such investments. Evaluating these practices through different classifications sheds light on those practices are most likely to provide a suite of environmental benefits. The growing interest in improving soil health recognizes these links as an opportunity to also improve environmental health by increasing water quality, enhance wildlife habitat, reduce greenhouse gas emissions, increase resilience to climate change, and sequester carbon (Griscom et al., 2017; Harrigan and Charney, 2019; Zimnicki et al., 2020).

Using different classifications to evaluate the practices allowed us to identify those with the highest potential to generate co-benefits. In our analysis, we found that these represent less than one-third of EQIP dollars on an annual basis, and overall a small fraction of USDA dollars allocated annually over the last decade. This is similar to a recent USDA-NRCS report which categorized ∼25% of acres enrolled in EQIP as "soil health practices" (USDA-NRCS, 2020d). We suggest that policymakers prioritize funding and outreach efforts to promote this set of optimal practices where applicable in order to increase return on public investment in conservation incentives. This finding echoes a recent Governmental Accountability Office (GAO) report indicating that EQIP could be better optimized to produce environmental outcomes (GAO, 2017). Specifically the GAO (2017) recommends that the USDA "modify guidance and ranking tools so that they more accurately value an EQIP application's anticipated environmental benefits relative to estimated costs." Our evaluation addresses this explicit recommendation, and by doing so identifies a few dozen practices (**Supplementary Table 1**) that could be promoted for their ability to generate a diversity of environmental and social benefits on managed lands, including enhancement of soil health, transitioning to more ecologically-based management and enhancing the adaptive capacity of agroecosystems.

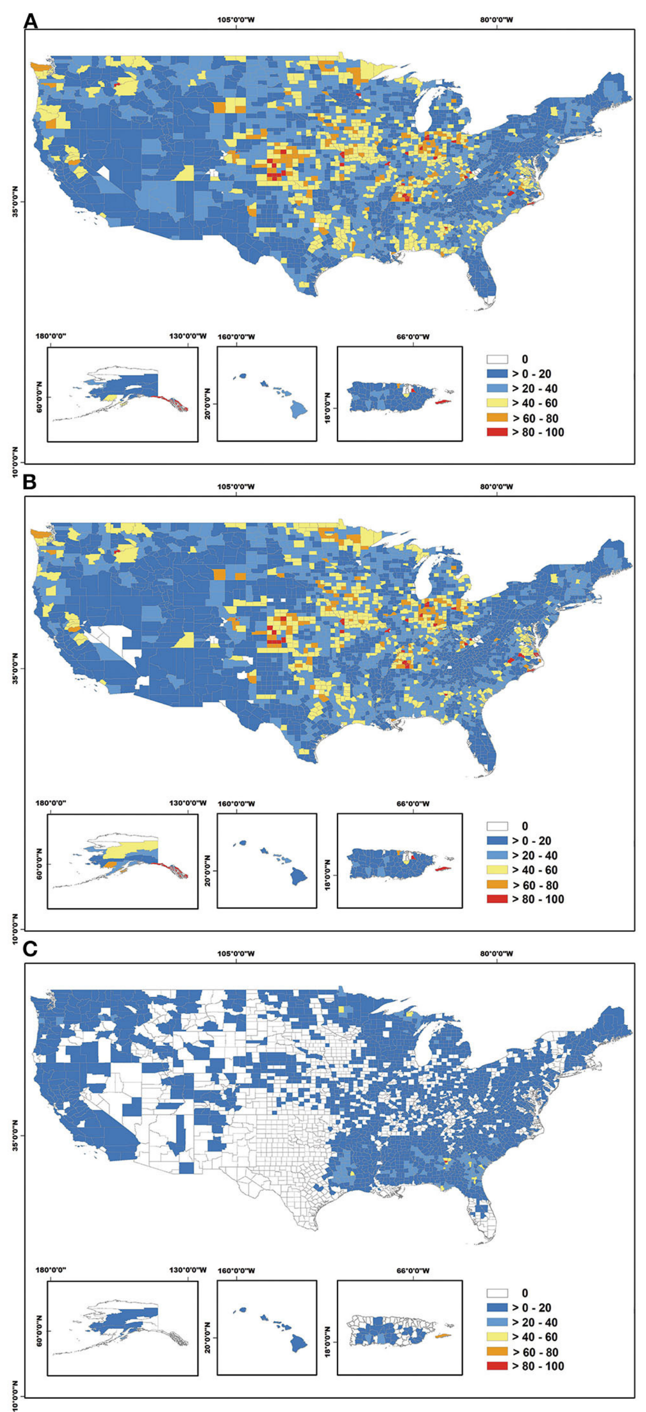

FIGURE 5 | Choropleth maps representing the percentage of total EQIP investments from 2009 to 2018 dedicated to practices that were categorized as having the greatest potential to improve soil health, transition to ecologically-based management, and expand adaptive strategy that were also executed *in-situ* on managed lands. **(A)** depicts the percent of total EQIP investments dedicated to *Level 3*, **(B)** depicts the percent of total EQIP investments dedicated to *Quadrant 4*, and **(C)** depicts the percent of total EQIP investments dedicated to *Transformation* practices.

This multi-dimensional classification system could be useful to the assessment of other existing agricultural conservation programs and could also inform strategic implementation of new public and private conservation initiatives. These frameworks can contribute to more informed discussions about the intent and effect of conservation investments. Future iterations of this approach could be applied to other conservation programs, such as CRP and CSP, provided comparable practice implementation data and practice descriptions become available.

We recognize that conservation can be a complicating factor to agronomic management, which is often cited by producers innovating with cover crops as a barrier to their wider use (Roesch-McNally et al., 2018). However, we argue that in most cases, conservation practices may also be viewed as having a positive impact on production. We reject the idea that practices must be an "either or" scenario and that production and profitability benefits do accrue from the outcomes-based practices outlined in this analysis, including examples such as increasing infiltration with more perennially-based agroecosystems (Basche and DeLonge, 2019), converting unprofitable land to an alternative use with perennial crops (Brandes et al., 2016), and improving weed suppression, productivity and yield stability with diverse crop rotations (Davis et al., 2012; Monast et al., 2018; Weisberger et al., 2019; Bowles et al., 2020).

Enhancing the Adaptive Capacity of Agriculture Through Transformational Change

Our analysis suggests that conservation practices that promote soil health by reducing disturbance and increasing biodiversity (*Quadrant 4*) and that those that fall into the system redesign agroecology level categories (*Level 3*) are the same practices that promote *Transformative* adaptation in agriculture in the adaptive strategy classification. The overlapping benefits described in all of the classifications underpin how transformational changes in agriculture can support better management of ongoing and future climate risks. The ability of a farm to respond to challenges or take advantage of opportunities related to climate risk occurs within an operating context consisting of the limits imposed by local ecological, social and economic realities (Walthall et al., 2013; Lengnick, 2014). The multi-dimensional classification system recognizes that expanding this operating context through increasing biodiversity, improving soil health and recoupling biological cycles creates more resilience in changing climate (Kates et al., 2012; Webb et al., 2017). It is important to recognize that productivity gains in agriculture achieved over the last several decades have occurred within a relatively stable climate period that we cannot expect to continue (Hatfield et al., 2014). While much of these productivity gains have been achieved through a focus on incremental improvements that might be described as efficiency or resistance approaches, increasingly, shifting to an emphasis on natural resource conservation, biological diversification and soil health to adapt to climate

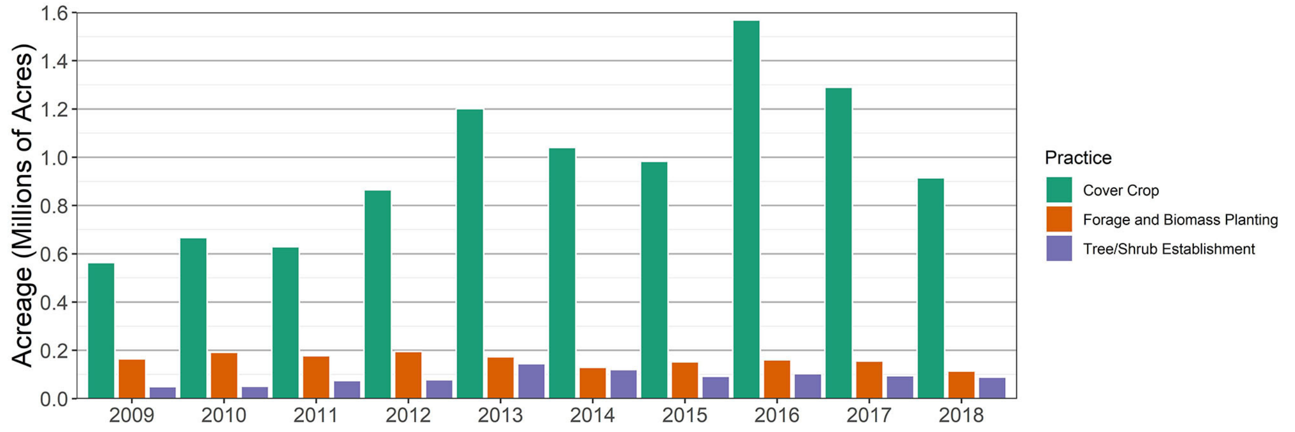

FIGURE 6 | EQIP acreage dedicated to cover crops (green; 340), forage and biomass planting (orange; 512), and tree/shrub establishment (purple; 612) from 2009 to 2018. We focused on these practices because they represented larger percentages of investments dedicated to top categories in our classification system (section Classification Systems Identify Overlapping, Complex Practices With the Largest Potential for Improving Environmental Health, **Table 3**) and were consistently reported in acres.

change is recognized by national and international assessments as a critical adaptation strategy (Janowiak et al., 2016; FAO, 2018; Gowda et al., 2018; HLPE, 2019). This shifting emphasis recognizes that transformative, ecologically-based strategies can improve ecosystem function; for example, improving soil health in order to reestablish biological relationships that occur naturally on the farm rather than simplifying them (such as water and/or nutrient cycling as well as pest management), can lead to increased profitability and productivity (LaCanne and Lundgren, 2018; Rosenzweig et al., 2018a,b). Overall, the practices highlighted recognize the multitude of co-benefits afforded by select conservation programs with opportunity to improve multiple aspects of environmental health.

Our analysis found that conservation investments made through the EQIP program are dominated by practices that promote input efficiencies (*Level 1*) and resistance strategies (those that increase adaptive capacity without change to structure and function of the existing agroecosystem). Soil health investments were about evenly divided between investments that promote changes in physical (*Quadrants 1* or *2*) vs. biological aspects of soil health (*Quadrants 3* or *4*). Considered together, EQIP investments are dominated by practices that promote incremental adaptation and offer limited support for conversion to ecologically-based management, or *Transformative* adaptations. While incremental adaptation has contributed to the persistence of agricultural production systems over time and through changing environmental and social conditions, and an overemphasis on incremental adaptation can promote maladaptation (Kates et al., 2012; Janowiak et al., 2016) and lead into an adaptation trap (Walthall et al., 2013).

There were differences in how cover crops were classified amongst our top categories, which impacted the range of results given that cover crops are one of the top ten EQIP funded practices. Although Gliessman (2014) considers cover crops a *Level 2* practice, in our expertise, we see that many innovative cover crop producers are utilizing the practice as an approach to diversify farms and reduce reliance on inputs, which we believe represents the description more of a *Level 3* than *Level 2* practice. In fact, innovative producers often describe their management system with cover crops as a "whole-system" (Basche and Roesch-McNally, 2017). We did not, however, believe that cover crops represent a *Transformation* practice where they fundamentally alter the structure and function of an agroecosystem.

Limitations of the Framework and Available Data

There were a few limitations to our analysis and ability to categorize practices, including the context under which practices are implemented as well as availability of detailed practice information. As a research team, we recognized that context is critical for making many important determinations of the impact of agricultural management including conservation. Where possible, we did our best to assume the most positive possible outcome of a practice and inferred context from the information available in the USDA-NRCS conservation practice descriptions. That is, some practices may indirectly impact soil health, but could be combined with other practices ("stacked") to provide a direct impact on soil health especially if implemented in a particular way (Tully and McAskill, 2020). For example, a fence alone does not directly impact soil health, but when used in combination with rotational grazing, could improve soil health outcomes. As we evaluated each practice individually, we did not have information of additional context for how it was implemented or combined with other practices. Future efforts could include looking more strategically at USDA-NRCS conservation plans or contract data, as they offer insights on how conservation practices evolve and are stacked when implemented on the farm. We recognize that it is critical to consider how practices might be utilized together or stacked to provide environmental benefits both on- and off-farm, however it is difficult to impossible to acquire comprehensive conservation practice data at a national scale. This would better match how producers implement and evolve conservation practices on their operations. Furthermore, quantifying the benefits from stacking practices at different scales will help improve soil health more holistically. As a result, we recognize that our estimates of dollars spent to achieve particular outcomes are not all encompassing and do not include as explicit a focus on some of the environmental co-benefits (reducing greenhouse gas emissions from agriculture, for example).

We do not discount the value of certain *ex-situ* (practices not conducted directly on the managed land) efforts that could positively address soil health (e.g., a conservation *plan* designed to address soil health resource concerns). However, the goal of this analysis was to focus on investments with the most direct and greatest potential to improve environmental health, and therefore would argue that *ex-situ* practices have an indirect rather than direct impact.

Finally, during the process of this work we found that many practice descriptions were simply unavailable or did not contain enough information to be able to classify into all of our categories. We could not find comprehensive information online for the interim practices (88 practices), and there were another 36 practices that were listed in our public data request but descriptions were not available. Although, we did consult state standards documents that were provided to us to see if information on practices was available there rather than the national catalog. However, we ultimately decided that if practice descriptions were not available through the National Standards catalog, we would not include them in the analysis, in order to create an analysis that was more national in scope. Our understanding is that the National Standards catalog is updated annually, and we would encourage comprehensive, publicly-available information continue to be made available about practices within the conservation programs.

Future Research and Conservation Program Implementation

Future research could utilize this framework in combination with a "hotspot" analysis to detect if specific conservation practices are geographically clustered or co-located. Such spatial analyses could further determine if practices are "stacked" and provide more context for soil and environmental health outcomes, as well as understanding of facilitators of wider practice use. Interest in implementing climate solutions has increased at a federal level and recent reports and legislation such as the 2020 Senate Climate Crisis Report, the Senate's Growing Climate Solutions Act of 2020, the 2020 House of Representatives Agriculture Resilience Act highlight a critical need for policy-makers to identify practices that offer climate change mitigation and adaptation co-benefits. Our framework, which includes classifications for soil health, environmental health and resilience, could be implemented to identify practices with co-benefits to support future program development. Further, the framework could be used at a state- or even international-level for evaluating conservation programs. For example, states such as Maryland, which have a recent history with implementing large conservation programs to address resource concerns, could utilize such a framework to identify those practices most effective at meeting program goals. Overall, this framework has potential to serve as a useful tool for new development of policies, and could further be utilized to shift or reprioritize current programs. This framework offers researchers and policy-makers a useful new tool to re-evaluate the long-standing emphasis in U.S. agriculture on technological risk management strategies to develop policies and programs designed to capture the benefits of ecosystem-based solutions that afford climate resilience to farms, communities and the planet.

CONCLUSIONS

There is a need to critically evaluate how conservation dollars are being spent, particularly with the potential of increased investments in the future given expanded interest in negating environmental impacts from agriculture. We used a multi-dimensional classification system to evaluate the potential for conservation practices and funding within EQIP to improve soil health, transition to ecologically-based management, and expand adaptive strategy. From 2009 to 2018, we found that there was limited investment in those practices that have the greatest potential to improve these aspects of environmental health, representing ∼2–27% of the program's expenditures, or less than one-tenth of 1% of USDA's annual budget ($13–121 million out of $144 billion in 2018). This multi-dimensional approach allowed us to determine where classifications converged and identify those practices that have the potential to achieve multiple improvements in soil and environmental health, including ∼28 practices fulfilling multiple of our top criteria and nine practices that fulfilled all of them. These practices represent diversified crop, livestock, and forestry management including silvopasture and other agroforestry practices, wetland creation and restoration, and wildlife habitat management. The potential of these diversified, complex practices to improve multiple aspects of environmental health underscores the need for investments to prioritize transformational changes in agriculture to better support management of ongoing and future climate risks. This analysis provides evidence that there is tremendous untapped potential for conservation programs to confer greater environmental health in U.S. agriculture. This framework could provide a model for how new policies are designed and possibly in shifting or reprioritizing how current programs are implemented.

DATA AVAILABILITY STATEMENT

The complete dataset analyzed in this paper will be publicly available in the USDA Forest Service Research Data Archive, at the following url: https://doi.org/10.2737/RDS-2020-0076.

AUTHOR CONTRIBUTIONS

AB, KT, NÁ-B, JR, LL, JM, TB, RS, and GR-M conceptualized the paper. AB, KT, LL, JM, and TB conceptualized the classifications. AB, KT, and LL categorized the practices and wrote the first draft of the paper. AB, NÁ-B, and JR performed the primary data analysis. LJ created the illustrations and diagrams. AB, KT, NÁ-B, JR, LL, JM, TB, RS, GR-M, and LJ edited drafts of the paper. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.547876/full#supplementary-material

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Maize-Pigeonpea Intercropping Outperforms Monocultures Under Drought

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There is an urgent need to develop resilient agroecosystems capable of helping smallholder farmers adapt to climate change, particularly drought. In East Africa, diversification of maize-based cropping systems by intercropping with grain and tree legumes may foster productivity and resilience to adverse weather conditions. We tested whether intercropping enhances drought resistance and crop and whole-system yields by imposing drought in monocultures and additive intercrops along a crop diversity gradient—sole maize (*Zea mays*), sole pigeonpea (*Cajanus cajan*), maize-pigeonpea, maize-gliricidia (*Gliricidia sepium*, a woody perennial), and maize-pigeonpea-gliricidia—with and without fertilizer application. We developed and tested a novel low-cost, above-canopy rainout shelter design for drought experiments made with locally-sourced materials that successfully reduced soil moisture without creating sizeable artifacts for the crop microenvironment. Drought reduced maize grain yield under fertilized conditions in some cropping systems but did not impact pigeonpea grain yield. Whole-system grain yield and theoretical caloric and protein yields in two intercropping systems, maize-pigeonpea and maize-gliricidia, were similar to the standard sole maize system. Maize-pigeonepea performed most strongly compared to other systems in terms of protein yield. Maize-pigeonpea was the only intercrop that consistently required less land than its corresponding monocultures to produce the same yield (Land Equivalent Ratio >1), particularly under drought. Despite intercropping systems having greater planting density than sole maize and theoretically greater competition for water, they were not more prone to yield loss with drought. Our results show that maize-pigeonpea intercropping provides opportunities to produce the same food on less land under drought and non-drought conditions, without compromising drought resistance of low-input smallholder maize systems.

Keywords: resilience, land equivalent ratio, drought, pigeonpea, gliricidia, maize, agroforestry, diversification

INTRODUCTION

Climate change and weather variability already affect farming conditions across sub-Saharan Africa, and the vulnerability of farmers, agricultural production, and food security will only increase in the future. A significant portion of smallholder farmer households in Tanzania (40%) have been negatively impacted by drought in the past 5 years (Reincke et al., 2018). In the future, seasonal temperature increases of 2°C for 2050 are predicted to cause yield losses of 13% for maize (*Zea mays)*, with additional 4–7% reductions in yields due to higher intra-seasonal rainfall variability in this region (Rowhani et al., 2011). With maize contributing 21–57% of total daily calorie supply in East Africa (Krivanek et al., 2007), there is an urgent need for adoption of drought-resilient agricultural management practices in the maize-based cropping systems predominant across East Africa.

Diversification of maize cropping systems, both in time and space, provides opportunities to decrease vulnerability and improve drought resilience through ecological intensification and production of more diverse food products (Lin, 2011; Altieri et al., 2015; Bullock et al., 2017; Degani et al., 2019; Steward et al., 2019). A resilient agroecosystem shows greater interannual yield stability due to higher resistance to stress or faster recovery after stress (Urruty et al., 2016; Peterson et al., 2018). Intercropping can enhance land use efficiency (food production per unit area) (Rusinamhodzi et al., 2012, Yu et al., 2015) and foster resilience through a portfolio effect whereby different plant species in mixtures have differential drought responses (Doak et al., 1998; Tilman et al., 1998; Tilman, 1999) and altering multiple plant and soil interactions regulating crop performance under drought. In smallholder cropping systems, intercropping C4 cereals with grain and tree legumes has been shown to positively impact soil carbon, fertility, infiltration, and moisture (Jackson et al., 2000; Makumba et al., 2006; Chirwa et al., 2007; Rusinamhodzi et al., 2012; Muchane et al., 2020). Intercropping can also alter plant traits involved in water acquisition and status such as root distribution (Makumba et al., 2009), the depth of plant water sourcing (Sekiya and Yano, 2004), and leaf water potential (Harris and Natarajan, 1987). Crop diversity has been shown to increase the stability of food production at the national and district scales, including to rainfall deficits, but testing of such a portfolio effect at the field scale has been limited (Birthal and Hazrana, 2019; Renard and Tilman, 2019). Together, these mechanisms and evidence suggest that field-scale diversification represents an underexplored opportunity for building resilience in low-input smallholder systems.

Whether these shifts in soil and plant processes and plant mixture composition translate into greater yield stability under drought has seldom been empirically demonstrated in low-input smallholder cropping systems. One study showed that sorghum-groundnut intercropping with a replacement design (i.e., same planting density in monoculture and intercropping) becomes progressively superior to monocultures under increasing drought (Natarajan and Willey, 1986). However, the drought response of additive intercropping (i.e., greater planting in intercropping than monoculture) remains unclear. One study in Malawi found that maize yield loss due to drought in no-till additive maize-cowpea intercropping systems was similar to or greater than in sole maize (Steward et al., 2019). Fully additive designs (planting density in intercropping is the sum of densities of monocultures) are nearly always superior to monocultures in terms of their productivity whereas replacement designs are superior to monocultures only about half the time (Yu et al., 2015), making an evaluation of the drought response of additive designs necessary given their importance for increasing smallholder food production. Although additive intercropping of maize with leguminous trees can enhance maize yield under less favorable conditions (Sileshi et al., 2011), higher competition for water with higher planting densities such as additive intercropping could also increase risk of yield loss (Lobell et al., 2014). Smallholder systems are often co-limited by resources, such as low soil fertility in addition to drought, which can further alter the relative advantage of intercropping (Sileshi et al., 2011; Rusinamhodzi et al., 2012). Most evidence to date on intercropping and drought is solely focused on yield performance, with an emphasis on maize. How current evidence scales up from maize yield to whole-system caloric or protein production from all crops (Snapp et al., 2010; Smith et al., 2016) needs to be assessed to evaluate the potential of intercropping to boost drought resilience of smallholder subsistence farming systems.

We investigated how intercropping impacts whole-system vulnerability to drought, in terms of crop and whole-system grain and nutritional yields and drought-induced yield losses, by excluding rainfall using rainout shelters at a field trial in semi-arid Tanzania. Rain exclusion systems have proven useful in assessing how agroecosystem resilience is affected by management practices in different agroecosystems (Degani et al., 2019; Steward et al., 2019). Such rainout shelters are particularly useful for testing practices with impacts that may emerge over years to even decades, and, as such, remain difficult to test at several sites along a rainfall gradient. We focused on smallholder maize systems in East Africa, where there are relatively high adoption rates of intercropping and tree planting on-farm (27–88 and 14–23% of households surveyed, respectively) in the last 10 years (Kristjanson et al., 2012). Maize intercropping with grain legumes such as pigeonpea (*Cajanus cajan*) or shrub/tree legumes such as gliricidia (*Gliricidia sepium*) have been among the most widely studied and adopted diversified systems by East African smallholder farmers (Garrity et al., 2010; Snapp et al., 2010). Pigeonpea is a drought-tolerant grain crop with deep early-season taproot development and slow initial shoot growth (Snapp et al., 2003). Gliricidia trees have been shown in a long-term trial to increase soil organic matter, soil fertility, and soil moisture at the end of the rainy season (Makumba et al., 2006). We measured the impact of additive intercropping of maize with pigeonpea and gliricidia on crop and whole-system grain, calorie, and protein yields and drought resistance with and without fertilizer. We hypothesized that intercropping would outperform sole cropping in food production across rainfall levels but most strongly under drought, with greater land use efficiency and drought resistance, especially when fertilized.

MATERIALS AND METHODS

Study Site

The experiment was conducted at an ongoing field trial established in 2015 in Manyusi village, Kongwa District, Dodoma, Tanzania (5° 33′ 56.16123" S, 36° 17′ 29.85319" E, elevation 1206.6 m, **Supplementary Figure 1**). Prior to establishment of the research trial, the site was under continuous maize cultivation by the landowner, a smallholder farmer. Like in other smallholder systems in the semiarid tropics, there was not fertilizer addition at this site prior to establishment of the trial. Tillage consisted of plowing with oxen for field preparation and weeding by hand hoe. The soil was an acidic (pH 5.16 ± 0.20) loamy sand with low organic carbon (0.54 ± 0.16%) (**Supplementary Table 1**) and minimal slope (1–2%) and classified as a chromic luvisol (Trans-SEC, 2017). This land management history and level of land degradation provide an opportunity to test the resilience potential of maize-legume intercropping. The trial was located in a semi-arid climate with a 30-year annual rainfall average of 635 mm (cropping season average of 425 mm) with a unimodal pattern with most rainfall between December and April and a dry period of 6–7 months (Shemsanga et al., 2016) (**Figure 1A**). Long-term historic (1988–2018) rainfall estimates for Manyusi village were downloaded from the Early Warning eXplorer (EWX) Lite time series database and web-based mapping tool (United States Geological Survey, 2020), and are similar to but slightly higher than average annual rainfall for the Dodoma region (589 mm) (Msongaleli et al., 2017).

Experimental Design

The field trial was established in a randomized complete block design (RCBD) with three replications in 2015. Because of the presence of gliricidia, a woody perennial, in some plots, tillage by plowing with oxen for field preparation and weeding by hand hoe—standard tillage practices in the study area—were used throughout the experimental period, except for the initial site preparation which was done by tractor. Five cropping systems, sole maize (M), sole pigeonpea (P), maize-pigeonpea (MP), maize-gliricidia (MG), and maize-gliricidia-pigeonpea (MGP), were randomly assigned to plots (16 × 16 m). Sole maize represents the standard farmer practice for growing maize in the study area. Intercropping maize with other annual crops like sunflower, groundnuts, and bambara groundnut is a traditional practice in the study area similar to other smallholder subsistence farming systems in East Africa. Gliricidia woodlots and gliricidia and pigeonpea integration into maize systems are a subject of ongoing research in the study area. Pigeonpea is a more common crop elsewhere in Tanzania. Gliricidia seedlings were transplanted at the establishment of the trial in 2015 at a spacing of 4 × 4 m (25 trees per plot or 625 trees per hectare). Beginning in 2016, gliricidia was pruned heavily to 50 cm height twice a year during the cropping season, once before seeding of maize and pigeonpea in January and once during maize vegetative growth. Green foliage was distributed evenly across its plot of origin as green manure and incorporated into the soil via cultivation by oxen at the first pruning and by hand hoeing at the second pruning. Each year, about 1 month after the onset of the rainy season and immediately following cultivation with oxen to prepare the land and incorporate gliricidia foliage, maize and pigeonpea were sown by hand at a spacing of 75 cm between rows and 60 cm within rows (**Supplementary Figure 2**). Three maize seeds or three pigeonpea seeds were sown per planting hill and thinned to two plants per hill during maize vegetative growth for a planting density of 44,444 plants per hectare in sole maize and sole pigeonpea. Intercropping was additive: pigeonpea

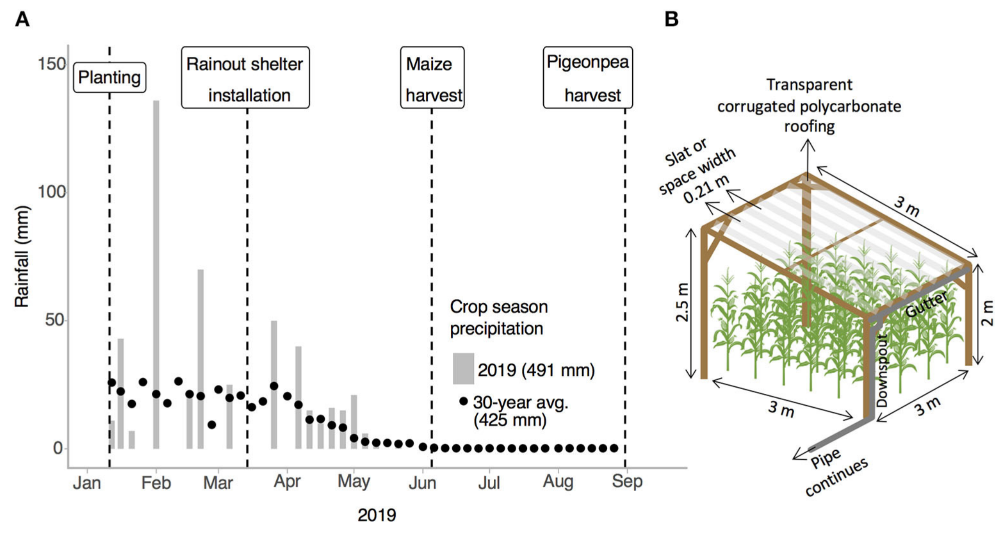

FIGURE 1 | (A) Drought experiment timeline and cropping season rainfall for the experiment and 30-year average (1988–2018). Total cropping season rainfall is indicated in parentheses in the figure legend. **(B)** Diagram of rainout shelter designed to intercept 50% of incoming rainfall. The pipe drainage system carries water to the field border. Shelters were bordered by a short (40 cm) bund of field soil to prevent surface runoff into the drought plots during rainfall events. Post burial depth (not pictured) was 0.5 m. Plant density not to scale (2 plants per hill). Photographs of shelters are provided in **Supplementary Figure 3**.

rows with a row spacing of 75 cm were sown between maize rows also with a row spacing of 75 cm (**Supplementary Figure 2**). The planting density of each crop individually was the same in intercrop and sole crop plots. The total planting density in intercrop plots was thus double that of sole crop plots for an intercrop planting density of 88,889 plants per hectare. In 2019, when the study we report here took place, the maize cultivar was Staha, the pigeonpea cultivar was ICEAP 0040, and the average soil moisture at planting was 7.92 g water g^{-1} soil.

In 2017, the experimental design was modified to a split-plot. Each cropping system main plot was divided into sub-plots (8 × 16 m) and randomly assigned one of two levels of fertilization: unfertilized or fertilized. Not applying fertilizer represents the standard farmer practice in the study area. Fertilization consisted of 72 and 100% of recommended rates for maize monocrop of nitrogen (N) and phosphorus (P), respectively. Soil potassium levels at the study site are non-limiting to crop production (**Supplementary Table 1**) (Landon, 2014). Starter fertilizer was broadcast at seeding and 13.4 kg N ha^{-1} and 15.0 kg P ha^{-1} were applied as diammonium phosphate (18-46-0). Side dress fertilizer was banded at the soil surface during maize vegetative growth and 30 kg N ha^{-1} were applied as urea (46-0-0), for a season total of 43 kg N ha^{-1}. Plots with gliricidia also received organic nutrients from gliricidia green foliage prunings incorporated into the soil at maize/pigeonpea seeding and during maize vegetative growth. The estimated gliricidia foliage production for the 2019 cropping season, when the study we report here took place, was 2.27 tons ha^{-1} in maize-gliricidia and 1.60 tons ha^{-1} in maize-pigeonpea-gliricidia (dry weight). Based on assumed N concentration of gliricidia foliage (Kimaro et al., 2008), crops received an additional 69.3 kg N ha^{-1} in maize-gliricidia and 48.8 kg N ha^{-1} in maize-pigeonpea-gliricidia. However, actual availability of N from gliricidia foliage incorporation and other pathways depends on multiple factors including soil moisture and decomposition rates that are beyond the scope of this experiment.

In 2019, the experimental design was modified to add water treatments. Each cropping system-fertilization combination was split and randomly assigned one of two levels of rainfall water inputs: ambient rainfall and drought (50% ambient rainfall), creating a split-split plot (3 × 3 m total area including borders). To simulate drought in the field, we designed a novel above-canopy partial rain exclusion system (**Figure 1B**, **Supplementary Figure 3**) adapted from previous designs. We combined the tall (≥2 m) stature of rainout shelters used to simulate drought in maize (Steward et al., 2019) with a slatted roof that intercepts rainfall and minimizes side effects on the crop microenvironment, as shown in longer term (>1 month) drought simulations in grassland and desert ecosystems (Yahdjian and Sala, 2002; Gherardi and Sala, 2013). Rainout shelters were 3 m wide × 2.96 m long × 2 and 2.5 m tall at their shortest and tallest heights, respectively. Roof slats were 3 m long × 0.21 m wide and cut from transparent corrugated polycarbonate roofing material. Slats were spaced every 0.42 m, such that the roof was designed to intercept 50% of incoming rainfall. Rainfall intercepted by rainout shelters was collected in gutters along the lower edge of the shelter roof and diverted by gravity flow to the edge of the field via a connected system of PVC pipes. Shelters were oriented with the tallest side to the northeast (i.e., the roof sloping down to the southwest) to maximize direct solar radiation from the north to crops beneath the rainout shelter and minimize indirect radiation passing through the roof to crops (Yahdjian and Sala, 2002). Rainout shelters were bordered by a short (40 cm) bund of field soil to prevent surface runoff into the drought plots during rainfall events. The shelters were installed at the onset of maize tasseling and maintained through the harvests of maize and pigeonpea (**Figure 1A**), in order to simulate drought during maize anthesis and grain filling, the growth stages most vulnerable to drought (Grant et al., 1989; Monneveux et al., 2006).

Weeds were removed with hand hoes, the standard farmer practice for weed management in the study area, twice, during maize vegetative and early reproductive growth. To manage fall armyworm in maize, an insecticide, Acetamiprid + Emamectin benzoate, was applied twice to all plots using a backpack sprayer, at late maize vegetative stage and at tasseling according to manufacturer rates.

Crop Environment Monitoring

Rainfall during the cropping season was measured using a rain gauge located in a border area between plots at 1 m height and recorded manually daily.

All data were collected in the center 2 × 2 m area of each plot. Temperature and relative humidity were measured and recorded at the height of the top of the crop canopy (slightly <2 m, the shortest height of the rainout shelter roof) with a Tramex DL-RHTA FeedBack Datalogger (Tramex Ltd, Orlando, FL, USA) on one date during maize grain filling at midday. The saturated partial pressure of water in the air was calculated from air temperature using the Buck equation (Buck, 1981) and used with relative humidity measurements to calculate actual partial pressure and vapor pressure deficit.

Photosynthetically Active Radiation (PAR) received by the top of the crop canopy was measured in all plots with and without shelters using an Accupar LP-80 ceptometer (Meter Group, Pullman, WA, USA/München, Germany) at one date during maize grain filling at midday with mixed cloudy light conditions typical of other data collection dates. The fraction of PAR transmitted (f_{TPAR}) by the rainout shelter roofing was calculated as the ratio of PAR under a rainout shelter to PAR in the paired ambient rainfall plot.

Soil moisture during the drought imposition period was measured by sampling soil (0–20 cm) at four random points within each sub-sub-plot and compositing subsamples. A subsample taken from the mixed, composited subsamples was analyzed at Sokoine University of Agriculture Department of Ecosystems and Conservation laboratory (Morogoro, Tanzania) for gravimetric water content.

Crop and Whole-System Yields

At maize physiological maturity, maize grain was harvested from the inner 1.5 × 1.8 m (2.7 m^2) plot areas. Total fresh grain weight was recorded, and subsamples were analyzed at Tanzania Coffee Research Institute (Moshi, Tanzania) for moisture content and dry matter, which were used to extrapolate fresh weight per plot area to dry yield per ha. Maize dry grain yield is reported at 12% moisture content. At pigeonpea physiological maturity, pigeonpea grain yield was harvested, recorded, subsampled, and analyzed for moisture content and dry matter which were used to calculate dry yield per ha. Pigeonpea grain yield is reported as dry grain yield (0% moisture content). Whole-system grain yield is reported as the sum of dry grain yields (0% moisture content) of maize and pigeonpea.

The land equivalent ratio (LER), a relative measure of the sole cropping area compared to the intercropping area required to achieve the same total crop production, was calculated as:

$$\mathbf{LER}=\frac{\mathbf{I_1}}{\mathbf{S_1}}+\frac{\mathbf{I_2}}{\mathbf{S_2}}$$

where I_1 and I_2 are the yields of species 1 and 2, respectively, in intercropping, and S_1 and S_2 are the yields of the species in sole cropping. The LER was calculated for maize and pigeonpea, the two cropping system outputs used directly for human consumption in the study area. For maize-pigeonpea and maize-pigeonpea-gliricidia, the LER was calculated relative to both sole maize and sole pigeonpea (i.e., I_1/S_1, I_2/S_2). For maize-gliricidia, LER was calculated relative to sole maize (i.e., I_1/S_1). We focused our LER calculation on cropping system outputs used directly for human consumption because (1) biomass for use as animal forage was not measured consistently for all crops, and (2) the field trial does not include a sole gliricidia treatment and therefore precludes including gliricidia fuelwood impacts on LER.

For all cropping systems, maize and pigeonpea grain yields per hectare were converted to theoretical calories and protein produced per hectare using published constant conversion factors of dry grain weight to calories or protein specific to Tanzania: 362 kcal 100 g^{-1} maize, 8.1 g protein 100 g^{-1} maize, 343 kcal 100 g^{-1} pigeonpea, and 21.7 g protein 100 g^{-1} pigeonpea (Lukmanji et al., 2008). For a given cropping system, calories or protein from both maize and pigeonpea were then summed to calculate total theoretical calorie or protein yield per hectare for the cropping system.

We focused on resistance to drought (smaller fluctuation from non-stress levels) as one aspect of system resilience (Peterson et al., 2018). Drought resistance was calculated as absolute (drought—ambient rainfall) change due to drought in yields of maize grain, pigeonpea grain, whole-system grain, calories, and protein for each cropping system-fertilization treatment replicate. Greater drought resistance indicates less change due to drought.

Statistical Analysis

All statistical analyses were conducted using R version 3.6.2. Linear mixed-effects models with cropping system, fertilization, and water as fixed effects, and block, main plot, sub-plot, and their interactions with treatments as random effects were used to test the effect of treatments on all response variables (*lmer()* command in *lmerTest* package (version 3.1-1) (Kuznetsova et al., 2017) in R), except for f_{TPAR} (see below). In the case of multiple sampling events (gravimetric water content), a date fixed effect, a sub-subplot random effect, and respective interaction terms were added to the model. For each response variable, the full model based on experimental design was fit and then reduced by eliminating random effects that accounted for zero variance to avoid overfitting warnings. The fullest model representative of experimental design that did not result in overfitting was used for analysis of variance (ANOVA). In cases where all random effects accounted for zero variance and mixed-effects models resulted in overfitting, random effects were removed from the model, and linear fixed-effects models were fit [*lm()* command in *stats* package (version 3.6.2)]. The assumptions of homogeneity of variance and normality of residuals were assessed visually using diagnostic plots and quantitatively [*shapiro.test()* command in *stats* package (version 3.6.2); *leveneTest()* command in *car* package (version 3.0-6) Fox and Weisberg, 2019]. Response variables were transformed as necessary to meet assumptions. ANOVA [*anova()* command in *stats* package (version 3.6.2)] and means comparisons (*CLD()* command and contrasts [*emmeans()* command) in *emmeans* package (version 1.4.3.01) Lenth, 2019] were conducted using Satterthwaite's method to approximate the degrees of freedom. In cases of multiple comparisons, the Tukey method of p-value adjustment was used to compare families of multiple (three or more) estimates with a significance level of alpha = 0.05. For contrasts testing the effect of the water treatment within each cropping system-fertilization(-date) combination, the Bonferroni method of p-value adjustment was used to simultaneously conduct multiple tests. To test whether the fraction of PAR transmitted (f_{TPAR}) through the rainout shelter roof slats was <1 (i.e., a null hypothesis that shelters transmit 100% of PAR), a one-sided t-test was conducted with a confidence level of 0.95 [*t.test()* command in *stats* package (version 3.6.2)]. Observations from one sub-plot where an underground termite nest significantly and visibly affected soil structure and plant growth were excluded from analyses for all crop response variables.

RESULTS

Soil Moisture and Crop Microenvironment

No effects of the rainout shelters were detected at the top of the crop canopy on air temperature ($p = 0.460$), relative humidity ($p = 0.141$), or vapor pressure deficit ($p = 0.658$) (**Figures 2B–D**). The mean fraction of transmitted PAR through the rainout shelter roof (f_{TPAR}) was 84.2% (15.8% reduction in PAR by rainout shelter) ($p < 0.001$) (**Figure 2E**). On a cloudy day typical of light conditions during data collection dates (maize anthesis through harvest), average light transmission through rainout shelters was 850 vs. 1,012 μmol m^{-2} s^{-1} without rainout shelters (**Supplementary Figure 4**).

The drought treatment using rainout shelters reduced gravimetric soil moisture by 12.5% on average ($p = 0.015$) with the magnitude of the water effect depending on cropping system, fertilization (cropping system:fertilization $p = 0.011$, cropping system:fertilization:water $p = 0.034$), and date (water:date $p < 0.001$) and varying in whether it was significant (**Figure 2A**). Reductions in soil moisture by rainout shelters was more often significant at the second sampling date after drought imposition. Gravimetric soil moisture was significantly affected by cropping

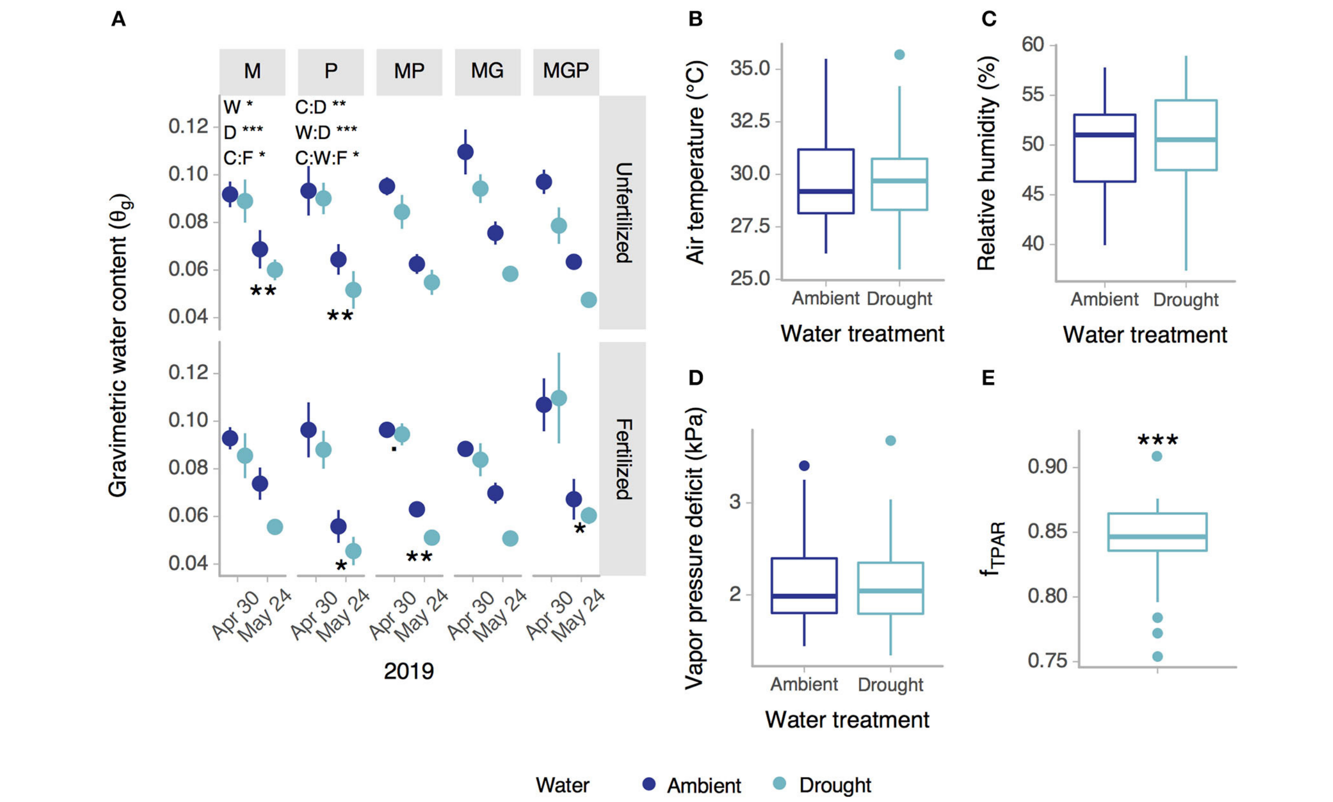

FIGURE 2 | Effects of rainout shelters on **(A)** gravimetric water content (0–20 cm), **(B)** air temperature, **(C)** relative humidity, **(D)** vapor pressure deficit, and **(E)** the fraction of transmitted photosynthetically active radiation (f_{TPAR}), **(A)** within or **(B–E)** across cropping system-fertilization combinations. Cropping system abbreviations: M, Maize; P, Pigeonpea; MP, Maize-pigeonpea; MG, Maize-gliricidia; MGP, Maize-gliricidia-pigeonpea. Treatment factor abbreviations: W, Water; C, Cropping system; F, Fertilization; D, Date. Asterisks indicate significant differences between ambient and drought (***$p < 0.001$, **$p < 0.01$, *$p < 0.05$). Error bars = standard error; **(A)** $n = 3$ or **(B–E)** $n = 30$.

system and fertilization with variation across dates (cropping system:date $p = 0.002$) (**Figure 2A**), but there was no evidence of higher soil moisture due to intercropping (**Figure 2A**).

Crop Yields and Drought Resistance

Mean maize grain yield by treatment combination ranged from 1.39 to 7.08 t ha^{-1} (12% moisture content) and was interactively affected by water level ($p = 0.049$) and fertilization ($p < 0.001$) across cropping systems (cropping system:water:fertilization $p = 0.006$) (**Figure 3A**, **Supplementary Table 2**). Under fertilized conditions, maize grain yield was significantly reduced by the drought treatment in sole maize and maize-pigeonpea intercropping but not in maize-gliricidia or maize-gliricidia-pigeonpea intercropping. Under unfertilized conditions, maize grain yield was not significantly by drought in any cropping system. Maize drought resistance (yield loss to drought) varied with cropping system and fertilization (fertilization $p = 0.015$, cropping system:fertilization $= 0.003$) (**Figure 3B**). Under fertilized conditions, maize drought resistance was significantly lower in sole maize and maize-pigeonpea intercropping than in maize-gliricidia-pigeonpea intercropping (**Figure 3B**). Under unfertilized conditions, maize drought resistance did not vary by cropping system.

Mean pigeonpea dry grain yield varied by cropping system and ranged from 0.39 to 1.09 t ha^{-1} (**Figure 3C**). It decreased significantly from sole pigeonpea to maize-pigeonpea-gliricidia (cropping system $p = 0.028$) but was not impacted by drought or whether or not fertilizer was applied (**Figure 3C**, **Supplementary Table 2**). Pigeonpea drought resistance was not significantly affected by cropping system and fertilization treatments (cropping system $p = 0.713$; fertilization $p = 0.737$; cropping system:fertilization $p = 0.196$) (**Figure 3D**).

Whole-System Grain and Nutritional Yields and Drought Resistance

Whole-system dry grain yields ranged from 0.91 to 6.23 t ha^{-1} and were interactively affected by water ($p = 0.003$), cropping system ($p = 0.001$), and fertilization ($p = 0.002$, water:fertilization $p = 0.047$, water:cropping system:fertilization $p = 0.002$) (**Figure 4A**, **Supplementary Table 2**). Under fertilized ambient rainfall conditions, whole-system yield was significantly higher in sole maize and maize-pigeonpea intercropping than sole pigeonpea and maize-gliricidia-pigeonpea intercropping. Under fertilized drought conditions, whole-system yield did not vary significantly between cropping systems except for

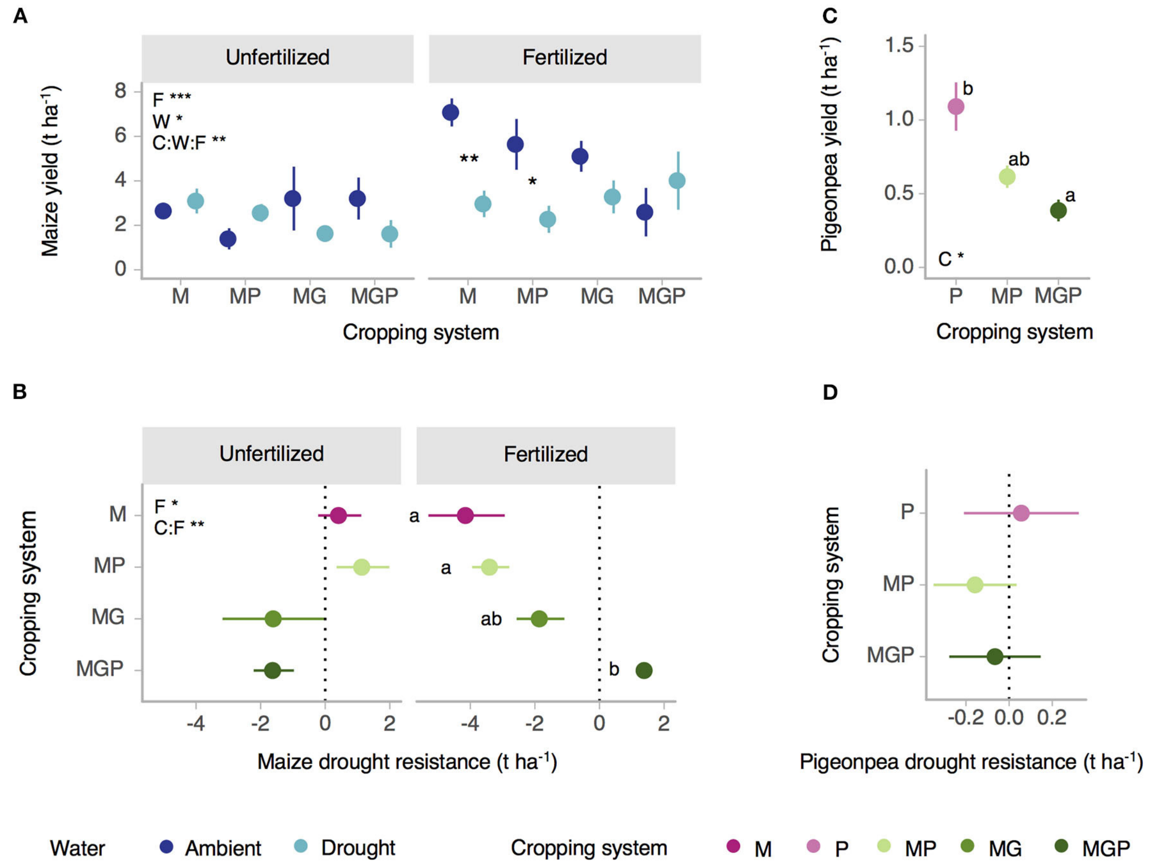

FIGURE 3 | Effects of water, cropping system, and fertilization on **(A)** maize grain yield at 12% moisture and **(C)** pigeonpea dry grain yield. Effects of cropping system and fertilization on yield drought resistance of **(B)** maize and **(D)** pigeonpea. Dotted lines indicate high drought resistance (zero change due to drought). Cropping system abbreviations: M, Maize; P, Pigeonpea; MP, Maize-pigeonpea; MG, Maize-gliricidia; MGP, Maize-gliricidia-pigeonpea. Treatment factor abbreviations: W, Water; C, Cropping system; F, Fertilization. Asterisks indicate significance of treatment effects or significant differences between ambient and drought (***$p < 0.001$, **$p < 0.01$, *$p < 0.05$). Means sharing the same letter are not significantly different within a panel of a graph (alpha = 0.05). Error bars = standard error; **(A,B)** $n = 3$ (fertilized MGP $n = 2$) or **(C,D)** $n = 12$ (MGP = 10).

being greater in maize-gliricidia-pigeonpea intercropping than sole pigeonpea. Under unfertilized conditions, there were no differences in whole-system yields between cropping systems under either drought or ambient rainfall conditions. Whole-system grain yield drought resistance varied by cropping system (cropping system:fertilization $p = 0.002$) and fertilization ($p = 0.023$) (**Figure 4C**). Sole pigeonpea was the cropping system most consistently resistant to drought across fertilization levels. Under fertilized conditions, the whole-system drought resistance of sole maize and maize-pigeonpea intercropping was lower than that of sole pigeonpea and maize-gliricidia-pigeonpea intercropping. Under unfertilized conditions, cropping systems did not vary significantly in their whole-system drought resistance.

Theoretical whole-system protein yield ranged from 143 to 566 kg protein ha^{-1} and was affected by fertilization ($p = 0.042$) with near significant interactions with water (water $p = 0.064$) and cropping system (water:cropping system:fertilization $p = 0.065$) (**Figure 4B**). Under fertilized ambient rainfall conditions, protein yield was significantly higher for maize-pigeonpea intercropping and sole maize than for sole pigeonpea. Under all drought and all unfertilized conditions, protein yield did not vary between cropping systems. The drought resistance of protein yield was similar to that of whole system grain yield (**Supplementary Figure 5**). Patterns of theoretical whole-system caloric yield and drought resistance were similar to those of whole-system grain yield (**Supplementary Figure 5**). Caloric yield ranged from 3,126 to 22,549 thousand kcal ha^{-1} and was interactively affected by cropping system ($p < 0.001$), water ($p = 0.021$), and fertilization ($p = 0.007$; cropping system:water:fertilization $p = 0.010$). Variation in theoretical nutritional yield reported here is due to treatment effects on measured crop yield not on grain nutritional content, with nutrient content within a crop assumed to be constant for all treatments (see Methods).

Land Use Efficiency

Maize-pigeonpea was the only intercropping system (cropping system $p = 0.078$) with a mean Land Equivalent Ratio (LER) >1 across water and fertilization levels, indicating a consistent advantage of over monocultures in that less land was required

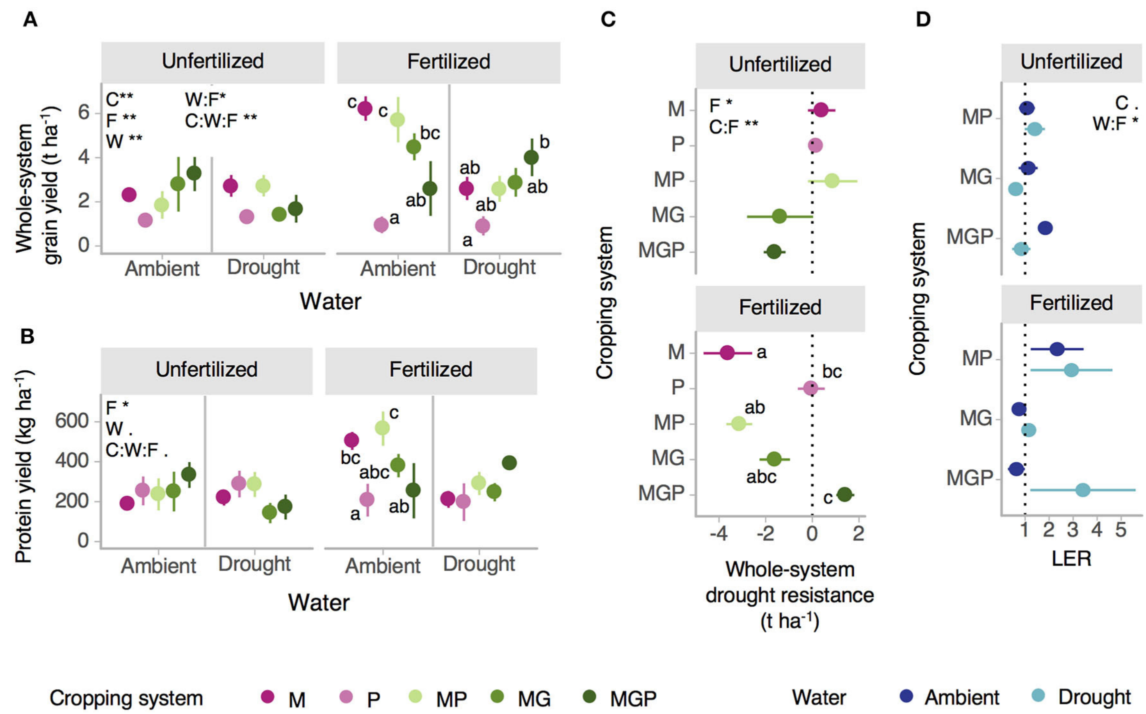

FIGURE 4 | Effects of water, cropping system, and fertilization on **(A)** whole-system system dry grain yield and **(B)** whole-system protein yield (see **Supplementary Figure 5** for whole-system caloric yield). **(C)** Effects of cropping system and fertilization on whole-system yield drought resistance; dotted lines indicate high drought resistance (zero change due to drought) (see **Supplementary Figure 5** for drought resistance of caloric and protein yield). **(D)** Land equivalent ratio (LER) for grain crops (maize, pigeonpea) in all intercropping systems under drought and ambient rainfall conditions with and without fertilizer. Dotted line indicates LER = 1, above which intercropping is more efficient than its corresponding monocultures in crop production per unit area. Cropping system abbreviations: M, Maize; P, Pigeonpea; MP, Maize-pigeonpea; MG, Maize-gliricidia; MGP, Maize-gliricidia-pigeonpea. Treatment factor abbreviations: W, Water; C, Cropping system; F, Fertilization. Asterisks indicate significance of treatment effects (** $p < 0.01$, * $p < 0.05$. < 0.1). Means sharing the same letter are not significantly different within a panel of a graph (alpha = 0.05). Error bars = standard error; $n = 3$ (fertilized MGP $n = 2$).

in maize-pigeonpea intercropping than in maize and pigeonpea monocultures to achieve the same grain production (**Figure 4D**). LER was interactively impacted by water and fertilization (water:fertilization $p = 0.031$). For maize-gliricidia and maize-pigeonpea-gliricidia intercropping, whether or not intercropping was superior to monocropping depended on the water level and whether or not fertilizer was applied. In these two intercrops, LER was above one in fertilized drought and unfertilized ambient conditions but was below one in unfertilized drought and fertilized ambient conditions.

DISCUSSION

In this experiment, we empirically tested whether intercropping enhances drought resistance and yield under drought at the single crop and whole-system scales, with and without fertilizer addition. We show that maize grain yield was negatively affected by drought in some cropping systems under fertilized conditions, whereas pigeonpea grain yield was not impacted by drought (**Figures 3A,C**). Whole-system grain yield and theoretical caloric and protein yields in two intercropping systems, maize-pigeonpea and maize-gliricidia, were as high as in sole maize across all water levels with and without fertilizer (**Figures 4A,B**, **Supplementary Figure 5**). Maize-pigeonpea performed most strongly compared to other systems in terms of protein yield (**Figure 4B**). Maize-pigeonpea was the only intercropping system that consistently required less land than its corresponding monocultures to produce the same yield under a broader range of inputs (mean Land Equivalent Ratio >1) (**Figure 4D**). All intercrops maintained or increased whole-system drought resistance compared to the standard sole maize across fertilization levels (**Figure 4C**). We also report a novel rainout shelter design for drought experiments made with locally sourced materials that successfully reduces soil moisture without creating sizable artifacts for the crop microenvironment (**Figure 2**).

Impact of Drought Differs Between Crops

Drought had no significant detrimental impact on pigeonpea yield but did significantly reduce maize yield under fertilized conditions in two cropping systems, sole maize and maize-pigeonpea (**Figures 3A,C**). When nutrients were limiting (a common scenario in low-input smallholder cropping systems), drought did not limit yields of maize, the staple crop, in any cropping system (**Figure 3A**). This superior drought response of pigeonpea was observed despite the drought imposition period spanning two-thirds of the growing season of pigeonpea compared to only half of the growing season of maize (**Figure 1**) and is consistent with the deep early-season taproot development and slow initial shoot growth of pigeonpea (Snapp et al., 2003).

Maize-Pigeonpea Intercropping Is Consistently Superior to Monocultures Across Rainfall and Fertilization Conditions

Additive maize-pigeonpea intercropping was the only intercropping system that consistently outperformed its corresponding monocultures: it used a smaller land area to produce the same amount of food, particularly under drought, regardless of whether or not fertilizer was added (mean Land Equivalent Ratio >1) (**Figure 4D**). Maize-pigeonpea was also one of two intercrops with whole-system grain, protein, and caloric yields that were as high as in sole maize, a finding consistent across all water levels with and without fertilizer (**Figures 4A,B**, **Supplementary Figure 5**). The high productivity of maize-pigeonpea intercropping is consistent with meta-analysis evidence showing that intercropping C4 cereals with C3 legumes, particularly additive intercropping, increases efficiency of land use for crop production compared to sole cropping (Yu et al., 2015). That maize-pigeonpea LER does not significantly increase under drought compared to ambient rainfall is consistent with meta-analysis results that that LER did not vary with irrigation level or aridity (Martin-Guay et al., 2018), but contrasts with one study that found that the LER of C4 cereal/C3 legume replacement intercropping systems increased with drought (Natarajan and Willey, 1986). Superior maize-pigeonpea performance across growing conditions could be due to limited niche overlap in rooting over time and space, more efficient use of existing of water, nutrient, and light, competition for soil water and nutrients below thresholds for negative impacts on whole-system yield, and/or facilitation of maize by legumes through decomposition of legume residue and maize uptake of mineralized nitrogen.

Under drought, the superior performance of maize-pigeonpea intercropping compared to monocultures could also be due to pigeonpea being the less drought-sensitive crop in the mixture (**Figure 3**) and/or long term shifts in soil properties due to intercropping. Trends toward greater drought resistance and LER under drought for maize-pigeonpea compared to sole maize (**Figure 4**) could indicate a portfolio effect whereby pigeonpea responds less negatively than maize to drought, similar to the stabilizing effect of crop diversification that has been observed at larger spatial scales (Doak et al., 1998; Tilman et al., 1998; Tilman, 1999; Birthal and Hazrana, 2019; Renard and Tilman, 2019). The advantage of maize-pigeonpea intercropping over monocultures under drought could also be driven by impacts of maize-legume intercropping on soil hydrology and fertility as shown in longer term studies. Smallholder intercropping systems similar to those tested here have been shown to impact soil and plant mechanisms by mediating facilitative interactions and plant nutrient and water acquisition. Intercropping maize with grain and tree legumes increases soil carbon and water infiltration with measurable gains in soil moisture during periods of peak rainfall especially near trees and particularly in sandy soils (Jackson et al., 2000; Makumba et al., 2006; Chirwa et al., 2007; Rusinamhodzi et al., 2012; Muchane et al., 2020). We did not find a significant benefit of intercropping for soil moisture *per se* (**Figure 2A**) and measuring water fluxes through soil and plants and plant water status under drought could provide more robust insight into drivers of greater food production in diversified systems in water-limited scenarios (Nyadzi et al., 2003; Kimaro et al., 2016). Improved leaf water potential in intercrops and hydraulic lift by pigeonpea to maize have been reported (Harris and Natarajan, 1987; Sekiya and Yano, 2004), but their importance for increasing crop yield rather than merely facilitating plant survival under drought is unclear and likely minimal.

Productivity Outcomes of Intercrops With Gliricidia Are Inconsistent

In contrast to maize-pigeonpea, maize-gliricidia and maize-pigeonpea-gliricidia intercropping were inferior or superior to monocultures depending on input combination (i.e., water level and whether or not fertilizer was added) (**Figure 4D**). Our results highlight the importance of empirically considering multiple potentially interacting resource limitations in the field when testing the resilience of diversified cropping systems. Linear trends in LER observed along broad global fertility and aridity resource gradients (Yu et al., 2015; Martin-Guay et al., 2018) may not reflect the response of low fertility, high aridity marginal smallholder systems to water, and nutrient limitations. The absence of reliable advantages in land-use efficiency of intercropping systems with gliricidia over monocultures could be due to competitive for resources between annual crops and gliricidia (Jackson et al., 2000; Chirwa et al., 2007; Makumba et al., 2009; Muthuri et al., 2009) that were not measured here, such as for light under more productive conditions (i.e., with fertilizer, without drought). Assessing the competitive ability of each crop in mixture through indices such as the competitive ratio and aggressivity (McGilchrist, 1965; Willey and Rao, 1980) would have allowed a fuller evaluation of intercropping but was precluded by the lack of a sole gliricidia control or measurement of gliricidia productivity, a limitation of this study.

Our study focuses only on grain yields of maize and pigeonpea because these are the main benefits determining the adoption of intercropping by smallholder farmers, and excludes other products of gliricidia. The gliricidia fuelwood yield (Kimaro et al., 2007) is a bonus product in intercropping, and its inclusion in LER calculations would make gliricidia intercropping systems more likely to be advantageous compared to monocultures. On-farm wood production also provides benefits that cannot directly be evaluated based on its contribution to food security. Studies showed that, depending on the location, people in rural areas in Tanzania—often women and children—spend a substantial amount of time collecting firewood. Firewood collection trips take up to several hours in Tanzania (Kegode et al., 2017), promoting gender-based violence against those most responsible for firewood collection (Levison et al., 2018).

No Downside of Additive Intercropping for Drought Resistance

This study offers evidence that additively intercropping maize with pigeonpea and/or gliricidia does not compromise drought resistance (i.e., avoids increasing the risk of yield loss) (**Figure 4C**), despite its higher plant density. At the individual crop scale, intercropping maintained, or increased maize drought resistance and did not impact pigeonpea drought resistance (**Figures 3B,D**). Similarly, at the whole-system level, all intercropping systems maintained or increased the drought resistance of whole-system grain, protein, and caloric yields compared to the standard sole maize system (**Figure 4C**, **Supplementary Figure 5**). This finding weakly supports the conclusion that intercropping increases crop yield stability, particularly in the tropics (Raseduzzaman and Jensen, 2017), in that we did not observe greater yield losses in intercropping than in sole cropping. The lack of a strong interaction between water input level and intercropping vs. monocropping that we found is similar to a study showing that maize yield response to maize-cowpea intercropping is not consistently affected by drought (Steward et al., 2019).

The lack of downside risk of additive intercropping for drought resistance suggests that water use patterns between species were sufficiently complementary or that increased competition for soil moisture was not enough to make intercrops more susceptible to drought, at least for the planting densities typical of the study area and within the ranges of rainfall and soil moisture in our study. Additive intercropping outcomes for whole-system yield under drought may be negative in intensive cropping systems with higher planting density.

Novel Tall Slatted Rainout Shelter With Few Crop Microenvironment Artifacts

We report a novel rainout shelter design for drought experiments that combines the slatted design used in lower stature systems such as grassland, deserts (Yahdjian and Sala, 2002; Gherardi and Sala, 2013), and wheat (Kundel et al., 2018) with the taller ($\geq$2 m) height of fully covered rainout shelters used previously in maize (Steward et al., 2019) and wheat (Degani et al., 2019). Our rainout shelter design successfully reduced soil moisture without creating crop microenvironment artifacts such as higher air temperature, relative humidity, or vapor pressure deficit (**Figures 2B–D**). Rainout shelters reduced photosynthetically active radiation (PAR) at midday compared to without shelters by an average of 16% (**Figure 2E**), lower than the maximum midday difference previously reported with rainout shelters in the field (25%) (Yahdjian and Sala, 2002). On a cloudy day typical of light conditions during data collection dates (maize anthesis through harvest), average light transmission through rainout shelters was 850 vs. 1,012 μmol m^{-2} s^{-1} without rainout shelters (**Supplementary Figure 4**). This difference corresponds to a 13% reduction in maize photosynthesis based on maize photosynthetic light response curves (Leakey et al., 2006). This gap in photosynthesis due to light interception by rainout shelters would theoretically become progressively smaller under sunny conditions as photosynthetic light response curves reach a light-saturated rate, although the canopy light response might be less saturating for maize as a C4 plant than for pigeonpea and gliricidia. We observed no negative effect of rainout shelters on maize yields in multiple cropping system-fertilization treatment combinations nor on pigeonpea yields (**Figures 3A,C**), indicating that light was not yield-limiting. We acknowledge the limitations of above-canopy rainout shelters in terms of the confounding of rainfall interception and light interception and potential impacts on yield [unless controlled for with additional rainout shelters without rainfall interception (Kundel et al., 2018)]. However, we note that rainout shelter impacts on light are often not reported (Degani et al., 2019; Steward et al., 2019) or are reported as roof material manufacturer specifications (Kundel et al., 2018) but not measured in the field. Also, despite our study being conducted over one season, we note the value of rainout shelter experiments for controlled manipulation of rainfall, effectively isolating the impact of drought in terms of rainfall amount while holding rainfall timing and other weather and sources of variation constant. We conducted our study in a season with rainfall slightly above average (**Figure 1**), and our water treatments thus span reasonable ambient rainfall and drought rainfall levels for this region.

Rainout shelter designs with slatted roofs allow varying amounts of incoming rainfall to be intercepted based on slat spacing and minimize side effects on the crop microenvironment by allowing greater air flow and some direct radiation. These qualities make them apt for longer term (>1 month) drought simulations (Yahdjian and Sala, 2002; Kundel et al., 2018). However, roof slats are often made of relatively expensive transparent acrylic bands (Yahdjian and Sala, 2002; Kundel et al., 2018). Other studies use more economical but less durable greenhouse plastic fully covered rainout shelter roofs with impacts on air temperature and crop microenvironment (Degani et al., 2019; Steward et al., 2019). We highlight our use of transparent corrugated polycarbonate roofing material as more economical than acrylic and sturdier than greenhouse plastic. This roofing material enables use of slatted roof rainout shelter designs in locations where corrugated polycarbonate roofing material is more widely available and affordable.

Significance for Socio-Economics and Drought Adaptation of Maize-Based Farming Systems

Superior performance of maize-pigeonpea intercropping compared to monocultures across rainfall and fertilization levels should be considered in its broader socio-economic context. Sole maize and maize-pigeonpea intercropping achieved similar whole-system grain, caloric, and protein yields, but maize-pigeonpea performed most strongly compared to other systems in terms of protein yield (**Figures 4A,B**, **Supplementary Figure 5**). Thus maize monoculture and maize-pigeonpea intercropping may be comparable in their importance for addressing food insecurity based on grain and calorie yields but maize-pigeonpea intercropping produces diverse nutrients (e.g., protein) and may therefore be better suited to address malnutrition. Despite this potential benefits and wider adoption and consumption of pigeonpea in other regions of Tanzania, the adoption of pigeonpea in semi-arid central Tanzania is currently limited by low access to pigeonpea seed via local supply chains (Shiferaw et al., 2008). Costs and benefits of intercropping in terms of labor and economics also influence use of intercropping. The change in crop output per unit labor input in intercropping is highly variable and generally around one, indicating that labor demand increases with intercropping but so does yield to a similar degree (Dahlin and Rusinamhodzi, 2019). On average across Africa, intercropping generally increases gross incomes but with significant variability depending on other management practices (Himmelstein et al., 2017). These differences underscore the importance of matching systems to both risks and individual community needs (Sinclair and Coe, 2019).

Our results have implications for efforts to identify smallholder cropping systems with not only greater productivity but also greater long-term stability (i.e., lower year-to-year fluctuations in yield) (Urruty et al., 2016; Peterson et al., 2018). We found that intercropping maintained or increased whole-system drought resistance (**Figure 4C**). This suggests that intercropping in most cases may avoid compromising long-term interannual stability and in one case may benefit it. We found greater differences between cropping systems in their drought resistance when fertilizer was applied, which suggests that fertilizers could compromise long-term yield stability in some cropping systems despite boosting productivity. This finding is consistent with adoption of diversification practices of intercropping and rotations by smallholder farmers in East Africa to adapt to change (Kristjanson et al., 2012) and modeling evidence that integrating legumes into maize systems maintains or increases the modeled chance of meeting smallholder household calorie and protein needs without increasing fertilizer inputs (Smith et al., 2016). The potential of intercropping and agroforestry practices that build soil carbon (Muchane et al., 2020) to boost cropping system drought resistance (Iizumi and Wagai, 2019) should be tested in long-term trials beyond the length of our study. Such testing would generate better understanding of how to mitigate greater drought limitation of crop yield as fertilizer promotion and adoption across semi-arid Tanzania reduce nutrient limitation of crop yield.

CONCLUSIONS

Our study presents one of the few assessments of additive maize-legume intercropping impacts on yield vulnerability to drought in resource-poor smallholder maize systems. For the environment tested here, we conclude that diversifying maize-systems through maize-pigeonpea intercropping lowers the land area needed to produce the same food, including under drought, avoids compromising—but does not build—drought resistance, and can help supply protein in maize-dominated landscapes. Outcomes of intercropping will likely vary with edaphic conditions, climate, and drought stress timing and intensity. We provide direct evidence that diversification of maize-based cropping systems via intercropping constitutes a tool adapted to low-input smallholder systems to build productivity across drought and non-drought conditions in the face of a changing climate.

DATA AVAILABILITY STATEMENT

Datasets are available on request. The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation. Datasets will be made publicly available following publication at the World Agroforestry Research Data Repository: https://data.worldagroforestry.org/.

AUTHOR CONTRIBUTIONS

AK conceived, designed, and established the intercropping field trial at which the drought experiment was conducted. LR, AG, AK, and TR conceived and designed the drought experiment. LR and JH established the drought experiment and collected data with support from AK. LR processed the data with contributions from JH and AK, and analyzed the data. LR drafted the manuscript, which was critically revised by AG, AK, JH, and TR. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2020.562663/full#supplementary-material

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PERSPECTIVE
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"Who Has the Power to Adapt?" Frameworks for Resilient Agriculture Must Contend With the Power Dynamics of Land Tenure

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This special issue aims to develop how Diversified Farming Systems (DFS) may contribute to adaptive capacity in order to confer resilience to agricultural systems. In this perspective article, I argue that a framework for DFS and adaptive capacity must adequately contend with the role of farmland tenure on the shape of food systems to be both internally coherent and socially redistributive. Yet, both DFS and adaptive capacity scholarship deemphasize or mischaracterize the role of farmland tenure in favor of ecosystem dynamics. In this paper, I bring together lessons from the agrarian change literature and established critiques of resilience thinking to demonstrate core problems with a framework aimed at linking DFS to adaptive capacity without adequately addressing the role of farmland tenure. Namely, applying resilience thinking as a framework to understand food systems change prioritizes concern over final "states" or processes of farming systems and may ignore who has the power to adapt or who derives benefits from adaptation. The critiques of resilience thinking inform that the result of this apolitical elision is (1) entrenchment of neoliberal logics that place responsibility to cultivate adaptation on individual farmers and (2) provisioning of legitimacy for land tenure systems that can most readily adopt DFS, without understanding how well these systems distribute public benefits. Resilience reformers call for ways to include more power aware analysis when applying resilience thinking to complex socio-technical systems. I suggest that centering the role of land tenure into the frameworks of DFS and adaptive capacity provides a lens to observe the power relations that mediate any benefits of agricultural diversification. Integrating analysis of the social and legal structures of the food system into the DFS for adaptive capacity formulation is a crucial step to transforming resilience thinking from an apolitical tool to transformative and power-aware applied science.

Keywords: land tenure, resilience thinking, food systems, diversified farming systems, property, adaptive capacity

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INTRODUCTION

This special issue on *Diversifying Farming Systems for Adaptive Capacity* is motivated by the need to foster resilient farming systems in the face of the Anthropocene's "triple challenge" of climate change, biodiversity loss, and sustainable resource provisioning. The issue-framing paper (Petersen-Rockney et al., 2020) introduces a formulation whereby Diversified Farming Systems

(DFS) supports the emergence of an equitable adaptive capacity when applied across a variety of ecological, agronomic, and social dimensions. In response to these threats, the authors of the framing paper suggest that agricultural systems can move along either simplifying or diversified pathways in pursuit of resilience. The central claim of this formulation is that "diversifying processes can weave a form of equitable and sustainable adaptive capacity that is fundamentally distinct from the narrow and brittle adaptive capacity produced through simplification" (Petersen-Rockney et al., 2020, *p.* 1). To further the ambitious aims of the framing paper, I provoke a deeper conversation about how farmland tenure complicates a framework linking DFS and adaptive capacity[1]. While I support the core logic of the framing paper, I argue that by focusing on the complexity of land tenure relations, it is possible to show how building on concepts of adaptive capacity and resilience introduces challenges to effectively engage with power-laden concepts like equitable distribution and justice. I argue a subtle critique of the types of thinking presented in the framing paper and in resilience approaches more generally to help clarify how DFS frameworks may achieve equitable adaptive capacity. Specifically, I suggest that a focus on land tenure relations provides a mechanism to observe (and perhaps act on) the way interventions to improve agricultural adaptability deliver benefits to different actors across space and time.

Land tenure, the relationships of social and legal order that allocate resources to people, is the sieve through which agricultural decisions are ultimately made. Land tenure is produced through formal artifacts like property law and lease agreements, through the socio-cultural imaginations about who has legitimate claims to benefit from land's resources, and notions of proper interactions across landscapes (Blomley, 2016). Without an adequate political emphasis on how land tenure, property relations, and land access constrain and enable farmer decision making, a framework response to the "triple threat" may fail on two fronts. First, recommendations for how to apply DFS practices to generate adaptive capacity may be muted by restrictive land tenure regimes in many agricultural contexts: farmers may be receptive to DFS, but with tenure insecurity may not have the power to make adaptive changes. Second, applying DFS for resilience alone may entrench unjust tenure regimes: farming systems may succeed in diversifying for resilience, but entrench the flow of ecosystem services and other benefits to those that control land access mechanisms. This second potential failure may provide legitimacy for unjust land tenure regimes as narrow conceptualizations of resilience become more normative in political and technical debates (Cretney, 2014). These problems are acute in geographies characterized by social and legal commitments to private property.

To support my arguments, I first examine the way resilience and DFS scholarship contends with the role of farmland tenure on decision making. I show how resilience thinking emphasizes individual farmer decision making for adaptive capacity, tending to abstract the tenure contexts in which these decisions are made. Within the adaptive capacity and DFS literature that focus on global North contexts, land tenure regimes are often viewed as immutable, where any goal of alternative DFS decisions are achieved *through* the logics of established tenure norms and property law. When the core logics of property are unchallenged, I argue that the framework paper (and other resilience frameworks) may end up entrenching legal structures that influence the equity of agricultural systems by not engaging with the power dynamics of how land is distributed.

I then draw from research on agrarian political economy and property theorists to show how farmland tenure is understood as an upstream driver of land decision making. Land tenure is much more than discrete categories like "rented" or "owned,"—it is a regime, consisting of the social, cultural, and legal systems that distribute the power to access land and effectively exclude others (Hall et al., 2011). In some cases, like in much horticultural land in the US, land tenure regimes are somewhat homogenous, dominated by rigid legal norms of property through which security of tenure is associated with possession of a formal, state-backed title. In the global South there are more examples of secure tenure that are dependent on customary rights, semi-private holdings, community-held, or commons rather than the possession of a formal title (Robinson et al., 2018).

In this understanding of how land tenure functions, the power to decide (and therefore the capacity to adapt) is imbricated in socio-cultural notions of ownership and property, how they have been shaped over time, and the legal commitments that deliver access or ownership rights (Trauger, 2014). Given this understanding of tenure, decision making authority is constantly "assembled" through a negotiation of the many interests in land's existing, imagined, and potential resources (Meinzen-Dick and Mwangi, 2008; Li, 2014).

The challenges of applying resilience frameworks to value-laden problems like land access stem from core problems with resilience thinking's noted inability to account for politics and justice. An overemphasis on preserving core socio-ecological function leaves out questions of who benefits from a wide array of stable states that deliver "services" and functionality. However, centering the role of farmland tenure allows researchers in pursuit of an equitable resilience to ask, "Who has the power to adapt?" In this way, an analysis of farmland tenure could be a lens to observe the social relations that mediate any benefits of agricultural diversification. Practically, this signifies championing existing—but under-represented—farmland tenure systems, reforming dominant mal-adaptive property relations, and the co-creation of new land tenure systems to meet the evolving challenge of the "triple threat." An exploration of "regenerative agriculture," arguably a deeply stable and resilient state that operates through dominant logics of restrictive land ownership, is illustrative.

To articulate my perspective, I first examine the way resilience thinking and adaptive capacity tend to deemphasize or mischaracterize the role of land tenure. I contrast this general "agnosticism" toward land governance with the way agrarian

[1]The author contributed to the special issue framing paper (Petersen-Rockney et al., 2020) in particular describing how land tenure presents challenges to diversified farming practices. In doing so, the author saw how land tenure complicates the framework presented and saw an opportunity to explore the role of land tenure of adaptive capacity more deeply in an effort to strengthen to framing paper and other resilience-based DFS formulations.

scholars prioritize land governance as a key site for of reform for unsustainable food systems. In conclusion, I argue that embracing the complexity of land tenure provides a window into the political aspects of food systems' transition, where emphasis on diversification alone is insufficient to marshal transformative change.

RESILIENCE THINKING'S AGNOSTICISM TOWARD LAND TENURE

In this section I analyze key resilience and DFS texts to show how resilience frameworks tend to elide, deemphasize, or mischaracterize land tenure's role in agricultural contexts.

The concept of adaptive capacity, as a process that confers resilience, is mobilized to encourage new land use decision making in the pursuit of sustainability outcomes (Folke et al., 2010). In this pursuit, adaptive capacity and resilience literature is agnostic to the normativity of property relations (Joseph, 2013). Resilience scholars note the forces that drive behaviors of land managers, but usually do not specify the conditions that grant these users power to make decisions. Some researchers have explored the role of discrete tenure categories on determining sustainability indicators, for example the effect of owned land vs. rented on conservation decisions (Deaton et al., 2018; Ontl et al., 2018), but foundational resiliency contributions tend to elide the social and political processes of land tenure on shaping social ecological systems (SESs). In the Darnhofer et al. (2010) paper that first linked resilience thinking to agricultural contexts, the authors entrench the idea of farm decision making as determined by ownership status:

> As decision making on farms is under direct influence from humans [...], applying resilience thinking to farming systems requires careful attention to the social domain. Indeed, *private ownership means that it is the farmers' right to manage their property* as they see fit [...] Thus it is ultimately the farmer who decides whether or not to cut down a windbreak, how much agrichemicals to use on his or her field, and whether to plant a woodlot or to drain a swamp (Darnhofer et al., 2010, p. 192, emphasis added).

Darnhofer et al.'s intention in the above is to highlight the forces that may shape an individual's decision making. To promote adaptive capacity, they argue, the motivations and knowledge capacity of an individual farmer must be understood. This line of thinking encourages much subsequent work aimed at understanding what motivates "behavior change" in the social worlds of landed farmers and farm managers (Sutherland and Darnhofer, 2012; Sutherland et al., 2012). The social domain that influences an individual land owner's decision making is important, but it ignores the power relations that shape who has the power to assume the role of land manager and what constraints they face because of their tenure context. The focus on the individual mind of the farmer assumes that all farmers have the power to make adaptive decisions (or perhaps the ones worth focusing on are the ones who have ownership rights).

In DFS scholarship, the role of land tenure is more recognized, but rather than being mischaracterized, it is viewed as part of the food system that must be worked through rather than transformed. Kremen et al. (2012) for example, recognizes how *de jure* and *de facto* discrimination in the US influenced a 98% decrease in the number of black farmers between 1910 and 1997. The authors discuss how many of these farmers practiced DFS, and that more diversity of US farmers would strengthen the social-ecological diversity of the food system. While the authors recommend a series of strategies to encourage inclusion of more farmers of diverse backgrounds, the legal and cultural systems that have shaped racially skewed land access regimes remain entrenched (Horst and Marion, 2018). In this way DFS frameworks recognize the need for diversified tenure categories, for example more smallholders of color, but do not challenge the property structures that create land tenure disparities.

THE ROLE OF LAND TENURE ON AGRARIAN CHANGE, FARMER DECISION MAKING, AND DISTRIBUTION OF RESOURCES

While resilience scholarship and DFS work do not foreground land tenure, research concerned with the political economy of agrarian change has long centered the foundational role of land tenure on food system composition. First, critical agrarian studies argue land enclosure, via the assertion of property rights, is the first pillar of agrarian capitalism (Bernstein, 2010). Beyond who possesses rights, scholars show how land *access*, or the many ways that institutions, individuals, policies, technologies, and power relations structure one's *ability* to benefit from a resource (Ribot and Peluso, 2009). Investigation of these access mechanisms, often codified in the informal and formal rules associated with land tenure, help explain who is able to make land use decisions and why DFS farmers are often marginalized in this process (Sikor and Lund, 2009). This research suggests that if the agency of agricultural decision making is found in the socio-legal structures that shape tenure, the role of property ownership and land governance is a "lock-in" that inhibits many food movement aspirations and is thus a target for change (Rotz et al., 2019; Lang, 2020). The constraining role of land access is a chief concern of global peasant groups like Via Campesina, who prioritize land reforms in the name of their agroecological objectives (Desmarais, 2002).

Land tenure regimes also shape who can *become* a land manager. Farmland financialization research shows how land registration and the formalization of property rights assemble land in a way that integrates with global capital and productivist agriculture (Li, 2014; Fairbairn, 2020). For young and beginning farmers who have the technical capacity to practice DFS, the land access barrier prevents their ability to enter the agricultural workforce at meaningful scales (Beckett and Galt, 2014; Carlisle et al., 2019) or limits their agency in restrictive tenant farming operations (Calo and De Master, 2016). Tenure also props up racial inequity, as those who control land access mechanisms align with the dominant groups in society (Horst and Marion, 2018; Figueroa et al., 2020).

Rethinking farmland tenure entitlements is becoming more frequent and pressing for the aims of food systems transition (IPES Food, 2019). Land tenure security is seen as a policy target to allow for alternative agriculture, environmental justice, and agrarian transitions (Lawry et al., 2014). Scholars who work with concepts such as agroecology and food sovereignty champion marginalized tenure regimes like common-held, indigenous, or customary management systems with long histories of sustainable use (Borras and Franco, 2013; Penniman, 2018; Giraldo and McCune, 2019).

Agroecologists view secure farmland tenure as an enabling condition *that must proceed* technical food system interventions (Anderson et al., 2019; Giraldo and McCune, 2019). Kepkiewicz and Dale (2018) argue that "challenging hegemonic assumptions about private property" must occur before distributive forms of agroecology can emerge in the settler-state context of Canada. Edelman et al. (2014) asks, "What kinds of (land) property relations might characterize a food-sovereign society?" Scholars note that unbalanced political power flows through entrenched property relations and thus serious attention to challenging the broader "episteme of ownership" is needed for food system reform (Trauger, 2014, p. 1144).

DISCUSSION: THE PROBLEM WITH LOSING SIGHT OF LAND TENURE FOR ADAPTIVE CAPACITY

Amidst the increasing call from agroecologists and advocates of food sovereignty for food system transformation *through* land governance interventions, resilience thinking remains agnostic toward land tenure reforms. This problem of losing sight of land tenure for adaptive capacity has the effect of failure on two fronts: a failure of misplaced agency and a failure of theory of change. The failure of misplaced agency may incorrectly locate the power of decision making in land tenure regimes where the "farmers" are overly constrained by the social relations that condition their land access. The failure of theory of change occurs where the goal of encouraging adaptive capacity through DFS succeeds but in land tenure regimes that entrench an unequal distribution of resources.

In the first failure, consider geographies of Westernized liberal democracies, where the cultural, legal, and social notions of property and ownership are hegemonic and farmers are embedded in rental, indebted, or contract relationships (Wittman et al., 2017). These tenure regimes are reflected by the dominance of insecure farm tenancies, contract farming relationships, and agribusiness operations where decisions of land managers are constrained by the will of the land owner or land owning entity (Barnett et al., 2020). The knowledge, perceptions, motivations, and capacities of an insecure tenant farmer matters little to the resilience of the farm if their actions are constrained by a month-to-month lease or unequal landlord tenant relationship (Calo and De Master, 2016). Even if a tenant farmer implements DFS practices geared toward resilience, the benefits may accrue to the land owner where the power of the owner trumps the entitlements of the user. Some diversification management practices deliver such near-term effects that even an insecure tenant farmer will benefit. Yet, as the benefits sought tend toward the long-term or the public facing, there appears little motivation or reward for insecure tenants to instigate decisions that lead to system adaptability.

In the second failure, without understanding who has the ability to implement new practices and who benefits from such changes, resilience thinking becomes dangerously apolitical. An unsettling result may come about if resilience thinking succeeds: the diversified farming practices that promote adaptive capacity may be more readily achieved through simplified land tenure regimes. In the Westernized liberal democracies, where the "fee simple absolute" form of property delivers strong and unlimited rights to decision making over land, the focus to incentivize a change to resilient states is likely to align with owners of private farmland property (Shoemaker, 2020). In regimes of more customary or collective tenure, the benefits of diversification will map onto the many contexts of land governance, each with their own distributional effects. In both cases, diversification for adaptive capacity reifies the dominant property relations by granting legitimacy through resiliency.

Imagine that the technical debates of DFS and resilience have been resolved. Land managers are now able to a follow a set of consensus steps to maximize resilience, safeguard their livelihoods against future shocks, and address the "triple threat." What farmland tenure regimes are most able to implement these changes? Farmers constrained in their long term decision making would potentially reject DFS practices and their "failure to adopt" would be attributed to lack of good information (Calo, 2018). Shifting cultivators without recognition of tenure may have the capacity to make changes in production practices, but lack the ability to implement changes over a continuous land area. In turn, the farms with simplified tenure could easily make changes at scale, reaping the benefits of diversification and meeting the indicators of resilience, which may be supported by policy incentives.

This thought experiment is more real than abstract. The rise of sustainable investment trusts, half earth philosophies, and land sharing advocates indicates the embrace of a resilience logic that prefers the "installation" of the correct type of land manager (Büscher and Fletcher, 2019). A management unit that can make large scale changes to land use is seen as a legitimate pathway to promote sustainable food systems. The power of this logic enrolls actors like conservation organizations, governments, and funding streams to focus on large plots of land with simplified land ownership as the targets for solutions like "regenerative" and "climate smart" agriculture (Borras and Franco, 2018). Under the heuristics of simplified vs. diversified pathways to adaptability, "regenerative agriculture" emerges as stable and diversified when examining the management practices and provisioning of ecosystem services, but simplified in terms of land tenure (and likely labor). The case of regenerative agriculture shows how both simplified and diversifying pathways can be pursued in parallel, with implications for the equity aspirations of DFS. This dynamic could explain how agribusiness interests show early articulation with the more technically oriented "regenerative agriculture" because it may offer a robust preservation of socio-ecological function without challenging the status quo of power relations (Wozniacka, 2019). Yet, a focus on land tenure dynamics of regenerative agricultural models offers a way to question if the diversified practices therein are capable of delivering equity.

Learning From the Critique of Resilience Thinking

The incongruency of the core logics of adaptive capacity with the complexity of land tenure is indicative of a broader critique of resilience approaches. In the critique, resilience's broad acceptance as a normative good is part of the project of neoliberalism (Reid, 2012; Aradau, 2014; Rotz and Fraser, 2018). The pitfalls described by resiliency's critics help illuminate the barriers to be overcome by DFS frameworks that also aspire for a just food systems transformation. Brad Evans and Julian Reid, leading resiliency critics write:

> Building resilient subjects involves the deliberate disabling of the political habits, tendencies and capacities of peoples and replacing them with adaptive ones. Resilient subjects are subjects that have accepted the imperative not to resist or secure themselves from the difficulties they are faced with but instead adapt to their enabling conditions (Evans and Reid, 2013, p. 85).

Resilience thinking places the responsibility to adapt and the requirement to change in the hands of the individual, replacing an entitlement of security with a responsibility to adapt. A "resilient subject" as Reid argues, is one that adapts to conditions without questioning what caused those conditions to arise. An insecure tenant is asked to implement practices to improve their margins over the short term, not question the tenant landlord relationship. A nomadic pastoralist is considered as over exploiting their resource base rather than as relegated from lowland pasture enclosure.

The consequence of this feature of resilience thinking is the way resiliency interventions build on the logic of individual capacity:

> The danger, for development policy and practice, of errantly interpreting the concept of resilience as a characteristic of individuals or groups is that it could be construed as a justification to blame those who are most vulnerable and least able to marshal the resources necessary for developing resilient trajectories. Such an approach fails to adequately recognize the ways in which the adaptive capacity of individuals and groups is constrained by a variety of structures and organizations, as well as the entrenched dynamics of power (Walsh-Dilley et al., 2016, p. 4).

When resilience thinking is applied to policy actions, it appears as "capacity building" at the expense of structural reform (Aradau, 2014). This is the case when applied to an individual's psychological capacity to respond (Murray and Zautra, 2012), an individual child's capacity to adapt in learning situations (Luthar et al., 2000), and individual communities in a development context to make appropriate choices to maximize productivity (Watts, 2011). In agricultural contexts, researchers should ask to what extent frameworks to achieve resilience act as a cudgel, a tool to pointing out "bad actors" with irrational behaviors in an era where all agriculture must change to meet the challenge of the triple threat.

While some more recent perturbations of resilience thinking have begun to ask "resilience for whom" (Cretney, 2014), there is a growing concern that resilience thinking in practice aligns too closely with the logics of individual agency and private property, ultimately producing neoliberal subjectivity (Reid, 2013).

CONCLUSION: RETHINKING THE EPISTEME OF PROPERTY FOR A ROBUST DFS FRAMEWORK

The harshest critics of resilience thinking argue that the most pernicious aspects of neoliberal governmentality are only deepened with resilience thinking's continued rise (Reid, 2012; Evans and Reid, 2015). When resilience is hegemonic, the expectation of continuous disaster becomes normalized, placing responsibility for security on individual preparedness. Should agricultural resilience frameworks succeed without challenging the underlying norms of property relations, a DFS to adaptive capacity framework may further a logic of "responsibility *without* power" (Cretney, 2014, p. 633). Adjusting land tenure along simplifying or diversifying pathways is insufficient to deliver equity without considering how land tenure operates through informal access negotiations and in formal law.

However, some scholars argue that integration of the pillars of food sovereignty with resilience is the way to integrate issues of power into concerns of resilient farming systems. Walsh-Dilley et al. (2016, p.4) suggests, "Making visible the politics of resilience is the first step; the second step is to build conceptualizations of resilience that force us to contend with these tensions, contestations, and relations."

For Walsh-Dilley et al. (2016), resilience frameworks tend to be based on indicators of land use outcomes, whereas food sovereignty is founded upon the promotion of new or strengthening of marginalized entitlements. In a food sovereignty context, these entitlements appear in the form of the right to food, the right to seed, and the right to land access. The way forward is thus to examine and then strengthen these entitlements: "A rights-based approach helps us to foreground these issues; people need access to the resources with which they might build resilience (Walsh-Dilley et al., 2016)."

In practice, a framework for adaptive capacity through DFS must develop tools and approaches for just tenure interventions across all possible contexts. Asking, "Who has the power to adapt?" provokes new lines of inquiry that support the justice dimensions of a DFS for adaptive capacity framework. Questioning the values behind and the distributional effects of existing land tenure dynamics of any given agricultural system allows proponents of DFS to understand who has the ability to make changes toward a "resilient trajectory" and what might need to be changed in order to broaden the class of potential diversified farmers (Carlisle et al., 2019).

This work would follow three strands. First, some tenure regimes are suitable for a just food system transition, but need to be safeguarded or expanded. Calls to assert state recognition of customary or indigenous land use rights fit this strand. Second, where dominant forms of land tenure regimes entrench inequity, a DFS framework must engage in the socio-legal processes required to reform land governance. The progress of the Land Reform (Scotland) Acts from 1997 to 2016, that create new powers to increase community land ownership is a notable example (Lovett, 2010). Third, the development and testing of new land tenure systems that facilitate DFS practices and redistribute benefits of diversification is a crucial area of research. Initiatives like the Agrarian Commons in the US and *Terre de Liens* in France, that seek to de-commodify agricultural land for diversified production are burgeoning examples.

In sum, the cultural, legal, and social norms that drive property relations must be examined, and in some cases contested, as a precursor for food system transformation frameworks. This same logic should be applied to the interaction of DFS and adaptive capacity. Left uncontested, diversifying farming systems for adaptive capacity could lead to resilient states where the indicators of diversification are satisfied, but the entrenchment of unjust power relations come along as a result.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

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Ecosystem Services and Cash Crop Tradeoffs of Summer Cover Crops in Northern Region Organic Vegetable Rotations

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Intensive production practices characterizing vegetable farming contribute to high productivity, but often at the expense of supporting and regulating ecosystem services. Diversification with cover crops may support increased resilience through soil organic matter (SOM) contributions and physical soil protection. Vegetable farming often includes spring and fall production, limiting establishment and productive potential of over-wintered cover crops that are more widely used in the USA. In northern climate vegetable systems, warm-season cover crops planted during short summer fallows could be a tool to build resilience via ecosystem service enhancement. This project evaluated summer cover crops in the northern USA (MN and WI) for biomass accumulation, weed suppression, and contribution to fall cash crop yield. Our study included four site years, during which we investigated the effects of four cover crop species treatments, grown for 30 (short duration, SD) or 50 days (long duration, LD) alongside bare fertilized and unfertilized control treatments: buckwheat (*Fagopyrum esculentum*) and sunn hemp (*Crotalaria juncea*) monocultures, and biculture of chickling vetch (*Lathyrus sativus*) or cowpea (*Vigna unguiculata*) with sorghum-sudangrass (sudex) (*Sorghum bicolor x S. bicolor var. Sudanese*). To quantify cover crop quantity, quality, and weed suppression capacity, we measured cover crop and weed biomass, and biomass C:N. To quantify effects on cash crops, we measured fall broccoli yield and biomass. Mean total biomass (cover crop + weeds) by site year ranged from 1,890 kg ha^{-1} in MN Y1 to 5,793 kg ha^{-1} in WI Y2 and varied among species in Y1 for both the SD and LD treatments. Most cover crops did not outcompete weeds, but treatments with less weeds produced more overall biomass. Data from Y1 show that cover crops were unable to replace fertilizer for fall broccoli yield, and led to reduced fall crop yield. Broccoli in Y2 did not reach maturity due to fall freeze. Summer cover crops, because of their biomass accumulation potential, may be used by farmers in northern climates to fit into cropping system niches that have historically been left as bare soil, but care with timing is necessary to optimize weed suppression and mitigate tradeoffs for cash crop production.

Keywords: summer cover crop, vegetable rotation, organic agriculture, ecosystem service, sorghum-sudangrass, cowpea, sunn hemp, buckwheat

INTRODUCTION

Intensive production practices characterizing typical vegetable farming focus on maximal cash crop yield (provisioning services) to the detriment of supporting or regulating ecosystem services (Smuckler et al., 2012). Cover crop integration into vegetable rotations can perform supporting and regulating services such as contributing to soil carbon, nitrogen contribution, and pest suppression (Ding et al., 2006; Bulan et al., 2015; Blesh, 2018). Because cover crops increase rotational diversity, they may also provide important contributions to farming system resilience (Bowles et al., 2020). Cover crop effectiveness is typically measured by the degree of contribution to supporting or regulating services, or indirect effects of maintained cash crop yield (Kaspar and Singer, 2011; Schipanski et al., 2014). Recent surveys indicate increased farmer interest in and adoption of cover crops, with the majority of respondents reporting that cover crops have improved soil health on their farms (SARE CTIC, 2017). This interest is particularly high among organic growers, who are mandated to follow practices that combine soil fertility and pest management with biological processes (Bellows, 2005).

Vegetable farmers often grow multiple cash crops during the growing season, leaving few periods of bare ground and thus limiting opportunities for cover crop use. Across the US Midwest agricultural region, cereal rye (*Secale cereale L.*) and other cool-season grasses remain the most common cover crops (Singer, 2008). Cool-season cover crops in northern regions require relatively long periods of growth in fall and spring to produce significant biomass, which may not be feasible for vegetable systems in which cash crops, such as greens or broccoli, occupy the spring and fall cropping period. To maximize cover crop benefits within the rotational schedule of vegetable growers, cover crops sown in the main summer season between cool-season cash crops may be an opportunity to enhance diversification and benefit from the ecosystem services that cover crops can provide. Regular summer precipitation during the summer growth period makes the opportunity of summer cover crops particularly attractive, though cover crop water uptake during growth could result in tradeoffs. For example, severely reduced soil moisture has been shown to limit microbial processing of residue (Herron et al., 2009), and water stress is well-known to limit cash crop growth.

Summer cover crops have the potential to significantly enhance regulatory and supporting ecosystem services through biomass production. Cover crop biomass residue can replenish SOM, thus preventing or reversing soil organic matter (SOM) loss over time in agricultural soil (Reicosky and Forcella, 1998; Dabney et al., 2001; Steenwerth and Belina, 2008; Boyhan et al., 2016). Biomass accumulation is usually highest from grass cover crop species, reaching up to 14 Mg ha^{-1} for sorghum-sudangrass (*Sorghum bicolor x S. bicolor var. Sudanese*; sudex), grown continuously over the summer (Stute and Shekinah, 2019), or when cut for hay or foraged repeatedly during a single season (Finney et al., 2009). When grown for a short period without cutting, sorghum-sudangrass can still accumulate considerable biomass, ranging from 10 Mg ha^{-1} biomass within 66–90 days after planting (DAP) and 7.2 Mg ha^{-1} (O'Connell et al., 2015) to almost 9 Mg ha^{-1} 36–75 DAP (Creamer and Baldwin, 2000; Brainard et al., 2011). The biomass accumulation of cover crops can suppress weed growth and seed set through competitive effects (Masilionyte et al., 2017). Buckwheat (*Fagopyrum esculentum*) is a particularly effective and often-used summer cover crop because of its quick growth (Kruse and Nair, 2016). Use of summer cover crops for weed suppression may have particular benefits because they can be used to outcompete weeds at the time of year when many weeds would otherwise reach maturity and set seed in fewer than 60 days (Miyanishi and Cavers, 1980; Brainard et al., 2011). Though there is farmer interest in using weedy fallows to gain some of the soil benefits that cover crops provide, cover crop species are more desirable because of their consistent growth and maturation. Use of cover crops instead of weedy fallows limits accidental contribution to the weed seed bank and future crop-weed interference (Wortman, 2016).

Cover crops can contribute to net N immobilization or N fertility, dependent in decomposition dynamics and cover crop quality. When cover crops contribute to N fertility to following cash crops, they do so through biomass decomposition and release of nutrients. Decomposition of grass cover crop species returns nitrogen taken up during plant growth, while legume cover crops confer an additional benefit of adding fertility through biological nitrogen fixation. Multiple legume cover crop species are well-suited as summer cover crops because of their potential for biomass accumulation and provision of fertility to subsequent crops (Creamer and Baldwin, 2000; O'Connell et al., 2015). Warm-season legume cover crops have demonstrated potential to contribute more than 100 kg ha^{-1} fixed N to following cash crop production, measured by nitrogen derived from the atmosphere (Ndfa) (Büchi et al., 2015), while total shoot N contribution from a cowpea (*Vigna unguiculata*) cover crop monoculture and biculture grown for 67 days ranged from 75 to 80 kg ha^{-1} (O'Connell et al., 2015). Summer legume cover crops such as sunn hemp (*Crotolaria juncea*) have demonstrated potential to produce high levels of biomass while providing fixed nitrogen to the soil and suppressing weeds (Price et al., 2012). Fertility benefits from cover crops, whether through nutrient recycling from biomass or fixed nitrogen from legumes, may be an important tool for organic farmers to supplement organic fertilizers while providing the aforementioned ecosystem benefits. However, high cover crop biomass does not necessarily lead to high fertility benefits; the balance of nutrient immobilization and mineralization during cover crop decomposition is affected by existing SOM, microbial activity, and biomass quality, and can therefore result in systemic tradeoffs rather than simple benefits.

Combining cover crop species as mixtures can realize multiple benefits based on the complementary characteristics of individual species (Finney and Kaye, 2017). Cover crop mixtures can be particularly effective at weed suppression (Brainard et al., 2011). However, cover crop mixtures often produce less total biomass than their component species planted as monocultures (Finney et al., 2016). A key reason to use cover crop mixes is to balance biomass productivity with N fertility, by pairing high C:N grass species with nitrogen-fixing legumes (Finney and Kaye, 2017; Finney et al., 2017).

Limited research suggests that the benefits of summer cover crops, including high biomass, weed suppression, and fertility are achievable even in northern climates (USDA plant hardiness zones 1–4) (Kruse and Nair, 2016; Stute and Shekinah, 2019), though establishment of cover crops and cash crops within the same short season remains challenging. High biomass accumulation of warm-season cover crops during a short period in summer would offer farmers a diversification tool to protect or improve soil structure and fertility. However, insufficient growing time could result in cover crops having a negative effect on fall cash crop growth by immobilizing nutrients without building SOM. Our aim was to increase understanding of promising summer cover crop species and mixtures grown in northern vegetable systems within the time constraints of spring and fall vegetable crops. We quantified the degree to which short-season summer cover crops grown in soils with contrasting OM content accumulated biomass and N, contributed to weed suppression, and served as a fertilizer replacement for fall cash crops. We hypothesized that the chosen quick-growing summer cover crops species would provide ecosystem services via biomass accumulation, weed suppression, and contributions to soil fertility, but that duration of cover crop growth would affect provision of the benefits.

MATERIALS AND METHODS

The experiment was conducted in the summers of 2017 (Y1) and 2018 (Y2) on two certified organic working vegetable farms in MN and WI, both in Zone 4A. The MN soil is a Braham loamy fine sand, measured SOM 11 g kg^{-1}. The WI soil is a Crystal Lake silt loam, measured SOM 23 g kg^{-1}. Cumulative GDD (with baseline 10 C) during the 50 days of cover crop growth for the MN site were 416.9 and 545.9 in Y1 and Y2, respectively, and for the WI site, 450.6 and 549.1 in Y1 and Y2 (**Table 1**).

Between-site management was kept as consistent as possible given the options provided by farmer equipment and normal practice, with a key difference of lack of irrigation capacity at the WI site and differences in fertilization. In Y1, all cover crops were terminated using a tractor-mounted rotary mower, while in Y2, all cover crops were terminated using a walk-behind flail mower (**Table 2**).

Experimental Design and Treatments

Each site (MN and WI) used a 5 × 2 factorial randomized complete block experimental design with four replicates. The first treatment factor consisted of four cover crops species and two bare fallow controls (with and without added fertilizer). The second treatment factor was duration: each of four cover crop treatment levels was planted on two dates, representing long and short cover crop growing durations, and two bare control treatments. Cover crop species treatments included two monocultures and two bicultures. Monocultures included buckwheat (*Fagopyrum esculentum*) (75 kg ha^{-1}) and sunn hemp (*Crotalaria juncea*) (38 kg ha^{-1}) and bicultures included chickling vetch (*Lathyrus sativus*) (75 kg ha^{-1}) and sudex (*Sorghum bicolor x S. bicolor var. Sudanese*) 42.6 kg ha^{-1}), and cowpea (*Vigna unguiculata*) (44.8 kg ha^{-1}) and sudex (42.6 kg ha^{-1}). Seeding rates were calculated based on bulk seed weight. Buckwheat, vetch, and sunn hemp all had 90% germination rate, sudex 85%, and cowpea 80%. All cover crops were VNS apart from the chickling vetch, which was AC Greenfix. Cover crop species were chosen for demonstrated ability to accumulate large amounts of biomass in short duration, and suitability for growth in the warm-weather climate of the Upper Midwest. Each experimental unit, a unique combination of species and duration, consisted of a plot 3 m wide and 4.5 m long in MN, and 3.6 m wide and 4.5 m long in WI. Species treatments were planted on two dates, 3 weeks apart, to create duration treatments representing realistic available planting windows on typical organic vegetable farms in northern climates. The long duration (LD) planting was seeded in early June following spring arugula harvest. The short duration (SD) planting was seeded 2 weeks after LD planting. Cover crops were seeded at a depth of 0.5–1 inches in five passes using a six-row Jang drill seeder (two ft wide) with variable plates to control seeding rate. Cover crops were irrigated in MN to aid establishment in both years. No irrigation was used in WI. No cover crops were fertilized. All cover crop plots were left unweeded throughout growth. Weeds were removed weekly from bare plots. All cover crop plots were terminated on the same date within a site year, 50 DAP for the LD plots and 30 DAP for the SD plots. Termination in Y1 was accomplished via a tractor-driven flail mower at both sites, while in Y2 a termination used a walk-behind flail mower. In both years, termination was followed by incorporation into the soil 2 days later with a tractor-driven disk. Soil samples were collected at peak cover crop growth, immediately before termination, and

TABLE 1 | Cumulative precipitation (mm) and GDD (baseline 10C) for each site year divided by duration treatment.

Year	Duration	MN		WI	
		Cumulative precipitation (mm)	Cumulative GDD	Cumulative precipitation (mm)	Cumulative GDD
Y1	LD	6.7	418.9	8.6	451.8
Y1	SD	5.4	249.5	5.0	279.9
Y2	LD	7.7	548.9	6.5	548.5
Y2	SD	5.1	360.0	4.6	332.0

TABLE 2 | Field management schedule for Y1 and Y2 field operations.

Field operation	MN Y1	MN Y2	WI Y1	WI Y2
Long duration CC planting & Baseline soil sample	31-May	5-June	5-June	11-June
Irrigation	5-June	NA	NA	NA
Short duration CC planting	20-June	25-June	24-June	1-July
CC biomass sample & peak growth soil sample	20-July	26-July	25-July	31-July
Broccoli transplant & early decomposition soil sample	27-July	2-August	2-August	7-August
Broccoli fertilization	1-August	10-August	9-August	15-August
Broccoli harvest	5-October	NA	8-October	NA
Broccoli harvest	11-October	NA	13-October	NA
Broccoli harvest	20-October	NA	18-October	NA
Broccoli harvest	26-October	NA	29-October	NA
Broccoli biomass sample	NA	18-November	NA	10-November

at broccoli transplant, and analyzed for labile C and N via a suite of indicators including inorganic and organic N (Wauters, 2020). Soil moisture was measured as volumetric water content (VWC) in three of four site years (MN Y1 and Y2, and WI Y2), and varied from 14 to 30%. Due to inconsistent data collection and resulting lack of replication, statistical comparison between treatments was not possible, though the bare control tended to be on the higher end of the range in Y2 at both sites (data not shown).

Cover Crop Biomass

Immediately prior to cover crop termination, two 0.1 m^2 quadrats of biomass were collected from each plot avoiding the edges, and divided by cover crop species (one or two for each treatment) and weeds (all species combined). Biomass was transferred to a 60°C oven for at least 48 h to achieve constant weight before being weighed for dry biomass yield and then ground and analyzed for C and N content using a dry combustion analyzer (Elementar VarioMax CN analyzer, Elementar Americas). Total biomass N was calculated for each cover crop species via the percentage of N in the biomass. Predominant weed species were noted but not collected separately.

Cash Crop Yield

Broccoli (Gypsy) was planted at 18-inch spacing between plants in four rows per plot (two paired rows on a 5-6 ft bed center), for a total of 80–88 plants per plot. Only the fertilized bare control received fertilizer, which was applied 2 weeks after transplant. Fertilizer was applied as pelletized organic chicken manure. The rate was established based on grower normal side dress fertilization, which was 67 kg ha^{-1} N in MN and 107 kg ha^{-1} N in WI. Due to field error, Y1 MN received 5-2-4 fertilizer in Y1, while MN Y2 and both years in WI received 8-4-4. Weeds were removed via tractor cultivation 2 and 5 weeks after transplant. In Y1, the broccoli was harvested four times between early October and early November from 16 plants from the center of each plot. Broccoli was graded according to USDA market standards 1 & 2, with all other harvestable heads treated as unmarketable yield. Persistent cold after the first frost prevented broccoli harvest in Y2; instead, two immature plants were collected from each plot, dried following the same protocols as for the cover crop biomass, and weighed for aboveground dry biomass.

Statistical Analysis

Total biomass, total biomass N, weed biomass, cover crop C:N, and broccoli yield were all modeled using estimated marginal means (EMMs), on a mixed model in which block was a random effect, and species and duration treatment were fixed effects. Due to interactions, each of the four site years was modeled separately except where noted. EMMs were used to account for imbalances in the data caused by missing samples (one sudex sample from Cala in Y2, and 25 samples across all site years for which there was insufficient biomass to measure CN). Biomass, biomass N, and C:N mean separation were calculated using Tukey's HSD on the mixed model, comparing the four cover crop species within a duration treatment. No bare control was included in these models because the bare treatments were kept biomass free. Pairwise comparison was used to assess differences between LD and SD within a single species, as well as to compare the legume biomass between the two legume-sudex mixtures. Weed biomass as a percentage of total biomass was calculated as a linear, quadratic, and break-point linear regression, with the most significant model chosen for display and discussion. Broccoli total marketable yield from Y1 and broccoli biomass from Y2 were modeled across durations, but separated by location due to different fertilization rates and interactions. Mean separation was calculated using Tukey's HSD on the mixed model with all treatments including the fertilized and unfertilized bare control, as well as on all of the unfertilized treatments compared without the fertilized control. Statistical analysis was carried out using R version 4.0.2, using the *lme4*, *multcomp*, and *segmented* packages for analysis, and *ggplot2* for visuals (Hothorn et al., 2008; R Core Team, 2013; Bates et al., 2015; Wickham, 2016).

RESULTS

Mean total biomass (cover crop + weeds) averaged across duration by site year ranged from 1,890 kg ha^{-1} in MN Y1 to 5,790 kg ha^{-1} in WI Y2. When divided by duration, total biomass varied among species in Y1 for both the SD and LD treatments

(**Table 3**). Specifically, buckwheat in Y1 produced between 37 and 47% more biomass than the next highest treatments in MN, while in WI, buckwheat outproduced the other species by 6%. In Y2, total biomass did not differ among species for either duration, although total biomass was generally higher in Y2 than in Y1. Total biomass in WI was roughly double that of MN for both years. Total biomass production for LD was higher than SD for all species in all site years (**Figure 1A**). Total biomass N contribution followed similar patterns as total biomass among species (**Table 3**). Mean biomass N averaged across duration ranged from 40.6 kg ha^{-1} in MN Y1 to 153 kg ha^{-1} in WI in Y2. Despite large total biomass differences between LD and SD, biomass N was similar across duration, due to higher C:N proportion in LD biomass. Biomass N contribution for SD treatments was equivalent across species in three of four site years (**Table 3**). Total plot biomass C:N varied across species in only one site year (MN Y1) for LD, in which buckwheat had a higher C:N than sunn hemp (**Table 3**).

Total legume biomass was low across most treatments with a mean proportion of legumes:non-legume biomass of 0.07. Comparison of the vetch & sudex and cowpea & sudex mixtures, found that the cowpeas outproduced vetch in MN in Y2 across LD and SD treatments (337 kg ha^{-1} = cowpea, 51.4 kg ha^{-1} = vetch, $p = 0.01$).

Overall biomass C:N for all species in all four site years was higher in LD than SD treatments ($p < 0.05$) (**Figure 1B**). Averaged across treatments, MN cover crop biomass C:N was 20.0 in Y1 and 31.6 in Y2, both of which were higher than the respective C:N in WI, which were 13.9 in Y1 and 15.8 in Y2 ($p < 0.001$).

Buckwheat suppressed weeds most effectively in LD treatments for all four site years, as well as WI Y2 in the SD treatment (**Table 3**). Buckwheat as a proportion of total biomass ranged from 48% in MN Y2 SD to 95% in WI Y1 LD. The sunn hemp LD treatment resulted in less weed suppression than at least one other treatment in all four site years. Sunn hemp biomass as a proportion of total biomass ranged from complete species loss (mean of 0%) in WI Y2 LD to 16% in MN Y1 SD. Across species, weed biomass as a proportion of total biomass decreased as total biomass increased to 2,169 kg ha^{-1} (adjusted $r^2 = 0.64$, $p < 0.001$), at which point weed biomass was approximately 25% of total biomass (**Figure 2**). There was no significant relationship between total biomass and weed proportion of biomass beyond 2,169 kg ha^{-1} (adjusted $r^2 = 0.019$, $p = 0.192$).

No cover cropped treatment produced a vegetable yield equivalent to that of the fertilized control treatment in Y1 (**Table 4**). When the fertilized treatment was removed from the model for comparison of the four species and unfertilized bare treatments, yield in the bare control was 30% higher than any cover cropped treatments in MN ($p = 0.062$), and 26% in WI ($p = 0.096$). Mean vegetable yield across all cover crop species treatments (without the bare control) in MN was 2,230 and 8,380 kg ha^{-1} in WI. Duration was marginally significant ($p = 0.061$), with a mean yield of 3,660 kg ha^{-1} in the SD treatment and 2,850 kg ha^{-1}, in the LD treatment.

TABLE 3 | Total cover + weed biomass, weed biomass, total biomass N, and total biomass C:N for all four sites years, separated by duration and species treatment.

			MN				WI			
			Total biomass	Weed biomass	Total biomass N	C:N	Total biomass	Weed biomass	Total biomass N	C:N
Year	Duration	Species	kg ha^{-1}				kg ha^{-1}			
Y1	SD	Buckwheat	2,320 a	709	55 a	17	3,290 ab	505 b	114	11.5
Y1	SD	Vetch & sudex	1,570 b	1,420	46.7 ab	15.6	3,110 ab	1,580 ab	121	10.6
Y1	SD	Cowpea & sudex	1,020 c	867	30.6 b	17.5	3,550 a	2,330 a	129	12.6
Y1	SD	Sunn hemp	971 c	820	32.8 b	15	2,220 b	1,970 ab	95.1	9.73
Y1	LD	Buckwheat	3,170 a	398 b	47.5	26.4 a	7,320 a	362 c	180 a	15
Y1	LD	Vetch & sudex	2,320 ab	1,510 a	41.8	23.1 ab	6,930 ab	2,060 b	161 ab	16.6
Y1	LD	Cowpea & sudex	1,990 b	1,410 a	36.5	25.1 ab	5,400 bc	3,020 ab	122 bc	18
Y1	LD	Sunn hemp	1,770 b	1,730 a	33.9	20.5 b	4,470 c	4,370 a	112 c	17.1
Y2	SD	Buckwheat	1,790	1,020	33.7	25.2	3,760	530	140	9.82
Y2	SD	Vetch & sudex	947	739	21.7	20.3	3,730	2,090	138	11.1
Y2	SD	Cowpea & sudex	1,810	1,600	27.1	26.6	3,160	751	116	11.4
Y2	SD	Sunn hemp	1,290	1,260	25.2	21.3	2,100	1,600	86.3	11
Y2	LD	Buckwheat	4,910	1,540 b	59.4 a	36.9	8,540	1,490 c	202 ab	16.7 c
Y2	LD	Vetch & sudex	3,750	2,880 ab	43.1 ab	40.5	10,400	3,730 b	236 a	24.9 a
Y2	LD	Cowpea & sudex	4,040	2,470 ab	46.3 ab	41.7	6,380	3,250 bc	128 b	23.9 ab
Y2	LD	Sunn hemp	3,860	3,840 a	35.0 b	40.6	8,270	8,260 a	198 ab	17.5 bc

Lowercase letters indicate significant differences among the four species treatments within a duration treatment for a single site year. All means are estimated marginal means, to account for missing data. Mean separation via Tukey's HSD ($p < 0.05$).

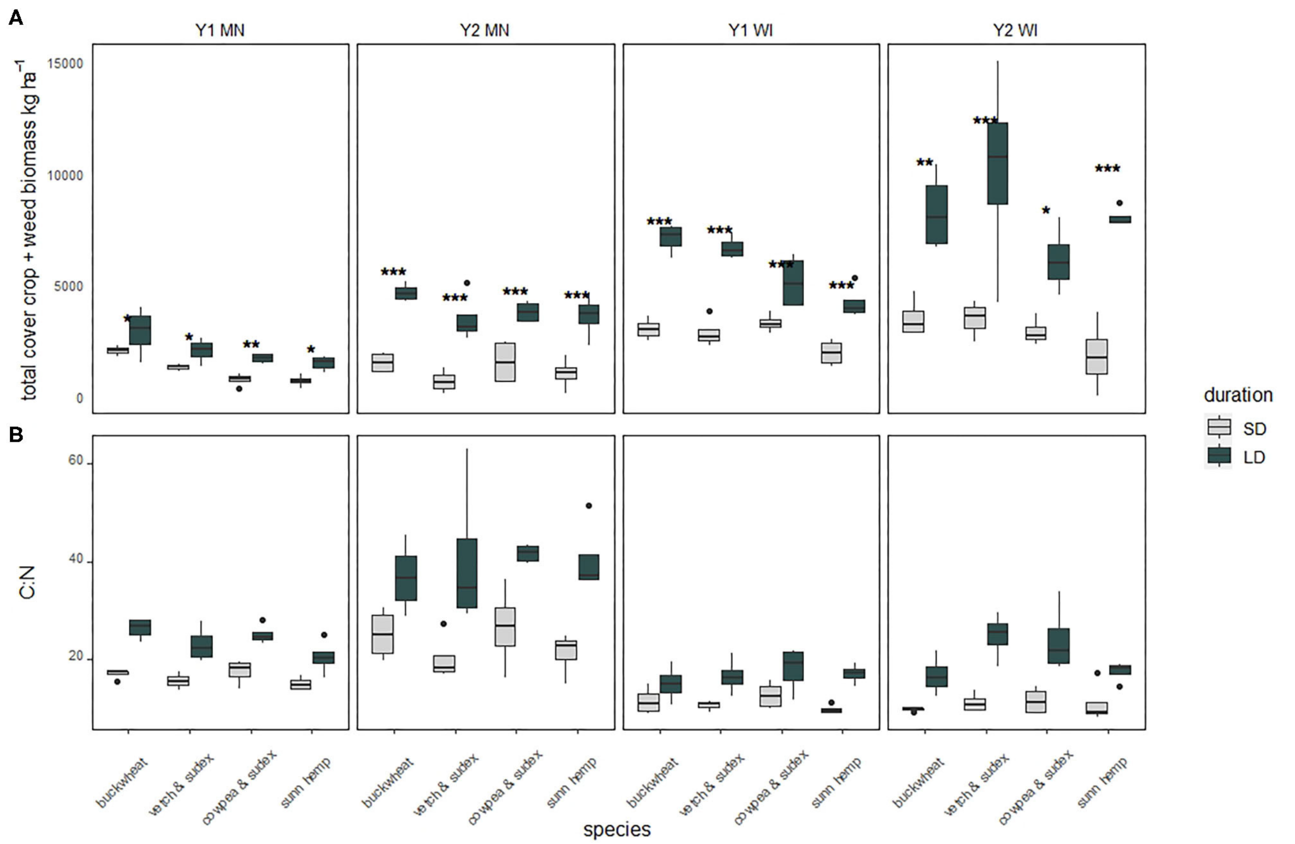

FIGURE 1 | Total biomass **(A)** and biomass C:N **(B)** from each site year divided by species, compared between durations. Short duration (SD) appear on the left of each pair (lighter shading), long duration (LD) on the right (darker shading). Significant differences between duration for a species are indicated by stars above the pair. $^{*}p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$.

Marketable yield as a percentage of total yield in Y1 differed among treatments within locations. In MN, cover crop duration did not affect marketable yield. Among species, broccoli plants in cowpea & sudex treatments produced a lower percentage of marketable yield than the unfertilized bare control treatment, 10 and 43% of total yield, respectively ($p < 0.001$). Marketable broccoli yield in MN from all unfertilized treatments did not match the percentage of marketable yield from the fertilized treatments (89%). In WI, SD treatment had overall higher percentages of marketable yield than LD, 72 and 83%, respectively ($p = 0.04$). The percentage of marketable yield from all cover cropped treatments was below that of the fertilized control (fertilized control = 94%), though the difference was only significant for sunn hemp (67%, $p = 0.03$). When comparing cover crop treatments without the bare treatments, MN had lower marketable yield than WI, and the SD treatment had higher marketable yield than LD.

Broccoli yield data for Y2 is not included in **Table 4** due to persistent cold after the first frost, which prevented broccoli plants from reaching full maturity. Dry biomass of plants in MN was higher in the fertilized treatment than in the unfertilized treatments (fertilized = 97.5 g, mean unfertilized = 40.86 g, SE = 8.4), but equivalent across all treatments and controls in WI (**Table 4**).

DISCUSSION

In this study we demonstrated that cover crops grown for short periods in the summer could provide supporting and regulating ecosystem services though high biomass accumulation, but they may do so at the expense of fall cash crop yield. Ecosystem service tradeoffs have been well-established for cool-season cover crops in field cropping systems, with greater N retention associated with decreased ability to provide fertility to the system (Finney et al., 2017). In vegetable systems, summer cover crops are often grown for >2–3 months (Boyhan et al., 2016; Stute and Shekinah, 2019), which can assure high biomass productivity but is longer than many growers can afford to take away from spring and fall cash crop production. In this study, we focused on 30 and 50 growing days, to fit the cover crops into realistic cool-season vegetable rotations of northern climates (USDA plant hardiness zones 1-4). Despite the brief growing period, buckwheat and sudex in both mixes accumulated biomass commensurate with that of more temperate climates (Creamer and Baldwin, 2000;

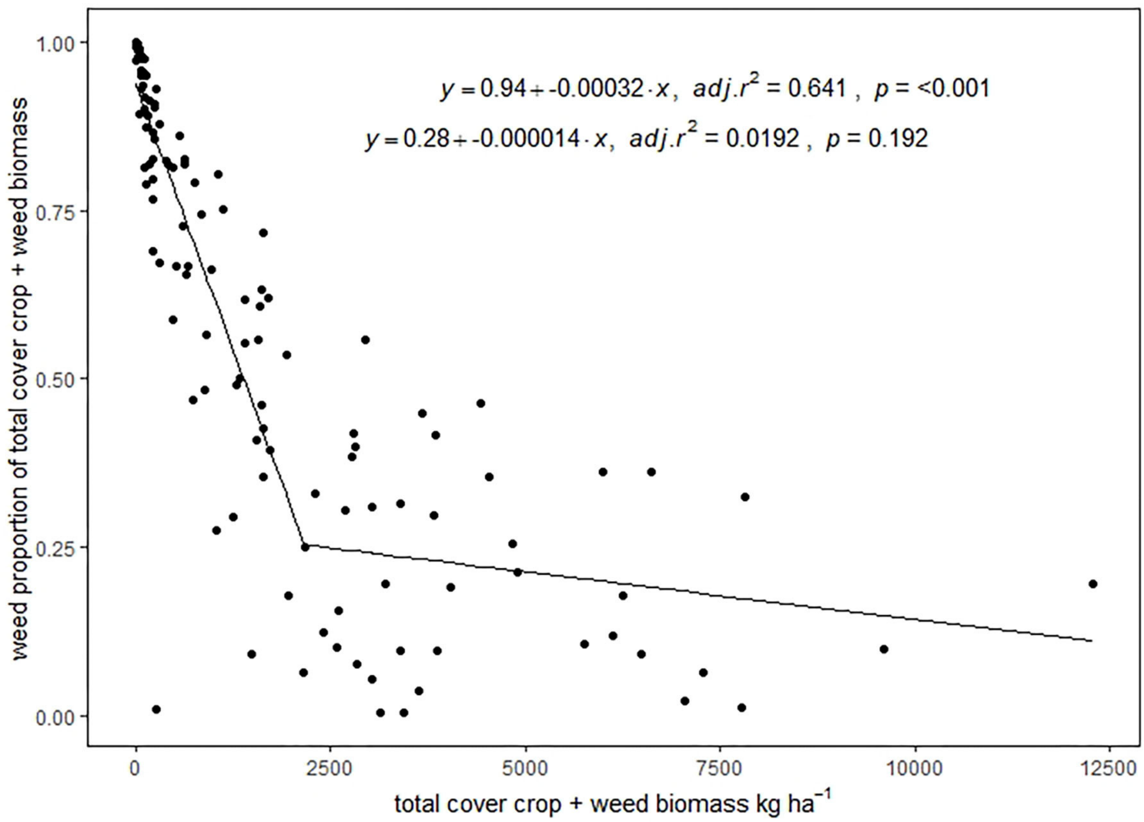

FIGURE 2 | Relationship between total biomass and percentage of biomass from weeds. Best fit model chosen from linear, quadratic, and linear plus plateau, with equations for each of the two lines. The top equation represents the line from 0 to 2,169 kg ha^{-1}, and the bottom equation represents the line >2,169 kg ha^{-1}.

TABLE 4 | Fall cash crop yield in Y1 and plant biomass in Y2 by location for each species, averaged over duration in the absence of interaction effects.

	Species treatment	MN	WI
Y1		**kg ha^{-1} yield**	
	Bare fertilized control	7,181 a	11,832 a
	Bare unfertilized control	3,429 b	10,987 ab
	Vetch & sudex	2,635 b	7,746 b
	Sunn hemp	2,531 b	8,649 ab
	Buckwheat	2,033 b	8,731 ab
	Cowpea & sudex	1,738 b	8,376 ab
Y2		**mg dry plant biomass**	
	Bare fertilized control	97.5 a	183.8
	Bare unfertilized control	55 b	204.1
	Vetch & sudex	41.7 b	183.5
	Sunn hemp	37.2 b	154
	Buckwheat	35.4 b	109.1
	Cowpea & sudex	35 b	147

Crop yield for Y1 and biomass for Y2 calculated via estimated marginal means (EMMs). Lower case letters indicate mean separation via Tukey's HSD, $p < 0.05$. Standard error (SE) for MN Y1 was 423 kg ha^{-1}, and for WI Y1, 1380 kg ha^{-1}.

O'Connell et al., 2015). The biomass potential of cover crops after a short period of growth makes them a viable option to enhance the supporting ecosystem services by replacing bare fallows and adding organic matter (Smuckler et al., 2012).

Biomass differences between the two sites were higher than expected and are best explained by variation in soil type, fertility, and water availability. The MN and WI sites were chosen to test the effects of summer cover crops on distinct soil quality circumstances; the Braham loamy sand soil of MN had 11 g kg^{-1} SOM and requires summer irrigation despite regular summer precipitation. The WI site was a Crystal silt loam with 23 g kg^{-1} SOM on which the farmer had never used irrigation. While cover crop performance was predicted to differ between sites, the contrast in cover crop performance was beyond expectation. Low biomass accumulation in MN persisted despite irrigation at germination to offset lower water holding capacity in the sandy soil. The added irrigation resulted in similar VWC across the two sites, indicating that low soil moisture was not the main determinant of biomass accumulation (data not shown). Low biomass suggests that, in some instances, cover crops may not be able to provide desired ecosystem services. Benefit may be more likely with fertilization. Pairing cover crops with fertility sources is not uncommon. Over-wintered cover crops are often planted in synchrony with fall manure application, such that the cover crop can prevent nutrient leaching from manure (Cambardella et al., 2010; Thilakarathna et al., 2015). Applying fertilizer specifically for cover crop success is mentioned in farmer-focused publications (Clark, 2013), but is lacking in academic literature. While cover crops are touted as a tool for improving poor soil, our results suggest that there may be thresholds of soil OM, available N, or water content below which cover crops cannot produce sufficient biomass to provide SOM-building benefits unless coupled with synchronous fertilizer or irrigation.

Circumstances in which cover crops may require fertilization highlights one of the potential limitations for their use. This has been observed for conservation agriculture practice in the highly eroded soils of sub-Saharan Africa, where higher input costs are often a necessary pre-condition to implement conservation practices. This requirement excludes cover crops as an option for the poorest farmers, even though these farmers might be farming soils that need the conservation strategy most (Giller et al., 2009). The relationship between cover crop growth and fertility requirements suggests a need for targeted cover crop experimentation in water-limited, highly eroded, and sandy soils to determine the conditions in which diversification via cover crops can deliver ecosystem services such as weed suppression and SOM contribution and when they may result in untenable tradeoffs.

The duration for which cover crops were grown drove differences in biomass accumulation in three of four site years. This may have been the result of insufficient GDD accumulation for the short duration treatments. Studies of summer and winter cover crops point to the importance of GDD in determining plant growth (Brennan and Boyd, 2012; Baraibar et al., 2018; Stute and Shekinah, 2019), where lower GDD DAP^{-1}(cumulative GDD divided by DAP) is correlated with lower overall growth (Brennan and Boyd, 2012). The 30-year average GDD DAP^{-1} in MN between June and October is 16.9 (baseline 10C) (UMN, 2019). However, the short summer season in northern regions (USDA Plant Hardiness Zones 1-4) may not provide sufficient GDD for cover crops planted between vegetable crops. Our results for GDD DAP^{-1} remained below the full season average, ranging from 8.3 to 10.9, such that SD treatments did not have sufficient GDD to reach their accumulation potential. Summer cover crops grown for any duration may be more successful when planted later in the season, after a long spring crop, to take more advantage of GDD during the late summer.

Biomass productivity was also heavily dependent on species treatment, indicating the importance of appropriate species selection for specific services. Legumes were included in the study for their potential to fix nitrogen and contribute fertility. However, legume biomass was notably low, limiting the potential for N fixation and associated fertility benefits. The proportion of legume in the total harvested biomass for each of the three legume species treatments (cowpea & sudex, vetch & sudex, and sunn hemp) ranged from 0 to 0.5, and the mean proportion of legumes as part of total cover crop biomass was only 0.07. Seeding rates in the sudex bicultures may have contributed to low legume biomass. Others have found that a legume-sudex mix planted 50–50% by seed weight resulted in biomass that was 85% grass and 15% legume (Stute and Shekinah, 2019). Our seeding rates were roughly 50–50% for the cowpea-sudex mix (44.8 and 42.6 kg ha^{-1}), and due to large seed size, the vetch-sudex mix was 63–36% (75 and 42.6 kg ha^{-1}), suggesting that higher seeding rates are necessary to encourage legume productivity, both in absolute terms and as a proportion of the mixture. Future studies should examine chickling vetch and sunn hemp under more optimal conditions. Chickling vetch has demonstrated high potential as a cover crop in drought and high salinity areas (Lambein et al., 2019), and a high potential for N fixation (Büchi et al., 2015). Sunn hemp also has demonstrated high potential for biomass production that was not achieved in this study; this may have resulted from low soil temperatures at planting, although sunn hemp can be planted any time after the final spring frost (Schonbeck and Morse, 2006).

Because of notable biomass differences, cover crop species differed in the ecosystem services provided. Negative cover crop effects on following cash crops as observed here have been established as a possible disservice from sorghum-sudangrass (Kruse and Nair, 2016). The current study did not provide evidence for species-specific detrimental effects, and thus suggests that the cover crop presence, perhaps because of nutrient and water use during growth, or slow decomposition, led to the yield penalty. Lower nutrient availability was confirmed via soil nitrate measurements, which exhibited values 10–13 times higher in bare soil than in cover crop treatments in WI, and 120–180% higher in the bare control than in cover crop treatments in MN (Wauters, 2020). Given sorghum-sudangrass' high rate of biomass accumulation, it may not be suitable as an immediate precursor to fall vegetables, despite its potential to contribute to ecosystem benefits such as building SOM and physically protecting soil from erosion (Finney et al., 2016). Of all species, buckwheat, which is already a common summer cover crop (Bulan et al., 2015), provided the most consistent combination of weed suppression and growth. Because of its added potential benefit to pollinators (Clark, 2013), the success of buckwheat also indicates that it may also be useful to focus on non-fertility benefits of cover crops during short periods in the summer.

Weed suppression services of cover crops are important insofar as they prevent weed seed maturation and subsequent replenishment of the weed seed bank, or as their allelopathic effects inhibit weed growth following cover crop termination. The low weed suppression capability of most cover crops in this study is of concern because some of the most common weeds observed in these systems, including *Portulaca oleracea*, *Amaranthus retroflexus*, and *Chenopodium album* have the potential to produce viable seed in as little as 6–8 weeks (Bassett and Crompton, 1978; Miyanishi and Cavers, 1980; Weaver and McWilliams, 1980). While it is probable that most of the weeds in this study were unable to set seed, hard seed from *Chenopodium album* was observed at the MN site, which had higher overall cover crop C:N, suggesting that plants matured more quickly in the sandy, low OM soil, perhaps due to water stress (Turner, 1986). The risk of even a single weed going to seed can be significant. For example, weed seed production from *Amaranthus* species in the presence of poor-competing cover crops can top 100,000 seeds m^{-1} (Brainard et al., 2011). While use of weeds as a cover crop may provide some of the same benefits as a planted cover crop (Wortman, 2016), the inverse relationship between total treatment biomass and weed biomass found in this study indicates that the benefits from biomass accumulation are better achieved via a planted cover crop. However, the lack of a weedy control in this study limits our ability to make predictions on how the cover crops would have compared to an unmanaged weedy fallow. Given that the biomass accumulation in the most and least weedy cover crop plots had similar effects on fall broccoli that differed from the bare control, we would expect that a true weedy control would have led to decreased broccoli yield. Future research comparing the effects of varied levels of weed management, such as a bare control, limited cultivation, and no cultivation, could help clarify the impacts of weedy summer growth on fall cash crop yield.

Cover crop maturity stage has important effects on biomass N mineralization and immobilization rates after termination. In our study, broccoli yield decreased in all cover crop treatments, suggesting that nutrients may have been immobilized by the cover crops and thus became unavailable for cash crop uptake. Cover crop C:N at termination determines the availability of cover crop nutrients to microbes, and thus affect the ecosystem services related to N retention and fertility (Finney et al., 2016). While some evidence suggests that C:N of 24:1 is ideal for microbial processing and nutrient release without immobilization (O'Connell et al., 2015), comparisons of legume residue with C:N $<$ 20 found differences in the rate of N release, indicating that immobilization can occur at lower C:N (Wagger et al., 1998). Modeling N release from biomass residue in soil found that C:N was positively correlated with N release until 11:1, and then decreased, indicating that immobilization may be a significant factor in nutrient availability well-below the 24:1 threshold (Clivot et al., 2017). In the current study, C:N was reliably below 24:1 in WI, but mineral N remained significantly lower in cover cropped treatments compared to bare control at broccoli transplant, which occurred 1 week after cover crop termination and incorporation (Wauters, 2020). The decreased N levels indicate nutrient immobilization, which may have contributed to decreased broccoli yield. Furthermore, vegetable yield was generally lower in LD treatment, especially at MN. In Y2, C:N in the LD treatment reached over 40, well-above an ideal range for microbial mineralization. Mineral and organic N in soil at peak growth and early decomposition time points were lower in the cover crop treatment than in the bare treatments (Wauters, 2020), indicating that the living and decomposing biomass both led to decreased N availability for the broccoli. Despite high soil moisture and temperature, decomposition processes did not release nutrients for the fall cash crop in time to avoid a yield penalty. The suggested window between cover crop termination and cash crop planting varies from 2 to 3 weeks (Clark, 2013), though others have found that N release can take place over multiple weeks and even months (Parr et al., 2014). In this study, the time between cover crop termination and cash crop planting was only 1 week, to improve the probability that the broccoli would mature before first frost and thus be able to withstand some freezing temperatures. Given the reduction in yield, 1 week appears to be insufficient. Additionally, the broccoli only reached maturity and was able to form heads in 1 of 2 years before persistent low temperatures arrested growth, indicating that both a spring and fall cash crop on either side of the cover crop is not feasible.

Summer cover crop use in northern climates could be a useful tool for vegetable growers seeking to protect and improve soil within complex rotations, especially in the northern climates experiencing an increase of extreme summer rain events that could erode bare soil. However, weed pressure and cash crop yield decreases remain significant barriers to adoption. In the MN soil, which had very low OM and less soil water holding capacity, the yield decrease between cover cropped and bare unfertilized treatments indicates that fall cash crop planting carried particularly high risk of reducing cash crop yield. In the higher OM soils in WI, the broccoli yield decrease was less dramatic, despite the bare control receiving more fertilizer than in MN; it would be worthwhile to quantify the cost of the tradeoffs between yield and the ecosystem services provided by the cover crop. Weed pressure can be reduced by summer cover crops, but not eliminated. These cover crops show potential for farmers, but care must be taken to integrate them into the system with enough time to reach maturity and decompose without impinging on cash crop growth. Demonstrating the benefits and limitations of cover crops as a diversification tool to enhance ecosystem services and resilience provides farmers with a clearer picture of how summer cover crops could be used in their operation, to respond to the multi-layered demands of food production and environmental stewardship to which farmers must continuously adapt.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

VW, JG, AP, and RC contributed to conception, design of the study, conducted field work, and data collection. VW and JG performed the statistical analysis. VW wrote the first draft of the manuscript. All authors contributed to manuscript revision, read, and approved the submitted version.

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frontiers in Sustainable Food Systems

REVIEW
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Narrow and Brittle or Broad and Nimble? Comparing Adaptive Capacity in Simplifying and Diversifying Farming Systems

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Humanity faces a triple threat of climate change, biodiversity loss, and global food insecurity. In response, increasing the general adaptive capacity of farming systems is essential. We identify two divergent strategies for building adaptive capacity. *Simplifying* processes seek to narrowly maximize production by shifting the basis of agricultural production toward centralized control of socially and ecologically homogenized systems. *Diversifying* processes cultivate social-ecological complexity in order to provide multiple ecosystem services, maintain management flexibility, and promote coordinated adaptation across levels. Through five primarily United States focused cases of distinct agricultural challenges—foodborne pathogens, drought, marginal lands, labor availability, and land access and tenure—we compare simplifying and diversifying responses to assess how these pathways differentially enhance or degrade the adaptive capacity of farming systems in the context of the triple threat. These cases show that diversifying processes can weave a form of broad and nimble adaptive capacity that is fundamentally distinct from the narrow and brittle adaptive capacity produced through simplification. We find that while there are structural limitations and tradeoffs to diversifying processes, adaptive capacity can be facilitated by empowering people and enhancing ecosystem functionality to proactively distribute resources and knowledge where needed and to nimbly respond to changing circumstances. Our cases suggest that, in order to garner the most adaptive benefits from diversification, farming systems should balance the pursuit of multiple goals, which in turn requires an inclusive process for active dialogue and negotiation among diverse perspectives. Instead of locking farming systems into pernicious cycles that reproduce social and ecological externalities, diversification processes can enable nimble responses to a broad spectrum of possible stressors and shocks, while also promoting social equity and ecological sustainability.

Keywords: diversified farming systems, marginal land, land access, farm labor, food safety, drought, adaptive capacity, equity

INTRODUCTION

Climate change, biodiversity loss, and global food insecurity present an Anthropocene triple threat for humanity (Kremen and Merenlender, 2018). The current global agrifood system contributes to the triple threat by emitting 23% of global greenhouse gases (IPCC, 2019), reducing biodiversity (Dainese et al., 2019), displacing traditional foodways and knowledge (Altieri, 1999; Hoover, 2017; White, 2017), and contributing to the decline of rural communities (Carolan, 2016). Although farmers have always dealt with climatic, ecological, socioeconomic, and political challenges that test their ability to continue farming, these long-standing "normal" challenges will be transformed, predominantly for the worse, by the novel shocks and stressors emanating from the triple threat (**Figure 1** and **Box 1**). These threats and challenges partly arise from and are exacerbated by the well-known social and environmental externalities generated by industrialized agricultural systems (Kremen and Merenlender, 2018). In order to reduce social inequity and environmental destruction, and adapt to an increasingly uncertain future, there is growing consensus that our agricultural system must undergo systemic, transformative change (McIntyre et al., 2010; International Panel of Experts on Sustainable Food Systems, 2018; IPCC, 2019). Transformation can occur rapidly, or can emerge from incremental progress along context-specific transition pathways

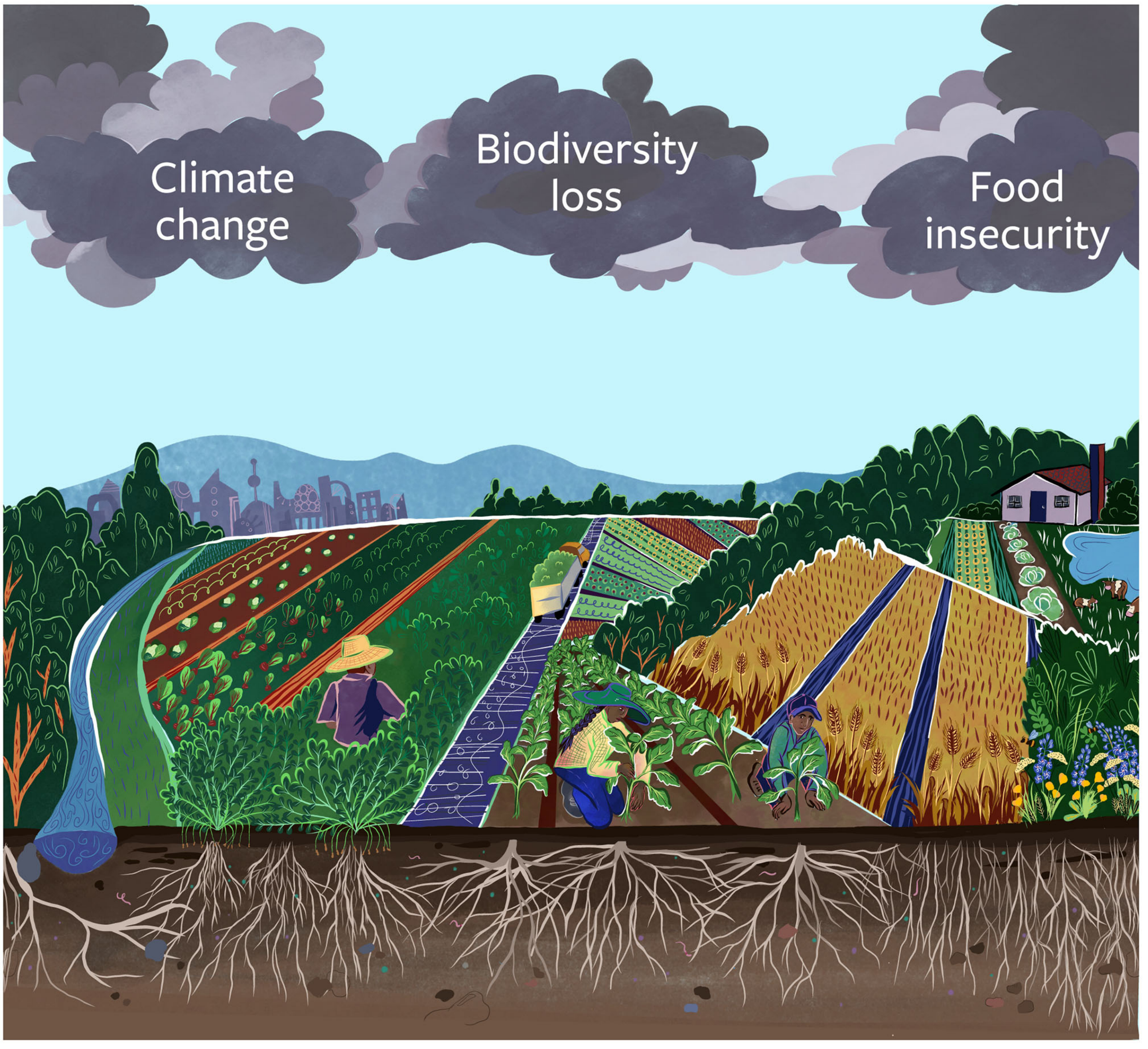

FIGURE 1 | The Anthropocene "triple threat"—climate change, biodiversity loss, and global food insecurity—interact in ways that will exacerbate long-standing climatic, ecological, socioeconomic, and political challenges for agriculture and food systems. The process through which social-ecological systems build adaptive capacity to these looming threats shapes future adaptation and transformation possibilities.

BOX 1 | Definitions of key terms.

For the purposes of this paper, we define several terms to distinguish between different types of pressures that influence farming systems:

- *Threats* operate at a global scale to drive change in the focal social-ecological system, often exacerbating *challenges*
- *Stressors* and *shocks* are temporally or spatially localized manifestations of threats. *Stressors* are persistent changes in "slow variables" that create gradual or chronic pressure on farming systems over time, while *shocks* are sudden changes in "quick variables" that create acute pressure on farming systems
- *Challenges* are "normal" pressures faced by agriculturalists, such as accessing enough water, land, and labor to produce crops

Response to these pressures and their impacts can, broadly speaking, form two divergent pathways toward adaptive capacity:

- *Simplifying* processes seek to narrowly maximize production by shifting the basis of agricultural production toward centralized control of socially and ecologically homogenized systems
- *Diversifying* processes cultivate social-ecological complexity in order to provide multiple ecosystem services, maintain management flexibility, and promote coordinated adaptation across levels
- *Processes* comprise the knowledges, strategies, practices, policies, and technologies that can alter farming systems across levels
- *Pathways* describe the directionality and continuity of a suite of processes that are situated in broader social and ecological contexts, wherein historical actions shape future possibilities

that influence *adaptive capacity* (**Box 1**), or the ability to respond flexibly and effectively to changing circumstances (Wilson, 2007; Tomich et al., 2011; Anderson et al., 2019; Chhetri et al., 2019). The scale and scope of response matters, and varies along a spectrum ranging from simply coping with the impacts of shocks and stressors in the moment to re-imagining and reconfiguring the structural conditions and drivers that give rise to those shocks and stressors (Van Noordwijk et al., 2020). As such, farmers, agricultural service providers, academics, and policymakers must consider not only how transformation pathways address present social and environmental problems, but also how they build, do not build, or undermine the capacity to adapt to rapidly changing and unexpected biophysical and social challenges into the future.

Many proposed approaches to increase adaptive capacity are goal-oriented, without an explicit focus on *process*. For example, "climate-smart agriculture" defines clear goals— to simultaneously increase yields, improve resilience to climate change, and reduce greenhouse emissions (Lipper et al., 2014)—yet appears agnostic with regard to the pathway taken to achieve those goals. Sustainable intensification similarly emphasizes optimizing stable productivity, which could be pursued through many routes ranging from narrowly increasing input use efficiency to completely redesigning agricultural systems (Campbell et al., 2014; Pretty, 2018). Approaches like these, which isolate the ends from the means, fail to differentiate how all the tools in the toolbox function socially, and avoid crucial processual questions: How do various strategies differentially distribute benefits and costs of adaptation? In what ways do different processes influence vulnerability or resilience of social and ecological functions beyond farm productivity? Who controls access to these tools, and who is excluded? No approach to adaptation can constitute a coherent strategy without also addressing these questions.

To address this gap, we propose a process-oriented approach to adaptation rooted in strategies of diversification. Drawing upon the paradigm of agroecology, the theory of diversified farming systems (DFS) proposes that adoption of biodiversity-enhancing practices can increase the magnitude and stability of ecosystem services and simultaneously reduce or eliminate the need for external inputs, reduce negative externalities, and increase positive on-farm outcomes (Kremen et al., 2012; Rosa-Schleich et al., 2019). Networking experiential and scientific knowledge helps farmers flexibly employ different suites of management practices to fit their situated goals and constraints (Vandermeer and Perfecto, 2017). Recent interdisciplinary scholarship further links DFS success to the diversity of disciplines, practitioners, markets, ideas, and cultures in farming systems through the knowledge density required to productively manage biodiversity in a specific place and time (Timmermann and Félix, 2015; Dumont et al., 2016; Carlisle et al., 2019a). Other scholarship has also synthesized how the diversification transition can happen, and the barriers and opportunities that exist across different institutional scales (International Panel of Experts on Sustainable Food Systems, 2016, 2018).

We expand upon the DFS framework in two important ways. First, while previous DFS work focused on biodiversification and managerial diversification at farm and landscape levels, we weave in further dimensions of cultural, economic, epistemic, and organizational diversity across multiple social relational levels from the farm enterprise to national policies (Carlisle, 2014). Thus, in addition to indicators in genetic, crop, ecosystem, functional, and managerial diversity, we include diversification of societal goals, market channels, governance arrangements, knowledge production infrastructures, and social networks. Second, we distinguish the state of being diversified from the process of diversification, which represents iterative socio-ecological transition pathways and can broaden individual and collective participation in sustainable adaptation irrespective of scale or starting point.

This paper frames a Research Topic on Diversifying Farming Systems for Adaptive Capacity in *Frontiers in Sustainable Food Systems* by: (a) Briefly reviewing the ways that climate change, biodiversity loss, and food insecurity impact the adaptive capacity of agricultural systems; (b) Contrasting the implications for equity and sustainability of diversifying as opposed to simplifying processes for building adaptive capacity; (c) Analyzing these processes through five cases that exemplify ongoing challenges compounded by the triple threat; and (d) Presenting a novel framework to explore how diversifying processes influence adaptive capacity to shocks and stressors emanating from the triple threat. Moreover, our framework explicitly considers whether and in what ways diversifying pathways can lead to the emergence of different qualities of adaptive capacity that also enhance sustainability and equity more broadly.

BACKGROUND

The Anthropocene Triple Threat

Climate change already impacts farmers around the world. The increase in frequency and intensity of weather extremes (e.g., droughts, heat waves, hurricanes, and floods), together with the spread of novel diseases and pests, altered growing seasons, and fewer chill hours, reduces crop yields in many regions and increases environmental degradation such as nitrogen pollution and soil erosion (Bowles et al., 2018; IPCC, 2019). The complex interaction of acute shocks and chronic stressors produces both discrete events that can lead to abrupt agroecosystem collapse, like widespread crop failure, and damage to farming infrastructure, as well as continuous deterioration that gradually undermines productivity and resilience to acute shocks (Tomich et al., 2011). Directly and indirectly, climate change also impacts livelihoods by inducing rural migration, reducing food security, worsening inequalities, and spurring resource conflict, to name a few examples (Hsiang et al., 2013; Burrows and Kinney, 2016; Nawrotzki et al., 2017; Jha et al., 2018). Although mitigation remains critical, adaptation imperative. In order to respond to ongoing and future climate change already incurred from past emissions, farmers must find long-term solutions suitable to new climatic norms (Steffen et al., 2018; IPCC, 2019).

Biodiversity is rapidly declining across the globe (Dainese et al., 2019; IPBES, 2019), altering ecosystem functions and jeopardizing ecosystem services that are essential for human well-being (Hooper et al., 2005). The alarming rate of species loss through extinction is compounded by dramatically declining biomass of taxa like insects and birds (Hallmann et al., 2017; Wagner, 2020). Some of the primary drivers of biodiversity loss are habitat loss and fragmentation, as well as chemical pollution from industrial agriculture (Dainese et al., 2019). Global agricultural simplification has also eroded crop genetic diversity, which is critical for adaptive crop breeding (Jackson et al., 2013; Veteto and Carlson, 2014; Zimmerer and de Haan, 2017) and productivity in marginal environments (Altieri, 1999). Some studies have shown that certain ecosystem services can persist with merely a few species under ideal conditions (Kleijn et al., 2015). But many more species are required when considering additional services, larger spatial or temporal scales, and variable environments (Kremen, 2005; Isbell et al., 2011, 2017; Reich et al., 2012). Increasing biodiversity in agricultural landscapes can help these systems maintain multiple critical functions, such as pest control and protection of water quality, in the face of climate change (Bowles et al., 2018; Kremen and Merenlender, 2018).

Confronting these momentous environmental changes (**Figure 1**), it is essential to produce food in ways that sustainably and equitably assure the basic human right to food (De Schutter, 2011). Globally, two billion people experience moderate or severe food insecurity, including uncertainty about obtaining food and compromising quality or quantity of food consumed, a number that is rapidly rising with the COVID-19 pandemic (FAO, 2020). Healthy ecosystems and rural livelihoods are integrally linked to food security (Chappell, 2018). While strategies such as sustainable intensification focus on maximizing productivity and reducing environmental externalities (Garnett et al., 2013; Rockström et al., 2017), they fail to address the underlying inequities that cause food insecurity and the ways in which capital-intensive "solutions" exacerbate social and ecological vulnerabilities (International Panel of Experts on Sustainable Food Systems, 2018), which perversely undermines the human right to food. Food insecurity is an issue of access, not production. The world currently produces enough food to feed all of humanity (Patel and Moore, 2017; Chappell, 2018), but a large portion is either wasted, used for animal feed, or used to manufacture non-food products such as biofuels (Cassidy et al., 2013). Globally, access to food continues to be grossly unequal (Patel and Moore, 2017), and food insecurity is linked to the erosion of agricultural sovereignty, local foodways, experiential knowledge, and farming livelihoods, as well as land degradation, particularly in many regions of the Global South (Altieri and Toledo, 2011; Wittman, 2011; Edelman, 2014). To fully realize the human right to food, agricultural systems must maintain critical ecosystem services while also meeting the intertwined challenges of access, adequacy, acceptability, appropriateness, and agency (Chappell, 2018).

The triple threat of climate change, biodiversity loss, and global food insecurity intersect to exacerbate the challenges farmers and ranchers already face (**Table 1**). For example, climate change increases the intensity and frequency of droughts, while diminished biodiversity limits ecological management options to cope with drought, and the combined effects ripple and magnify through synchronized markets, reducing global food security.

Defining Adaptive Capacity

Adaptive capacity is the ability to adapt to changing circumstances (Engle, 2011). In much of the literature, adaptive capacity is used specifically in the context of climate change (McLeman and Hunter, 2010; Liverman, 2015), but the concept also accommodates other types of change. Adaptive capacity, vulnerability, and resilience are highly interrelated concepts (Gallopín, 2006) that all describe how changes affect a system in terms of susceptibility and responses to change. The vulnerability of a system to a particular stress or shock is widely accepted to be a function of (1) the *sensitivity* and *exposure* of that system to the perturbation and (2) the *response capacity*, described as the system's ability to cope, resist, adapt, recover, or take advantage of the opportunities arising from the consequences of the perturbation (Smit and Wandel, 2006). Adaptive capacity is sometimes seen as interchangeable with response capacity (Smit and Wandel, 2006; IPCC, 2019), but others recognize adaptive capacity as a broader concept (Gallopín, 2006), since specific adaptations may actually influence the sensitivity or exposure of a system to particular perturbations, or increase a system's resilience (Walker et al., 2004). A concept with roots in ecology, resilience has traditionally been defined as the extent to which systems can absorb a perturbation while remaining in, or returning to, a state with essentially the same structure, function, identity, and feedbacks (Gunderson and Holling, 2001; Walker et al., 2004; Folke, 2006). Resilience has been extended to include possibilities of transformation to other stable states with

TABLE 1 | Increased stresses from, and potential diversifying adaptations to, the triple threat for each of the five cases.

Cases	Climate change	Biodiversity loss	Food insecurity
Foodborne pathogens	*Stress*: Expansion in range of disease vectors and increased pathogen growth/survival *Adaptation*: Crop rotations and polyculture practices to flexibly shift spatial and temporal distribution of risks	*Stress*: Reduced biodiversity may increase disease transmission *Adaptation*: Cultivate ecosystem services to suppress or attenuate pathogens in the farm environment	*Stress*: Year-round demand further centralizes distribution, magnifying risk *Adaptation*: Localize production/distribution systems to create sustainable livelihoods and reduce the magnitude of outbreaks
Drought	*Stress*: Greater intensity/frequency of droughts *Adaptation*: Crop diversity to mitigates risks through portfolio effect and increases water capture, storage, and productive use though improved soil health	*Stress*: Reduced crop and livestock diversity limits options for adaptive breeding *Adaptation*: Protect wild relatives and traditional genotypes and promote locally-adapted varieties/breeds	*Stress*: Highly-specialized systems geared toward commodity production lead to synchronized shocks *Adaptation*: Diversify food crop portfolios from local to national levels
Marginal land	*Stress*: Further degrading lands due to variable weather patterns, increased drought, and decreased soil health *Adaptation*: Diversification techniques coupled with landscape modification increase land resilience, restore degraded soil, and increase ecosystem functions	*Stress*: Reduction in biodiversity can exacerbate land degradation on already marginally lands *Adaptation*: Practices like intercropping, agroforestry, and silvopastoralism provide wildlife habitat, soil fertility, and increase response diversity	*Stress*: Simplified farming systems aimed at commodity production *Adaptation*: Diversifying practices improve marginal soil productivity and mitigate disturbances while supporting livelihoods and food security
Labor	*Stress*: Rigid work schedules inhibit agricultural professionals' flexibility in adapting to climate extremes *Adaptation*: Policies that help develop human capital and redistribute decision making power among agricultural professionals to promote climate adaptation	*Stress*: When workers are treated as "unskilled" and exchangeable, their specialized knowledge needed to manage biodiversity is missed *Adaptation*: Empower and support agricultural professionals with expertise to enhance ecosystem services	*Stress*: Economic treadmill pushes owners to undervalue labor and rely on a contingent and vulnerable migrant labor pool *Adaptation*: Diversify crop portfolios and expand local markets to stabilize food production and income and re-circulate wealth within local communities
Land access and tenure	*Stress*: Exclusionary land markets and insecure tenure inhibit adaptive planning and long-term climate change investments *Adaptation*: Broaden who has the power to implement and benefit from diversification	*Stress*: Land markets limit alternative, land transfer, succession, and production pathways *Adaptation*: Prioritize diverse land tenure models, and incentivize transfer to new farmers and for diversifying farm practices	*Stress*: Self-exploitation by farmers who compete for land, lack mobility, and respond first to land prices and second to food production *Adaptation*: Use zoning and planning to match farmers with regional food security needs

more desirable attributes (Folke et al., 2010), which is crucial in our understanding of adaptive capacity that encompasses transformation.

Although scholars often apply these concepts to either social or biophysical dimensions, an interconnected social-ecological system is the most relevant analytical unit in agricultural systems (Folke, 2006; Gallopín, 2006). We thus define agricultural adaptive capacity as the extent to which agricultural systems can respond to the triple threat in ways that, at a minimum, preserve core social-ecological functions, and which ideally make progress toward greater equity and sustainability. Connections between adaptive capacity and sustainability are well-captured in the term "sustainagility," which emphasizes the need for agile responses to unforeseen change while also considering sustainability tradeoffs across multiple levels (Jackson et al., 2010). Conceptualizing agricultural systems as complex social-ecological systems captures the reciprocal interactions between people and the environment and can be defined at multiple levels (e.g., an individual farm or household, a community, a region etc.). Since the adaptive capacity of each level depends on levels below and above and can vary in time and space, a multidimensional perspective is essential for understanding adaptive capacity. Adaptive capacity must be conceptualized as an emergent property of social-ecological systems. It cannot be broken down into component parts or studied in isolation. For instance, at the scale of a farm, adaptive capacity emerges from the collective, intertwined relationships happening on the farm and in the surrounding landscapes and communities. Moreover, the qualities of adaptive capacity that emerge vary depending on the social-ecological processes of the system from which it emerges. More diverse and inclusive processes, for example, may be better able to create qualities of adaptive capacity that include components of social justice and sustainability.

Strengthening the adaptive capacity of agricultural systems depends on several factors (Darnhofer et al., 2010). Adaptive capacity encompasses both proactive and reactive responses to change, reducing vulnerability, and increasing resilience to a particular stressor (Engle, 2011). Proactive measures depend not only on the ability to anticipate what might happen in the future, but also on the ability to learn from past experiences and from other examples of what has and has not worked in similar circumstances (Fazey et al., 2007; Darnhofer et al., 2010; Engle, 2011). Yet since the future may not have a prolog in the past, and since changes and impacts may be varied and uncertain, *flexibility* is also key to strengthening adaptive capacity (Darnhofer et al., 2010). *Diversity* enables flexibility by increasing options for adaptation in the face of stressors, and also lowers vulnerability by helping to spread risks. Finally, adaptive capacity also depends on the *resource base* available (e.g., agrobiodiversity; Jackson et al., 2010), and the human, economic, and social capital needed to make use of it.

Simplifying vs. Diversifying Pathways for Adaptive Capacity

Adaptation strategies can be based on *simplifying processes* or *diversifying processes*. Farmers and farming systems may follow either set of processes in seeking to adapt to changing biophysical and socioeconomic conditions, which form divergent but directionally reinforcing pathways. Adaptation *pathways* embed changing conditions and response processes within broader social-ecological contexts, wherein historical actions shape future possibilities (Wyborn et al., 2015). Although defining approaches to adaptive capacity along a single axis cannot capture the full complexity involved, we believe these broadly divergent pathways provide a useful heuristic that can be adjusted to specific contexts.

Simplifying Pathways

Around the globe and across various kinds of agriculture, simplifying processes iteratively shift the basis of agricultural production from complex ecological systems toward centralized control of socially and ecologically homogenized systems (Vandermeer et al., 1998), although the extent varies by biome, availability of capital assets, agroecological knowledge, and sociopolitical organization (Jackson et al., 2012). Against the perennial challenges of variable environments and markets, simplifying "fixes" promise greater control and scalability in agriculture (Henke, 2008). Simplifying farming systems are characterized by (1) high-yielding crop and livestock varieties dependent on non-renewable, synthetic inputs manufactured off-farm (i.e., seeds, agrichemicals, equipment), and (2) increasingly concentrated markets, both for those upstream inputs to agriculture and for downstream markets for agricultural products (Block, 1990). Such processes result in greater specialization and uniformity in ecologies, landscapes, technologies, labor practices, and knowledge across large scales.

Simplifying processes offer short-term benefits to some growers, generally those who can access capital-intensive technologies, inputs, and other resources that grant them temporary production advantages over their market competitors. However, that advantage fades as other farmers either follow suit or exit agriculture, setting up the next cycle of a "technological treadmill" (Cochrane, 1993) and locking farmers into dependence on purchased proprietary inputs (Busch, 2010). Many farmers do not choose simplifying processes *per-se*, but are compelled to simplify in order to compete in a globalized economy shaped by the interlocking forces of market concentration, land consolidation, and crop and livestock homogenization (International Panel of Experts on Sustainable Food Systems, 2017). *Concentration* of market shares for agricultural inputs (e.g., machinery or agrichemicals) and products (food, fiber, and fuel) occurs through horizontal and vertical integration, in which a few firms steadily buy up their competitors and/or their suppliers (Hendrickson and Heffernan, 2002; Howard, 2016). *Consolidation* of farm and land ownership occurs both as farmers become locked into a downward economic spiral—in which they must take on debt to purchase increasingly capital-intensive inputs in the face of steadily shrinking profit margins—and through farmland financialization, in which non-farmers use new forms of financial investment to profit from farmland. This process drives a trend toward increasing farmer tenancy and absentee land ownership, which siphons wealth away from rural communities and limits the range of viable farm business models (Cochrane, 1993; Hendrickson and Heffernan, 2002; Bernstein, 2010; Howard, 2016; Fairbairn et al., 2021). *Homogenization* refers to the rapid decline of crop and livestock diversity across both farm and landscape scales due to specialization in commodity crops for global markets (Khoury et al., 2014); increasing concentration of the global seed market (Howard, 2020); and privatization of plant genetic resources (Kloppenburg, 2005; Montenegro de Wit, 2017b). These interlocking forces result in more homogenous landscapes characterized by the widespread cultivation of just a few varieties of crops or livestock and severe reduction of natural habitats.

In essence, these simplifying forces produce many losers and a few winners, exacerbating inequity in farming systems. In the US, for example, owner-operated farms have declined in number over the last century, especially for Black farmers (White, 2018), as concentration, consolidation, and homogenization have disadvantaged small and midsize farmers (De Master, 2018). While some farms grow larger and more profitable, benefiting more from government subsidies and bailouts, the majority of small and mid-sized farms, especially those operated by farmers of color, struggle to survive. Meanwhile, the remaining larger farms tend to become inflexibly integrated into fixed national and international supply chains, rendering the food system less flexible and adaptable to dramatic market changes. As food crises caused by the COVID-19 epidemic illustrate, the vulnerability of long supply chains and centralized food distribution channels renders this highly simplified system vulnerable (Heinberg, 2020; Ransom et al., 2020).

As agriculturalists respond to the triple threat, existing economic structures, production philosophies, capital-intensive technologies, public policies, and physical infrastructure associated with simplification processes create a strong predisposition to continue down a simplifying pathway (International Panel of Experts on Sustainable Food Systems, 2016). For those few already benefiting from the status quo, these structures may provide additional opportunities. Yet these lock-ins also constrain adaptation choices and reduce farm-level flexibility for everybody, adding further weight to the forces of simplification.

Diversifying Pathways

As explained in the Introduction, diversifying processes offer farmers an alternative pathway (Wezel et al., 2020). By strategically managing biodiversity and landscapes to increase the magnitude and range of ecosystem services flowing to and from agriculture (Zhang et al., 2007), diversifying processes leverage "nature's technologies" that rely on common-pool resources rather than capital-intensive technologies subject to privatization. Diversifying farming systems requires place-based knowledge of agroecosystems and context-specific innovations derived from collaboration among traditional, experiential, and multi-disciplinary scientific sources of knowledge. Diversifying processes may also promote more inclusively networked systems where alternatives to the vertically integrated supply chain model can flourish (International Panel of Experts on Sustainable Food Systems, 2016), eschewing trends toward concentration, consolidation, and homogenization of farming systems.

Research Questions and Objectives

Building on prior work showing the potential of diversified farming systems to improve social-ecological outcomes of agriculture (Kremen et al., 2012), we explore what happens when farming systems adapt to the triple threat through diversifying pathways as opposed to simplifying pathways. This exploration is motivated by several questions: What properties and qualities of adaptive capacity emerge from diversifying as compared to simplifying processes across different challenges? How might diversifying processes promote sustainability and equity across multiple levels, scales, and functions simultaneously? What challenges and opportunities might manifest through diversifying farming systems? What are key knowledge gaps for understanding how diversifying processes affect adaptive capacity?

Our objective is to address these questions through structured analyses of five cases of challenges in which farming systems struggle to adapt to the triple threat under different types of shocks and stressors (**Box 1**): living with foodborne pathogens, weathering drought, farming marginal land, dignifying labor, and enhancing land access and tenure (**Figure 2**). We selected these cases to represent challenges that range across the social-ecological spectrum and based on our expertise and research experience as participants in the Diversified Farming Systems Research Group at the University of California, Berkeley. Each case is presented primarily in the context of US agriculture, though the challenges discussed are common to farming systems worldwide. We analyze each challenge area according to a four-point framework:

1) reviewing the potential for the triple threat to exacerbate each farming challenge;
2) describing simplifying pathway trends for that challenge;
3) comparing those trends to the potential for diversifying pathways to enhance adaptive capacity to the challenge;
4) identifying barriers to diversifying pathways.

We do not expect most readers to read every case. Rather, we present a diverse palette of cases as self-contained applications of the framework from which

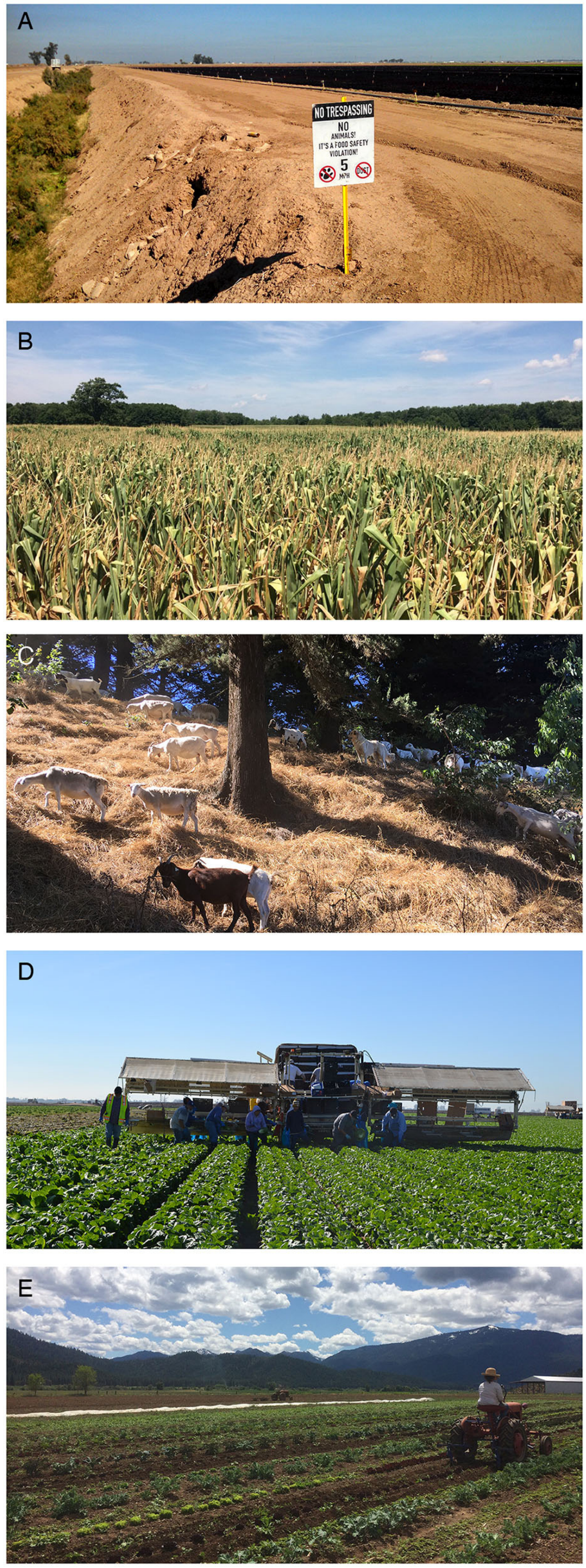

FIGURE 2 | Five cases of challenges in which farming systems must adapt to the triple threat: **(A)** Pathogens: A no-trespassing sign at the edge of a lettuce field in California, warning, "No animals! It's a food safety violation!"

(Continued)

FIGURE 2 | Many crop farms maintain bare-ground buffers around field borders, actively stripped of vegetation, in an attempt to control foodborne pathogen risks. Photo: Patrick Baur; **(B)** Maize showing symptoms of drought stress grows in a field in southern Ontario, Canada during a drought in summer 2016. Photo: Leah Renwick; **(C)** Marginal lands: Rotating livestock, like goats, on marginal land can, if managed appropriately to their context, diversify livelihoods and provide ecosystem services like fire fuel load reduction. Photo: Margiana Petersen-Rockney; **(D)** Labor: Farmworkers who harvest crops like this lettuce are disproportionately impacted by shocks and stressors like heat waves and COVID19, which exacerbate the inequities and risks they already bear. Photo: Patrick Baur; **(E)** Land access: New-entrant and socially disadvantaged farmers are often more likely to adopt diversifying farming practices, but consistently cite land access and tenure as their greatest barriers to success. Photo: Margiana Petersen-Rockney.

readers may selectively choose according to their interests before continuing to the Discussion. For quick reference and ease of comparison, we also direct readers to our two summary tables: **Table 1** summarizes our findings on increased stresses from, and potential diversifying adaptations to, the triple threat for each challenge; **Table 2** summarizes our findings on simplifying processes and opportunities for, and barriers to, diversification for each challenge.

CASE STUDIES

Living With Foodborne Pathogens

Background: The Triple Threat Increases Microbial Food Safety Risk

While risks from zoonotic diseases have long been associated with animal production systems (Sofos, 2008; Karesh et al., 2012; Rahman et al., 2020), and especially concentrated animal feeding operations (Gilchrist et al., 2007), over the past decade, foodborne human pathogens have newly emerged as a significant challenge for vegetable and fruit agriculture. In the United States, for example, repeated major outbreaks of foodborne illness—most recently several outbreaks of Shiga-toxigenic *E. coli* (STEC) associated with romaine lettuce (Marshall et al., 2020)—have been linked to in-field contamination of fresh produce crops (Bennett et al., 2018; Li et al., 2018; Turner et al., 2019). Outbreaks can cause significant human morbidity and mortality but also result in second-order shocks to farmers through lost sales, damage to market reputation, and lawsuits (Baur et al., 2017). Moreover, recurring outbreaks induce governments and private industry to introduce precautionary measures (Lytton, 2019), creating a persistent regulatory stressor on farmers to eliminate environmental sources of potential pathogenic risk (Karp et al., 2015a). The triple threat heightens microbial food safety risk (**Table 1**). Climate change may exacerbate foodborne infectious disease risks through multiple mechanisms, such as altered temperature and moisture patterns that directly influence pathogen growth and survival, as well as shifts in the distribution of disease vectors that may introduce foodborne pathogens to novel human populations (Tirado et al., 2010; Hellberg and Chu, 2015; Lake and Barker, 2018). At the same time, emerging evidence also suggests that, at least in some systems, biodiversity loss can lead to higher likelihood of disease transmission by increasing the relative abundance of species most competent to host and transmit pathogens (Keesing et al., 2010; Ostfeld and Keesing, 2012; Civitello et al., 2015; Mendoza et al., 2019), particularly at local scales (Halliday and Rohr, 2019). Compounding these potential trends, there is rising demand for year-round fresh produce to meet the requirements for nutritional food security. Yet the US food system depends on a very few major sites of production to supply this demand, leading to more intense pressure on the already consolidated, and hence vulnerable (Hendrickson, 2015), regions that specialize in vegetable, fruit, and nut crops. This leads to further centralization of distribution systems and magnification of cross-contamination and outbreak risks (DeLind and Howard, 2008; Stuart and Worosz, 2012). As described below, current simplifying trends in produce agriculture may make these farming systems more vulnerable to foodborne human pathogen stress (**Table 2**).

Simplifying Pathway Trends: Heightened Vulnerability and Magnified Risks

Many human pathogens that contaminate fruit and vegetable crops originate in the guts of cattle and poultry (Heredia and García, 2018). Concentrating animals in densely-populated locations, such as feedlots, may heighten the prevalence and transmission risk of pathogens such as STEC, *Salmonella*, and *Campylobacter* (Valcour, 2002; Frank et al., 2008; Gast et al., 2017; Poulsen et al., 2018). Simplified livestock diets may further accentuate this risk. For example, cattle eating grain-heavy diets have been shown to shed more STEC than do cattle eating diverse, forage-based diets (Callaway et al., 2003, 2009). Likewise, homogenization may increase the vulnerability of plants to pathogenic contamination originating from livestock. Monocrop fields tend to support lower levels of soil and vegetative biodiversity, which impairs ecosystem services, such as microbial competition or physical filtration, that may mitigate the transfer of human pathogens to crops (Karp et al., 2015b, 2016; Sellers et al., 2018; Jones et al., 2019).

The policy response to the risks magnified by concentrated and homogenous production environments has largely followed a simplifying process fixated on increasing technological and regulatory controls (Ansell and Baur, 2018). In the context of a siloed US policy system (Broad Leib and Pollans, 2019; Baur, 2020), such controls drive further ecological and social simplification in agriculture, leading to a self-reinforcing cycle of crisis-and-reform (Baur et al., 2017). On the ecological side, the narrowly precautionary stance embedded within food safety controls reinforces homogenization. In the absence of definitive proof to the contrary, both natural habitat (e.g., riparian vegetation) and managed beneficial vegetation (e.g., hedgerows) are presumed to be risky because they provide habitat for animals that might vector pathogens onto the field (Olimpi et al., 2019). There is thus strong incentive to "purify" farmland (DuPuis, 2015, p. 111–124) by physically separating cultivated fields from biodiverse ecosystems, leading

TABLE 2 | Simplification processes, opportunities for and barriers to diversifying processes through which farming systems could strengthen agricultural adaptive capacity.

Stressor case	**Simplification processes**	**Opportunities for diversification**	**Barriers to diversification**
Living with foodborne pathogens	• Concentration in animal production increases prevalence of pathogens • Ecological homogenization in row-crop agriculture heightens vulnerability to contamination • Centralized supply chains magnify public health risks • Standardized, top-down policies myopically focus on pathogens and limit local flexibility	• Regulate pathogens according to their disease ecology • Integrate institutional mechanisms that enable local decision-making authority and innovation • Nest microbiological food safety goals within a broader governing framework for healthy food systems	• Acute risks of foodborne disease are much simpler to identify and measure than are chronic/distributed risks • Conflation of biophysical risks with legal liability risks • Food safety regulatory regime is institutionally siloed
Weathering drought	• Farming system homogenization degrades soil, decreasing hydrologic functioning • Regional specialization in just one or two crops increases synchrony of drought impacts • Economic concentration of breeding and seed production narrows range of, and access to, drought resistant varieties	• Improve soil health to reduce sensitivity and vulnerability to drought • Increase crop diversity to stabilize food production and provide a more diverse nutrient portfolio • Support participatory breeding programs to create open source, locally-adapted, and drought resistant varieties	• Short-term costs hinder adoption of farm management practices that may only show benefits in the longer term • Capital-intensive and proprietary biotechnology dominates research funding for plant breeding • As water costs increase during drought, high-value, luxury crops are often favored over more diverse crop portfolios
Farming marginal land	• Centralized market and political forces undermine local control over farmland margins, displacing farmers and experiential knowledge • Financial investment and speculation drive homogenized production • Extractive cultivation of marginal lands, degrading soils and biodiversity	• Diversify crops to increase soil fertility while providing a sustained product return • Couple diversification strategies with landscape modification to increase resilience on erosion-prone land • Marginal lands may provide flexible production zones to help farmers handle variable climate and market conditions	• Lack of financial capital, extension support, or labor to implement diversification practices may especially challenging to marginal farmers • Socioeconomic and political pressures may outweigh the benefits of diversified farming practices, compromising farmers on marginal lands
Dignifying labor	• Land consolidation and economic concentration widen inequity between owners/operators and agricultural workers • Crop homogenization magnifies seasonal swings in labor intensiveness, requiring workers to migrate and increasing their legal and economic vulnerability • Mechanization and automation can devalue agricultural labor while driving further simplification	• Invest in agroecologically-skilled labor to improve farm outcomes through diversifying practices • Support more year-round employment with a diversified crop portfolio • Build knowledge needed for biodiversity-based management through training programs to increase dignified employment opportunities	• Greater labor intensity in diversified systems coupled with labor shortages restrict farmers' capacity for diversifying • Lack of markets that demand better farm labor conditions • Long-standing cultural belief that farm work is "unskilled" and undesirable • Farmworkers often face institutional racism barriers to becoming farm owners/operators
Enhancing land access and tenure	• Farmland consolidation drives landowners to prioritize rental profit over protecting natural resources, ecosystem services, and livelihoods • Intersectional race, class, and gender homogeneity among landowners undermines capacity to trust and empower diverse tenants to manage farmland, reducing land access and security of tenure	• Fix structural factors that limit land tenure and access • Ensure that training programs for diversifying farming are coupled with plans for land access and tenant rights • Support farmer cooperatives that share resources like land, equipment, and knowledge • Facilitate land transfer to farmers who have historically been excluded from land ownership based on social identities like race and gender	• Insecure land tenure can inhibit diversification pathways because benefits take too long to realize or do not accrue to farmers who are tenants or workers • Informal agreements with landlords, especially those conditioned by race and ethnicity, leave agriculturalists vulnerable to eviction • Property boundaries do not map to the scale of management needed for high adaptive capacity

to more biologically homogenous agriculture (Beretti and Stuart, 2008; Stuart, 2009; Baur et al., 2016; Olimpi et al., 2019).

On the socioeconomic side, this pernicious cycle also reinforces concentration and consolidation through several mechanisms. First, food safety precautions require money, time, and labor, but farmers rarely receive a corresponding price premium to offset this cost. In addition, the relative cost of compliance is higher for smaller scale as compared to larger-scale farm operations (Astill et al., 2018; Bovay et al., 2018), driving further farm consolidation due to imposed competitive disadvantage (DuPuis, 2015; Karp et al., 2015a; Olimpi et al., 2019). Second, food safety's precautionary stance disincentivizes rotation, polyculture, or integrated livestock practices, because complex agricultural management techniques multiply the burden on farmers to prove that such techniques are safe (Olimpi et al., 2019). Third, food safety standards are generally set by experts external to the target agricultural system with minimal design input by the farmers who must then implement those standards (Baur et al., 2017; Verbruggen, 2017). This top-down decision-making structure concentrates power and adopts a homogenous risk management system that rewards simplified farming systems and limits local flexibility and adaptation. Fourth, myopic focus on producing crops free from human pathogens obscures interrelationships among multiple agricultural functions and objectives (McMahon, 2013; Broad Leib and Pollans, 2019), undermining the capacity of farms to adapt to the novel food safety challenges posed by the triple threat. In these ways, the simplifying process of adaptation to pathogenic risks—based on a model of control designed for factories rather than agroecosystems (Karp et al., 2015a)—forms a pernicious feedback loop that iteratively renders farming systems more vulnerable to the challenges posed by foodborne pathogens (**Table 2**).

Diversifying Pathway Opportunities and Barriers to Increasing Adaptive Capacity: Harnessing Ecosystem Services and Distributing Authority to Mitigate Risks

This case reveals three areas of opportunity to enhance adaptive capacity toward foodborne human pathogens by diversifying farming systems that grow fresh fruits and vegetables (**Table 2**), with the goal of enabling specific adaptations to the triple threat such as those posited in **Table 1**. First, if farmers and regulators recognize the role that high biological diversity—at the farm and landscape level—might play in mitigating foodborne pathogen risks, then research effort could be directed to identify and validate novel management options for cultivating pathogen-suppressing ecosystem services (Karp et al., 2015b; Olimpi et al., 2019). For example, emerging evidence suggests that managing healthy soils for biodiverse microbial and insect communities with practices like maintaining soil cover and high above-ground diversity may effectively mitigate pathogenic strains of *E. coli* in feces (Jones et al., 2019). Second, integrating institutional mechanisms that allow for nested, multi-level standard-setting could help equalize decision-making authority between farmers and external experts and permit greater flexibility and innovation, especially for producers with less access to scientific expertise (Olimpi et al., 2019). An example would be for national regulatory agencies to delegate standard-setting and monitoring authority to smallholder cooperatives, which would be responsible for governing day-to-day food safety risks among their membership. Third, at the policy level, an opportunity exists to shift toward a perspective that accepts that pathogens are endemic to their host systems, and thus cannot simply be eliminated from the farm environment. Such an adjustment of perspective would allow diversification of food safety objectives beyond simply controlling the points of contamination where pathogen meets edible crop to also minimize the genesis of dangerous pathogens (e.g., in high-density, confined animal feeding systems) and limit their risk-factor multipliers (e.g., through centralized processing facilities) (Stuart and Worosz, 2012; Broad Leib and Pollans, 2019). To date, these opportunities remain largely unexplored (see **Appendix 1**).

The primary barrier to diversifying opportunities for ecological management of pathogens at the farm scale originates with the simplifying assumption, tacitly held by powerful market and regulatory actors, that the presence of natural ecosystems near fields automatically increases food safety risk (Olimpi et al., 2019). Efforts to ease this barrier through further agroecological research into pathogen disease ecologies are hindered by the mingling of perceptions about biophysical and legal liability risks in informing food safety decisions (Baur et al., 2017). In turn, the fragmentation of food safety governance into uncoordinated institutional silos, in the US at least (GAO, 2017), complicates any effort to overcome the preceding barriers. For example, in the US, microbial food safety for animal products is regulated separately from fruits and vegetables, while both regimes operate independently of regulatory agencies charged with overseeing other safety concerns such as pesticide risks or occupational hazards to farmworkers (Broad Leib and Pollans, 2019). Due to fragmentation and siloing, there is a general failure to acknowledge the dampening effects that microbial food safety efforts impose on attempts to manage agriculture adaptively for other goals, including those that affect public health (**Table 2**).

In summary, the simplifying pathway seeks standardized methods to control the spread of foodborne pathogens without addressing the growing vulnerabilities to pathogenic risks created through operational concentration, agroecological homogenization, and supply chain centralization. A diversifying pathway, in contrast, would seek to (a) reduce those vulnerabilities by creating strategic heterogeneity in operations, agroecological systems, and supply chains and (b) promote local resilience through ecosystem services that regulate pathogen disease ecologies and by increasing local decision-making authority to innovate place-specific mitigation strategies.

Weathering Drought

Background: Droughts Will Increase in Intensity and Frequency

In the coming decades, climate change will further increase the intensity and frequency of droughts in agricultural landscapes, especially in temperate regions (Hatfield et al., 2011; Trenberth et al., 2014), impacting farmers' immediate ability to grow crops, raise livestock, and sustain their livelihoods. These impacts will have rippling effects throughout the food system. As an example, in 2012 a widespread and intense drought across two-thirds of the continental United States reduced corn yields by 25%, which was accompanied by a 53% global price spike (Boyer et al., 2013). Dramatic declines in crop biodiversity further worsen the impacts of drought by reducing variation in how crops respond, i.e., response diversity (Elmqvist et al., 2003; Laliberté et al., 2010).

Drought acts not only as an acute shock but can also become a long-term stressor for agriculture depending on the drought's duration. Both irrigated and rainfed agroecosystems are affected when water availability becomes limited by scarcity or policy. For instance, the 2011–2016 drought in California, where most crops are irrigated, caused estimated losses of 21,000 jobs and $2.7 billion in agricultural output in 2015 alone (Howitt et al., 2015). Economic losses would have been far greater if farmers had not switched to groundwater for irrigation, though this in turn led to substantial groundwater overdraft and land subsidence (Faunt et al., 2016). Climate change-driven reductions in the snowpack that recharges groundwater exacerbate these overdraws (Pathak et al., 2018). Although new policies in California that regulate future groundwater withdrawal may reduce overdraft (Harter, 2015), this example highlights how the biophysical impacts of climate change can interact with policy change to create or exacerbate complex, multi-dimensional stressors for farmers (**Table 1**).

Simplifying Pathway Trends: Simplification Has Increased Vulnerability to Drought

The vulnerability to drought that results from simplifying processes (**Table 2**) is exemplified in the Corn Belt of the central United States, where intensive rainfed commodity corn and soybean production suffered at the center of the 2012 drought. Over the course of the last century, significant homogenization of farming systems occurred in response to federal and state policies and increasing downward economic pressures from rampant concentration of input suppliers and grain processors (Philpott, 2020), resulting in farms that now almost exclusively grow corn and soybeans rather than the small grains, hay and integrated animal pasture they once also produced (Brown and Schulte, 2011; Liebman and Schulte, 2015). Reductions in crop diversity, disintegration of crop and livestock production, and other concurrent changes in management have led to widespread soil degradation (Karlen et al., 1994; O'Brien et al., 2020), including declining soil organic matter and topsoil erosion, which in turn undermines hydrologic functioning critical for rainfed systems. At wider scales, specialization in just these two crops coupled with increasing climatic sensitivity of corn production increase regional sensitivity to drought (Lobell et al., 2014, 2020; Ortiz-Bobea et al., 2018). While the particularities of this simplification process are unique to the U.S. Corn Belt, similarly homogenized farming systems across the world increase the potential for globally synchronized climatic shocks that threaten food production (Tigchelaar et al., 2018). Genetic engineering of drought- and heat-resistant crop genotypes—one of the most commonly recommended strategies for addressing the projected increase in drought severity (Hu and Xiong, 2014; Ortiz-Bobea and Tack, 2018; Ortiz-Bobea et al., 2018; Tigchelaar et al., 2018)—will further entrench the trajectory of simplification by increasing reliance on proprietary and capital-intensive biotechnology. Despite substantial investments, genotypes engineered for drought resistance show only modest improvement, if any, over decentralized, traditional breeding approaches for drought resistance (Gilbert, 2014). Widespread use of these genotypes may in turn impact ongoing declines in crop genetic diversity if only a few engineered crop varieties displace a multitude of other varieties.

Other capital-intensive responses to water limitations exist, but may come with unexpected tradeoffs that also reinforce simplification pathways (**Table 2**). In irrigated cropping systems, field-level investments in purchased inputs like drip irrigation and water sensors can potentially reduce exposure to drought by increasing water use efficiency. In the absence of policy and institutional support for resource conservation, however, such investments can also lead to tradeoffs for soil health, such as decreased soil aggregation (Schmidt et al., 2018) and higher water consumption on a regional basis (Grafton et al., 2018). The latter phenomenon is an example of Jevon's paradox, which can occur when increased irrigation efficiency at the field level incentivizes farmers to switch to higher-value, but more water-intensive, crops, thereby causing an overall increase in water use at a regional level. Relatively expensive capital upgrades, like water sensor networks, often accompany simplifying strategies, as they are better suited to large-scale production of uniform crops. Drought itself may reduce the diversity of crops grown due to water shortage and commodity prices. For example, high value, luxury crops like wine grapes may be favored during drought over food staples with lower value, like rice (Bradsher, 2008). These shifts in production have the potential not only to affect global food supply and potentially exacerbate food insecurity, but also to push farmers along a simplification pathway.

Diversifying Pathway Opportunities and Barriers to Increasing Adaptive Capacity: Improve Soil and Increase Crop Diversity at Multiple Scales

Diversifying pathways can reduce exposure and vulnerability to drought while also providing other benefits (**Table 2**) and adaptations to the triple threat (**Table 1**). Cropping system diversification is one process that reduces impacts from drought, likely mediated through soil improvements (Lotter et al., 2003; Gaudin et al., 2015). For example, long-term evidence across multiple sites in the U.S. and Canadian Corn Belt showed that rotational diversification reduced corn yield losses by 14 to 90% in various drought years (Bowles et al., 2020). In general, improving soil's capacity to capture, store, and supply water to crops and forage increases resistance to droughts, especially in rainfed systems. Field-scale diversification practices, like cover cropping, crop rotation, application of organic amendments, and reduced soil disturbance, often increase soil organic matter (Marriott and Wander, 2006; McDaniel et al., 2014) and the abundance and diversity of soil organisms (Bender et al., 2016; Bowles et al., 2018) along with soil water holding capacity, infiltration, and porosity (Basche and DeLonge, 2017, 2019). Empirical evidence supporting our robust theoretical understanding of how these improvements increase crop performance under water limitation is only just emerging

(Gaudin et al., 2015; Gil et al., 2017; Solorio et al., 2017; Bowles et al., 2020).

Increasing crop diversity at multiple scales, from intercropping to whole farms to regional scales, can also reduce drought risk in ways other than changes to soil (Lin, 2011; Renwick et al., 2020). Increasing the diversity of crops grown at the farm-scale to include ones that differ in their water use, drought tolerance, or phenology helps reduce risks to farm-level yield and income through a "portfolio effect" (Helmers et al., 2001, Isbell et al., 2017). Reflecting this principle at broader scales, recent work shows that greater crop diversity provides a more diverse set of human nutrients and stabilizes food production at the national scale (Renard and Tilman, 2019). Breeding new crop varieties with greater drought resistance can also be a diversifying strategy, if the result of breeding programs expands rather than contracts genetic diversity. Participatory, decentralized breeding programs that develop open source, locally adapted drought-resistant varieties are a promising development (Gilbert, 2014), though significant legal, cultural, and social network transformations are needed to sustain a seed commons in a global seed market dominated by multinational corporations (Montenegro de Wit, 2017a).

Farmers face several barriers to diversification as a strategy for adapting to drought (**Table 2**). Short-term costs hinder adoption of farm management practices that may only show benefits in the longer term (DeVincentis et al., 2020). For example, while growing cover crops clearly provides several long-term benefits for agroecosystems, their establishment entails both short-term fixed costs like seeds, field operations and labor, and potential risks like disruption to planting or harvest contract schedules (Jackson et al., 2004). Policy, market, and research and development structures currently incentivize and retrench low cropping system diversity while failing to support diversification strategies (Mortensen and Smith, 2020, this special issue). Programs that provide incentives for farmers to adopt diversification practices, like California's Healthy Soils Program, can reduce barriers related to fixed costs, but may not be enough to address opportunity costs of high-value crop production. Another barrier is that diversification practices are knowledge-intensive (Carlisle et al., 2019a) and cannot be applied in a "plug and play" manner as in simplifying technological approaches like applying non-renewable fertilizers. For instance, in especially arid climates, cover crops can compete with the cash crop for water and must be carefully managed to avoid a net loss of water (Bodner et al., 2007). Even when examples of successful diversifying management practices exist in such regions, perceptions of the challenges by farmers and technical assistance providers can be a barrier to adoption.

In summary, simplifying pathways are primarily comprised of capital-intensive, large-scale technological fixes that help well-resourced farms survive acute drought crises without taking steps to reverse the crop homogenization and seed concentration trends that produce chronic vulnerability to drought. A diversifying pathway, in contrast, would seek regional resilience by promoting local-scale, and more accessible, solutions through investment in soil health, crop diversity, and participatory breeding programs.

Farming Marginal Land

Background: Shifting Boundaries of Land on the Margins

Farmers across the globe—especially those with limited access to markets, financial resources, infrastructure, and natural resources like water—have always sought innovative ways to extend production onto the margins and boundaries of arable land (Kumar et al., 2015; Calderón et al., 2018). While the definition of marginal land is highly contingent and reflective of shifting, context-specific, and interconnected biophysical and political-economic processes (CGIAR Technical Advisory Committee, 2000), in common usage these lands are often characterized by low or compromised soil quality, suboptimal precipitation or temperature, rugged or steep topography, and low or inter-annually irregular productivity (Kang et al., 2013; Peter et al., 2017). As climate change shifts the boundaries of arability, more land will be pushed toward this "marginal" category (Reed and Stringer, 2016). Simultaneously, increasing farmland consolidation limits land access and pushes smallholder farmers into regions of relatively poor fertility (Naranjo, 2012). Biodiversity loss has the potential to decrease both *in situ* ecosystem service provisioning and ecological response diversity, exacerbating the economic and ecological marginalization of these lands and those who rely on them (**Table 1**).

Simplifying Pathway Trends: Extracting Value From Marginalized Land and Those Who Farm It

Marginalizing certain lands and conflating marginal lands with the people who use them (CGIAR Technical Advisory Committee, 2000), have served to simplify agricultural landscapes and communities by promoting the replacement of complex local knowledge-based agricultural systems with homogenized commodity crop production (McNeely and Schroth, 2006; McMichael, 2012; Naranjo, 2012). The growing trend of farmland financialization in the United States offers an example of how so-called marginal lands continue to be leveraged to justify the simplification of farming systems (**Table 2**). Financial institutions often seek marginal farmland, for example land with low soil quality and little annual rainfall, for speculative investment. And this often removes that land from the hands of local farmers (Fairbairn et al., 2021). Studies from around the world suggest that marginal lands can be used for bioenergy crops (Helliwell, 2018; Koide et al., 2018), livestock production (Hall, 2018), or removed from cultivation for restoration and conservation (Merckx and Pereira, 2015). These uses can simplify or diversify farming systems, depending on how they are implemented and by whom. Many existing studies disregard the ways in which top-down approaches to transition marginal lands to more capitally productive uses can take marginal lands out of local community control or smallholder cultivation, and in the process displace resource-poor or subsistence farmers (Wells et al., 2018), exacerbating food insecurity and potentially forcing intensive cultivation into sensitive ecological areas.

Transitioning marginal lands to intensive cultivation can have devastating ecological and social consequences. Capital-intensive or technocratic approaches, which function to simplify production, can at least temporarily increase the productive potential of marginal lands, but in doing so often exacerbate underlying stressors and vulnerabilities. For example, the west side of the San Joaquin Valley in California is agriculturally constrained by salinization, selenium contamination, low groundwater availability, and impermeable clays (Ohlendorf, 1989; Garone, 1998, 2011). Despite these challenges, this landscape has been developed into one of the highest-output agricultural regions in the United States, mainly due to massive irrigation projects. Such projects remain highly controversial and have led to environmental and social harms including further-depleted aquifers, land subsidence, greater salinization, rapid die-off of flora and fauna at the Kesterson Reservoir, and increased concentration of simplified agricultural operations (Ohlendorf, 1989; Garone, 1998, 2011).

Diversifying Pathway Opportunities and Barriers to Increasing Adaptive Capacity: Tools to Mitigate Against Social and Ecological Stressors

Diversifying farming practices may allow farmers to farm productively on marginal lands by mitigating ecological stressors, including acute disturbances such as weather extremes, while also helping to restore degraded soil or mitigate inherent soil limitations (**Table 2**; Altieri, 2002). Crop diversification is a foundational agroecological technique that has helped farmers cope with the stressors they face on marginal lands and is key for adapting to the triple threat (**Table 1**). Selecting for drought-tolerant cultivars, for example, has increased climate resilience in the water-limited southwestern United States (Elias et al., 2018). Crop rotation or intercropping may also improve soil fertility while providing a low but sustained return (Ewel and Hiremath, 1998; Mader, 2002). On steeper, erosion-prone lands, coupling diversification practices with landscape modification like terracing can increase resilience (e.g., Bocco and Napoletano, 2017). For example, Nicaraguan farms on steep slopes that employed agroforestry and cover cropping for at least 3–5 years were more resilient to Hurricane Mitch's impacts (Holt-Giménez, 2002). Introducing perennial crops and livestock onto marginal lands can also improve agroecosystem functioning, ameliorate extreme weather impacts, improve soil fertility (Speakman, 2018), sustain economic returns (Peter et al., 2017; Rois-Díaz et al., 2018), and conserve cultural landscapes (Vries et al., 2015; Zanten et al., 2016).

For marginal lands, diversification practices are especially important to mitigate against social stressors, such as food insecurity. Several studies on marginal lands farmed by resource-poor farmers have found that diversification of agricultural production is co-linked to food security and diet diversity at the household level (Kumar et al., 2015; Calderón et al., 2018), although notable exceptions exist (Sibhatu et al., 2015). For example, Oyarzun et al. (2013) found a weak but significantly positive correlation between the number of species of crops grown by smallholders in the Ecuadorian highlands and dietary diversity within families. The data suggests this relationship results from the positive correlation between on-farm agrobiodiversity and consumption of on-farm products. Families with less agrobiodiversity consumed more off-farm foods and had lower overall diet diversity.

In some cases, marginal lands could provide opportunities for diversifying farming systems, especially in regions with low land values (**Table 2**). For example, Gabriel et al. (2009) found that lower quality land, often with lower land values, is likely to predispose farmers to convert to organic farming, which then encourages other nearby farmers to convert to organic. In other cases, without financial capital, knowledge, extension support, or accessible labor, diversifying marginal lands poses challenges to farmers, such as implementation costs of diversification practices (Altieri, 1999; Iles and Marsh, 2012). If farms are marginal to the dominant political, economic, and market system due to their small size or production methods, it may not be possible to maintain farmer livelihoods even with diversified agroecological techniques (Naranjo, 2012). For example, there may simply not be a market for crops grown at relatively low volumes, posing a severe economic barrier for farmers seeking to diversify on marginal lands (Sharma, 2011; Naranjo, 2012). Thus, market limitations (e.g., demands and infrastructural limitations) may further limit the adoption of diversifying farming practices on marginal lands (Sharma, 2011; Naranjo, 2012).

In summary, the simplifying pathway seeks to increase productivity on marginal lands by enrolling them in commodity crop markets that promise cash flow but may over-exploit these fragile ecosystems and undermine local food sovereignty. A diversifying pathway, in contrast, might seek to empower local communities to utilize marginal lands as flexible production zones from which a variety of farm products can be derived to complement, rather than compete with, the production portfolio of neighboring farmland.

Dignifying Labor

Background: A Double Crisis of Agricultural Labor

Agriculture in the United States faces a labor crisis. Agricultural workers are poorly paid, with few legal protections, while also facing challenging working conditions, including exposure to toxic chemicals, dangerous physical demands, extreme heat, and social hazards that threaten their health and well-being (Shreck et al., 2006; Holmes, 2013; Castillo et al., 2021). Simultaneously, due to demographic trends and migration policies at the federal level, employers face shortages and instability of labor supply (Martin et al., 2016). This situation affects critical tasks including planting, cultivating, and harvesting, which threatens food production and farm profitability and undermines farmers' ability to both adapt to climate change and use diversified farming practices. Finding ways to ensure stable, healthy, and dignified farm livelihoods that sustain sufficient food production will become more difficult as the triple threat intensifies.

Additional stressors from climate change exacerbate the inequities and risks that farmworkers already bear (**Table 1**). Extreme events such as heat waves can cause significant health consequences and socio-economic hardship for workers—while also potentially disrupting farm operations (Castillo et al., 2021). When workers continue working in extreme heat, they can suffer both short- and long-term negative health consequences such as dizziness, heatstroke, and chronic kidney disease (Fleischer et al., 2013; Stoecklin-Marois et al., 2013). A recent occupational safety rule in California requires farmworkers to take paid rest for at least 10 min every 2 h when temperatures reach 95°F (observations by co-author Castillo). Farm managers, however, often ignore this policy, resulting in income losses to a population already among the lowest paid in the United States (ibid). Drought can also adversely affect agricultural workers, for example, by causing the loss of work or increased travel time to alternative work sites if cropland is fallowed (ibid). More broadly, extreme weather events reduce worker productivity and availability, with potential negative impacts on rural economies and food production (Kjellstrom et al., 2009; Hsiang, 2010), especially when coupled with direct impacts of extreme weather on crop productivity. Farmers and farming systems must adapt to these labor challenges, either through simplifying pathways that replace labor with capital-intensive inputs (notably agricultural chemicals and new technologies whose use encourages monoculture farming methods), or through diversifying pathways that improve farm working conditions around multiple crops and farm biodiversity, by emphasizing knowledge-intensive management, offering employment stability, and valuing farmworker skills.

Simplifying Pathway Trends: Labor Shortages Are a Product of Agricultural Simplification

Consolidation of farm and farmland ownership not only hinders new-entry access to agriculture (see section Enhancing Land Access and Tenure), but also further cements the social divide between owners/operators and agricultural workers. This divide results in the devaluation of farm work, leading to a negative feedback loop in which only the most economically vulnerable workers seek employment in agriculture, which in turn leads to further disinvestment in improving farm labor wages and working conditions. A simplifying approach to this challenge perceives only a labor shortage, to be remedied by either finding new populations of workers to exploit (e.g., through migrant farmworker programs) or obviating the need for farm workers in the first place (e.g., through mechanization).

The United States, like most market economies, has opened its borders to migrant agricultural laborers (Pfeffer, 1983; Weiler et al., 2016). Historically, US immigration policies have resulted in a flow of low-wage migrant laborers from successive geographic regions, each arriving with few legal rights or protections, and who are subject to high rates of wage theft, sexual harassment and assault, and other forms of violence based on their race, gender, immigration status, and economic positionality (Walker, 2004). With a steady supply of migrant, cheap, and right-less labor, farm owners have had little incentive to internalize the production risks that these laborers have borne (Pfeffer, 1983). If, for example, drought devastates a crop one year, farmers can readily lay off migratory workers, whereas they must absorb the costs of repaying loans for an expensive harvester that is depreciating and losing value. However, in recent years, the agricultural workforce has aged (Hertz, 2019), partly due to slowed immigration from Mexico and partly because immigrant farmworkers often encourage their children to enter other careers (Martin et al., 2017). The Trump Administration targeted undocumented farmworkers for deportation, creating a fearful atmosphere that further deterred immigration (Goldbaum, 2019). An aging labor force is also more susceptible to injuries and health problems (Varney, 2017).

In the face of harsh working conditions and poor pay, migrant farm workers in the post-World War II era have tended to exit agriculture for other sectors that offer greater economic opportunity as soon as they are able. The importation of immigrant agricultural labor is therefore always in a race with the steady outflow of farm workers. US farmers have thus sought a more lasting solution: replacing human labor with machine labor. Beginning in the 1920s, mechanization for greater scales of efficiency has progressively made substantial inroads into commodity crops like wheat, cotton, tomatoes, and corn, significantly reducing demand for labor in these sectors (Schmitz and Moss, 2015). Since the 1950s, other crop sectors, such as row crops and nut orchards, have been mechanized to varying degrees. This has reinforced the simplifying tendency of farms to specialize in monoculture fields or orchards and grow larger in size (Fitzgerald, 2008). Yet many capital-intensive farms remain only partially mechanized and still rely on numerous seasonal human workers to carry out critical farming activities. For example, harvesting strawberries or romaine lettuce is not mechanized (Price, 2019). Indeed, within California, acreage of labor-intensive crops has increased due to growing market demand for these products over the past 30 years, increasing agricultural labor demand (Martin et al., 2016). Crop homogenization means that demand for labor is very seasonal: particularly at harvest, workers must constantly move between regions where specific crops are picked for a few weeks at a time. This pressure to migrate internally further increases their legal and economic vulnerability.

Facing a labor shortage, employers may be forced to spend additional time finding farmworkers, offer better pay and working conditions, or reduce production by leaving land fallow or crops unharvested (Kitroeff and Mohan, 2017; Morris, 2017). Well-capitalized farming interests may respond by re-committing to keep the labor force available, cheap, mobile, and disposable with few rights, resources, or recognition (Mitchell, 1996). If they choose a simplifying pathway, employers in the US are more likely to seek increased mechanization and automation to replace workers in labor-intensive tasks. A current example is the heavy investment in developing robotic strawberry harvesters to replace human pickers (Seabrook, 2019). Historically, many crops have proven difficult to mechanize, but with growing labor scarcity, technology developers are redoubling efforts to combine cameras, GPS, 3-dimensional mapping, and other technologies to substitute for worker dexterity and intelligence.

The overall effect of the automation trend on diversification is unclear. Adopting automated tools that replace human workers can decouple diversified farms from an unstable labor supply, potentially leading to increased farm stability. Small scale automation could improve working conditions and health outcomes, by eliminating dangerous tasks and paying fewer, but more ecologically skilled, workers better (Price, 2019). But widespread reliance on capital-intensive automation—which incurs high upfront financial costs and may take many years to pay off—may reinforce simplification processes and lock farms into inflexible production regimes that cannot nimbly adapt to novel stressors. Robots and other expensive automation technologies require further simplifying farming systems into even larger fields with uniform crops to operate efficiently (Seabrook, 2019). Ultimately, then, automation to replace labor could diminish farm economic stability. Additionally, adoption of automation can make it harder for smaller scale or more diversified farms to compete economically by driving down production costs even further.

Diversifying Pathway Opportunities and Barriers to Increasing Adaptive Capacity

The extent to which diversifying farms in the US can build adaptive capacity through their labor to maintain production and adjust to environmental changes is largely unexplored empirically. Farmers' ability to adopt labor-friendly measures to overcome current labor shortages and reduce worker exposure to climate-related extremes is also uncertain. In principle, increasing crop diversity and focusing on developing agroecological management skills rather than capital-intensive inputs such as automation could potentially increase opportunities for both workers and new-entry farmers (**Table 2**; Carlisle et al., 2019a,b). Working conditions may improve incrementally in more diversified systems due to reductions in chemical exposure, greater mental stimulation leading to increased job satisfaction, and more possibilities for year-round employment from diverse cropping and livestock systems that spread peak labor needs more evenly across seasons (Shreck et al., 2006; Bacon et al., 2012; Timmermann and Félix, 2015). This could make agriculture more attractive to younger workers, thus expanding the labor pool. It may also allow diversifying farmers to stabilize their production in conditions where simplifying farmers struggle to find enough labor. Critically, having workers who are experienced in observing and managing farm conditions can directly strengthen a farm's adaptive capacity (Hammond et al., 2013).

However, simply adopting diversification practices—in the absence of changes to the overall socio-economic environment—likely has limited potential to improve farm labor conditions at either the systemic or individual scales. Research on labor in organic agriculture in the United States has demonstrated that a greater number of jobs in the organic sector does not necessarily translate into more socially equitable jobs (Shreck et al., 2006). Organic farmworkers may still face conditions of poor pay, food insecurity, and lack of access to housing and health care, especially where farmers feel they must compete with conventional producers on price to gain entry to key markets (Guthman, 2004). Recent European research further supports this observation. By comparing working conditions for farm owners and workers across agroecological, organic, and conventional farms in Belgium, Dumont and Baret (2017) concluded that several practices—including increased crop diversity of both winter and summer crops and opportunities for laborers to participate in a variety of tasks from production to marketing— were necessary, but not sufficient, to support better working conditions for both farm employers and workers.

How pursuing diversifying pathways affects farmer responses to extreme weather events is also unclear. The impact of heat waves on farmworkers may be mitigated through measures such as providing accessible water stations, longer and more frequent rests, shaded rest areas, and adjusting harvesting schedules to avoid the highest temperatures (Stoecklin-Marois et al., 2013). Adopting these measures requires a labor-friendly farm operator. However, it is not clear whether there is a correlation between agroecological diversification and equalization of the historical power differential between farm employers and farmworkers. Diversifying farms may still not recognize workers as knowledgeable agricultural experts or share decision-making power between managers and workers (Dumont and Baret, 2017; International Panel of Experts on Sustainable Food Systems, 2018). As a result, workers can remain at the bottom of a management hierarchy. This relationship remains under-studied in the context of farm diversification. Moreover, without structural changes in markets, policies, and institutions that prevent farm owners and managers from exploiting their workers, efforts to ecologically diversify farms could actually impose further harmful burdens on farmworkers: for example, diversifying practices could require more physically intensive labor without empowering workers or improving work conditions.

While jobs in simplified farming systems may be undesirable because of the poor conditions, low pay, physical danger, and even stigma, jobs in more diverse and complex farming systems may be more socially desirable, requiring a high degree of recognized skill and knowledge (Carlisle et al., 2019a,b). If wages properly reflect the greater human capital required to diversify, then employers would need to pay these ecologically skilled workers a higher wage, which could be partly offset by reduced costs of external farm inputs and greater market value of products (Carlisle et al., 2019a). However, many farms that adopt diversification practices are smaller in scale with fewer financial resources than their larger market competitors, leaving these farm operators struggling to pay both themselves and their workers and unable to provide higher wages (Harrison and Getz, 2015; Dumont and Baret, 2017). To internalize social and ecological externalities in a diversifying system will require markets and buyers that demand better labor conditions and reward early adopters with higher prices for their products.

In summary, the simplifying pathway responds to labor problems by seeking either more exploitable workers or to replace farmworkers with machines—both processes further the devaluation of labor produced by widening inequity between farm owners/operators and agricultural workers. A diversifying pathway, in contrast, could seek to restore the value and dignity of farm work through recognition of, and investment in, the agroecological skills necessary for ecologically-based farm management.

Enhancing Land Access and Tenure

Background: Access to Land Shapes Adaptation Potential

Our final case focuses on how farmland tenure and access, primarily in the United States, shape pathways for adaptation.

The economic factors driving land use are increasingly divorced from the day-to-day operational decisions of working farms. Each year, more land is taken out of food production for other uses—e.g., energy production, residential development, or conservation reserves—while farmland that remains in agriculture is increasingly purchased by non-farmers as a capital investment (Fairbairn et al., 2021). These trends make it harder for farmers to own and access farmland, reducing their control over its dispensation. Decreased land access, tenure, and control by farmers is expected to exacerbate food insecurity and disincentivize sustainable farm practices (Trauger, 2014; Borras et al., 2015), although targeted policies may mitigate this tenure effect in some cases (Leonhardt et al., 2021). Furthermore, land access and tenure are consistently cited as the greatest barriers to the establishment of new-entrant farmers who would otherwise bring the skills, aspirations, and labor necessary for agricultural diversification (Beckett and Galt, 2014; Carlisle et al., 2019b). Heightened climate change risks coupled with biodiversity loss of ecosystem service providers will exacerbate barriers to entry for new-entrant farmers (Carlisle et al., 2019b).

Simplifying Pathway Trends: Capital-Intensive Solutions Align With Simplified Land Tenure

Capital-intensive solutions to the triple threat, like climate-smart agriculture or sustainable intensification, tend to favor simplified land tenure regimes (**Table 1**). These strategies for adaptation align with centralized decision making and consolidated land ownership. Certain high-value crop regimes dictate the "highest and best use" of farmland, further inhibiting adoption of conservation-based practices, especially those that may rely on high crop diversity (Guthman, 2004, 2019). Regional trends like natural gas development in Pennsylvania (Malin and De Master, 2016), biofuel investments in the US Midwest (White and Selfa, 2013), or corporate CAFO development in Illinois' corn belt (Ashwood et al., 2014) can put pressure on farmers—whether they lease or own the land—to maximize their production value (i.e., plant only the highest-value crops) to pay debts or to hold on to the land when it could be sold for more lucrative uses or to investment firms and hedge funds. Becoming locked-in to narrow goals of yield and profit by the ever-rising value of land itself, farmers face significant structural barriers to diversifying, limiting their potential to enhance adaptive capacity (**Table 2**).

Land access is shaped by structural factors that influence land tenure and complicate farmers' ability to diversify. Recent research has examined the role of race, ethnicity, and gender-based factors in determining inequities in farmland access and tenure (Calo and De Master, 2016; Minkoff-Zern, 2019). Exclusionary policies shape land ownership trends in the US, such that most farmland is owned by white males (Horst and Marion, 2019), a trend that grows stronger with increasing farm size and wealth (USDA Census, 2017).

As land tenure regimes continue to simplify—particularly as farmland is accumulated by distant owners interested in land as an asset (Fairbairn, 2020) and those who work on the land are disenfranchised tenant farmers—we are likely to see greater homogenization in management regimes. In this context, the characteristics of actors (class, ethnic background, motivation) who have capacity to make decisions becomes less diverse. More importantly, the ability of tenant farmers to influence changes to the landscape diminishes, as they follow prescribed production pathways that allow them to meet the conditions of their lease. Under such tenure regimes, the capacity for field and landscape level diversification shrinks.

Access to land is mediated by social mechanisms beyond property regimes that determine the ways in which agriculturalists can actually derive benefit from land and to what extent (Ribot and Peluso, 2003). This suggests that having title to land does not guarantee that the holder will be able to gain full benefit from the lands' total capacity. Instead, potential land use options are constrained by structural factors, such as food safety regulations (Olimpi et al., 2019), water rights (Calo and De Master, 2016), or neighboring land uses (e.g., pesticide drift). Building agroecosystems through diversification often takes years, at which point those benefits may be realized by the landlord if the farmer was renting, or the next owner. In other words, land tenure does not necessarily determine who might benefit from even farm-scale diversification. Despite the fact that simply identifying who owns land in itself is not enough to understand whether a farmer will seek to diversify, the majority of related research to date focuses on the relationship between land ownership and farm practices.

Diversifying Pathway Trends: Support New Entrant Farmers and Alternative Land Access Structures

When considering diversifying farming systems, it is important to consider farmers themselves as an axis of diversity who bring, as social network theorists posit, innovation and new ideas introduced at the margins of networks (Granovetter, 2005). The ability of farmers to build adaptive capacity through diversification, therefore, relies not only on access to land, but also on the ability to build and use their knowledge of their land (**Table 2**).

Research indicates that new-entrant and socially disadvantaged farmers (e.g., women, immigrants, racial/ethnic minorities, and young farmers) may be more willing and likely to adopt diversifying farming practices (Deaton et al., 2018). Many immigrant farmers in the US have agroecological expertise and experience using diversifying farming practices that could improve adaptive practices to a wide range of conditions (Shava et al., 2010; Minkoff-Zern, 2019). Carney (2020) points to the cultural knowledge and social memories of Afrodescendant smallholder farming systems that have "long prioritized agrobiodiversity and agroecological practices." Latinx farmers not only provide the vast majority of US agricultural labor today, but also bring expertise and diversification values (Minkoff-Zern, 2018). It is therefore crucial—for both equity and sustainability—to expand land access for new-entrant and socially disadvantaged farmers.

While most funding to support new-entrant and socially disadvantaged farmers in the United States has focused on farmer education and training programs, these have little impact on the adoption of diversified farming practices if farmers do not have the agency to implement them due to limited land access or insecure tenure (Calo, 2018). Training programs that, alternatively, decentralize expertise and enhance farmer networks—such as farmer-to-farmer learning networks—can help overcome land access issues by sharing knowledge and building solidarity and collaborative relationships (Holt-Giménez, 2006; Bacon et al., 2012; Carlisle, 2016). Additionally, governments can incentivize land transfer programs. Agricultural conservation easements, for example, are an important tool to lower the price of farmland, making it more affordable to new-entrant farmers, and farmers enrolled in the federal Conservation Reserve Program can receive an additional 2 years of government payments if they rent or sell that land to new-entrant farmers (Carlisle et al., 2019b). Programs like these should be expanded, protections from corporate capture put in place, and new entry and socially disadvantaged farmers prioritized (Calo and Petersen-Rockney, 2018).

As fewer farmers own the land that they farm, there are some promising signs that non-operator landlords—from private conservation-minded individuals, to conservation non-profits, to government agencies—increasingly recognize the ecosystem services that diversifying farming systems can provide, like managed grazing to improve endangered species habitat or reduce fuel load in fire-prone areas (Plieninger et al., 2012). Landowners can make lease agreements that specify the use of conservation practices, and can choose to prioritize farmers who manage their farm enterprises in ways that benefit the land, and farmers can negotiate more favorable lease terms or prices in exchange for providing these services to landowners, thus improving the land and/or generating ecosystem service payments through healthy soils or carbon farming programs (Ribaudo et al., 2010; Iles and Marsh, 2012; Ma et al., 2012; Petrzelka et al., 2013). Education and outreach programs for landowners regarding conservation practices can also be effective at increasing incorporation of diversified farming practices into agreements. One innovative example from the US Midwest tailors conservation programs for women non-operator landowners, who own half of the farmland but participate less in conservation decisions than non-operator landlord men (Wells and Eells, 2011). Following women-only field training in conservation practices, women non-operator landowners were substantially more likely to participate in decisions with tenants to implement conservation practices (Sreenviasan, 2020). Novel institutional opportunities to shift leasing norms are also emerging, for example through agricultural land trusts and agricultural easements that maintain land in agriculture into perpetuity.

Alternative ownership structures, like grower cooperatives—in which producers own a collective stake in the farm business—may facilitate diversification pathways and lead to greater adaptive capacity. For example, in the southern US, the Federation of Southern Cooperatives began reenergizing the cooperative farming model that had been popular among Black farmers in the late nineteenth century and early twentieth century to facilitate the sharing of experiences and expenses and slow the tide of farmland loss among Black farmers (White, 2018). Farming cooperatives, equipment cooperatives, and farm incubator programs all provide institutional mechanisms for farmers to share resources, like specialized equipment such as seed drills, which can be prohibitively expensive for individual farmers to purchase, posing a barrier to implementing diversifying practices like reduced tillage (Carlisle et al., 2019b).

Because biodiversity-enhancing strategies are best managed by coalitions of land managers working at a landscape scale (Brodt et al., 2008), diversifying farming systems offers opportunities to coordinate land managers in regionally scaled networks. In the 1990s, with government support, the grassroots Landcare movement in Australia motivated thousands of farmers to form local groups to cooperate in conservation projects, like controlling invasive species or managing soil erosion (Curtis and De Lacy, 1996; Sobels et al., 2001). Similarly, US farmers and ranchers often coordinate management efforts informally through "norms of neighborliness" (Yung and Belsky, 2007) or formally through coordinated ranch management planning for habitat restoration goals (Petersen-Rockney, in progress).

In summary, the simplifying pathway limits long-term adaptive capacity by failing to address the significant barriers that insecure land tenure poses to adopting sustainable farming practices, and that limited land access poses to diverse new-entrant farmers. A diversifying pathway, in contrast, would seek to secure tenure and expand land access through both innovative resource-sharing mechanisms and legal and policy reforms, foundational for the emergence of just and sustainable adaptive capacity.

DISCUSSION: DIVERSIFYING FARMING SYSTEMS FOR ADAPTIVE CAPACITY

Using the same integrated four-point framework (see section Research Questions and Objectives and **Tables 1**, **2**), we have contrasted the properties and qualities of adaptive capacity that emerge from simplifying vs. diversifying adaptation pathways for five widely varying agricultural challenges. Synthesizing insights across these five cases illuminates the potential for diversifying strategies to preserve and, ideally, enhance core social-ecological functioning in the face of climate change, biodiversity loss, and global food insecurity (**Figure 3** and **Table 1**). Diversifying processes, we argue, weave a form of broad and nimble adaptive capacity that is fundamentally distinct from the narrow and brittle adaptive capacity produced through simplification. Diversifying processes also demonstrate potential to enhance equity and sustainability. Yet our analysis reveals some cross-cutting barriers to diversification, such as exclusionary land tenure regimes and lack of available markets for diverse farm products. To give a specific example, marginal land may be drought-prone, and those farming it more likely to be disenfranchised with insecure land tenure (section Farming Marginal Land). While there are barriers, there are also synergies and positive feedback dynamics that can arise among the ecological, managerial, economic, scientific, and institutional opportunities

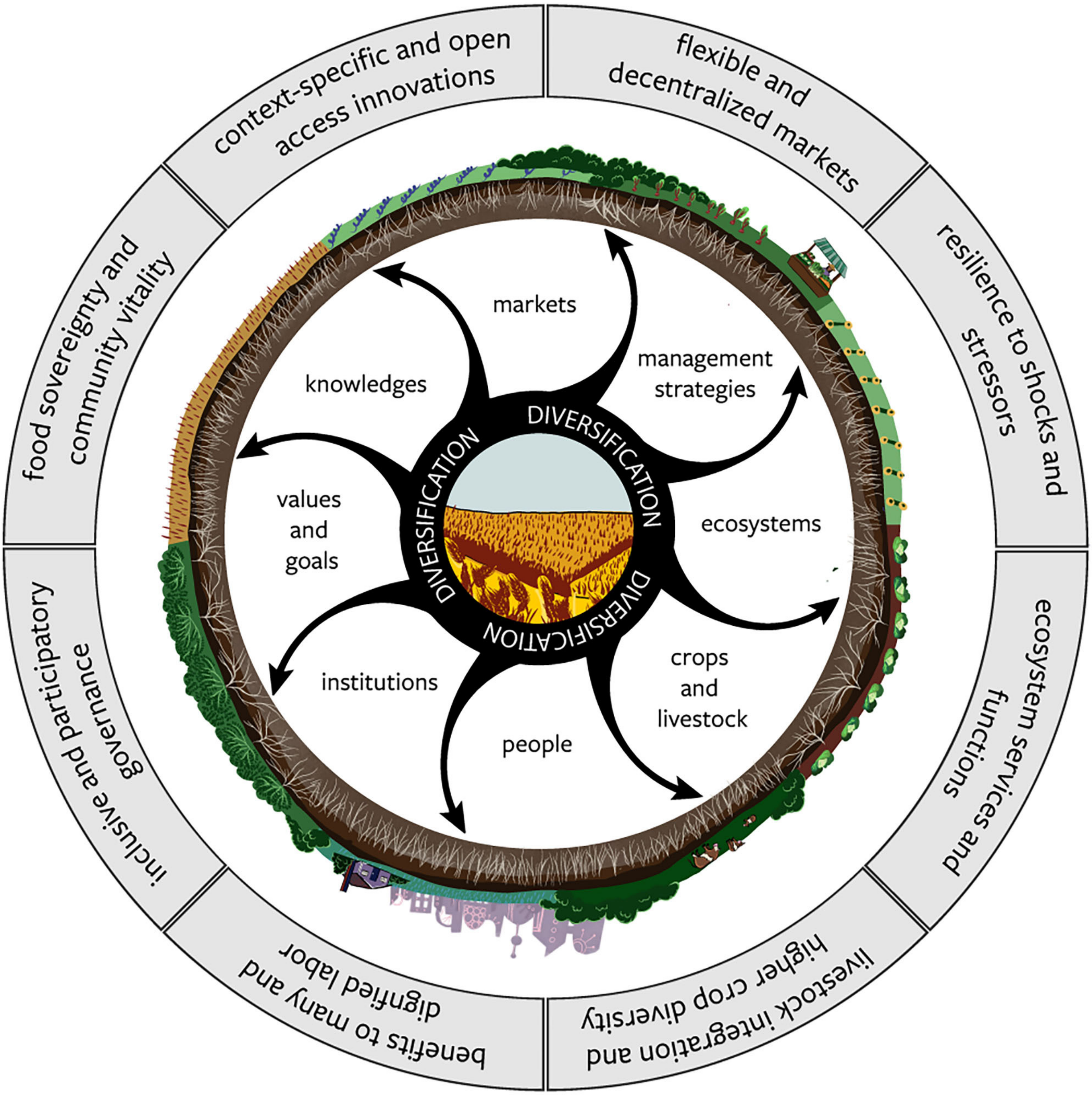

FIGURE 3 | As farming systems move from a simplified state (center of wheel) to a more diversified state (outer ring of wheel) along multiple dimensions (spokes of the wheel) adaptive capacity can emerge. But the wheel will only "turn well" when multiple dimensions grow together and interact, i.e., progress along only one or two dimensions will not be sufficient to create these properties.

to diversify farming systems (**Figure 3** and **Table 2**). Finally, we acknowledge the limitations of our approach and analysis, and suggest new research directions to fill key gaps in our understanding of the potential to diversify farming systems for adaptive capacity.

Simplifying Farming Systems Leads to Narrow and Brittle Adaptive Capacity

Stressors and shocks of various kinds will force farm management responses that, under current structural conditions, are likely to further simplify farming systems via greater reliance on proprietary capital-intensive inputs and concentrated markets. As noted earlier, while the majority of farms will struggle to survive, some well-capitalized farms will likely prosper under these conditions. Yet even those limited benefits may be vulnerable as the triple threat intensifies. At the system level, across our five cases we observe that the intensity of stressors and the magnitude of potential shocks increase as farms become locked-in to simplifying pathways of production that are expected to hold constant across space and time.

As the five cases highlighted, although simplifying pathways may appear to increase adaptive capacity to field-level shocks or stressors when those problems are considered in isolation, they often prove to be *brittle* responses to the complex and intersecting challenges of the Anthropocene. Simplification strategies can sometimes "resolve" field-level challenges and stabilize yields of specific crops under a narrow set of predictable conditions (Swift et al., 2004). However, these fixes may be prone to sudden catastrophic failure, inasmuch as when they break, they break big. For example, when the food safety system does experience a local lapse in control, and some pathogenic contamination is able to enter the supply chain, the combination of centralized processing and long-distance distribution can send the entire system into shock, i.e., a pathogen outbreak (section Living With Foodborne Pathogens). As another example, farmland financialization may improve irrigation infrastructure that can boost yields on marginal land in the short term, or allow for planting high-value perennial crops, but those gains will fade as climate change brings multi-year droughts unprecedented in modern times, leading to depletion of groundwater resources and increased soil salinity (sections Weathering Drought and Farming Marginal Land). Regional stressors like droughts and labor shortages can also cause synchronized production shocks that destabilize entire markets, especially when farms specialize in only a handful of commodities and supply chains (**Table 2**). With sufficient access to capital and institutional support structures, simplified farms can respond to singular stressors or shocks in isolation. A few well-resourced farms may, for example, be able to track market trends closely and quickly replace low-demand crops with high-demand/value alternatives, as some produce growers have done in response to shifting consumer demands in the COVID-19 pandemic (Yaffe-Bellany and Corkery, 2020). Yet the COVID-19 pandemic has highlighted how brittle our current food system is. Supply chains have broken under the strain of rapid consumer and labor shifts, forcing dairy farmers to dump their milk down the drain, meat producers to cull animals when the only facilities that can process their products shut down, produce farmers to let their crops rot in the field, and farmworkers to face the impossible choice between unemployment (and financial ruin) and risking life-threatening illness on the job (Ransom et al., 2020). In summary, apparent gains through simplification represent a form of temporary and "highly optimized tolerance" that makes the system inflexible to novel or compounded stressors and shocks (Janssen et al., 2007).

The kind of adaptive capacity pursued through simplification is also *narrow*, in two senses. First, by focusing on maximizing production of a limited number of commodities by fewer and fewer firms, simplification narrows the range of ecological and social functions sought to be preserved through adaptation, in part because they are not prioritized in the first place (Kloppenburg, 2005; Khoury et al., 2014; Montenegro de Wit, 2017b; Howard, 2020). Planting a single high-yielding and drought-tolerant patented crop variety, for example, may increase yields during a moderate drought, but fail during more severe droughts or other extreme weather (e.g., floods or high winds) in the absence of other adaptive practices (section Weathering Drought). Second, the benefits of simplification accrue to a narrow slice of people—a privileged minority—while production risks associated with simplification are often externalized into the public sphere through programs like commodity price supports and subsidized crop insurance (Graddy-Lovelace and Diamond, 2017), and onto the bodies of farmworkers and low-income and majority-black-and-brown communities who are disproportionately exposed to dangerous work and pollutants (Harrison, 2006; Fleischman and Franklin, 2017). For instance, the capacity for market flexibility following COVID-19 mentioned above does not account for farmworker health risks as an externality (Chang and Holmes, 2020). In general, the narrowing of adaptive responses leads to further inequity in distributions of benefits and burdens as, in the earlier example of patented drought-tolerant varieties, owners of the germplasm, land, and other inputs profit while farmers and farmworkers bear the burdens of risks like harvest failures (section Dignifying Labor).

Our cases show that a rigid social hierarchy exacerbates exploitation and produces further externalities (section Dignifying Labor), centralized regulatory systems disproportionately burden small scale agriculturalists (section Living With Foodborne Pathogens), and an inflexible property regime leads to gradual consolidation of land over time, cutting off access to socioeconomically disadvantaged agriculturalists (sections Farming Marginal Land and Enhancing Land Access and Tenure). In sum, fewer people benefit in fewer ways from the kind of narrow adaptive capacity that emerges through further simplification.

That simplifying processes produce narrow and brittle adaptive capacity is borne out by mounting evidence that agricultural simplification itself is a major cause of the Anthropocene triple threat. As long as the underlying structural paradigms of our dominant global food system persist, without specific attention to the processes by which they meet their goals of "incentivizing and enabling intensification" (Campbell et al., 2014), these approaches portend further entrenchment of simplified agriculture. Additionally, simplified farming systems must be homogenous over large spatial and temporal scales to allow for standardization, which leads to greater agroecosystem sensitivity to environmental stressors (Lobell et al., 2014; Ortiz-Bobea et al., 2018; Renard and Tilman, 2019). To the extent that "diversity and redundancy are an insurance against uncertainty and surprise," simplification not only exacerbates climate change and biodiversity loss, but also impairs effective responses to these stressors (Darnhofer et al., 2010). Moving toward further simplification in reaction to these challenges will likely undermine the processes by which a more sustainable and equitable adaptive capacity to the triple threat could emerge.

Diversifying Farming Systems Leads to Broad and Nimble Adaptive Capacity

We also asked specifically how diversifying processes might promote sustainability and equity across multiple levels, scales, and functions simultaneously. Our cases show that pursuing diversification pathways through multiple social and ecological domains can lead to a virtuous cycle in which multiple beneficial qualities can emerge. The cycle, however, will only 'turn well' when multiple dimensions grow together and interact, i.e., progress along only one or two dimensions will not be sufficient to create these properties (**Figure 3**). For example, while adopting diversifying agroecological field management practices can increase resilience to climate shocks and stressors at the farm scale, that adaptation alone cannot defuse global threats or build systemic adaptive capacity (Holt-Giménez et al., 2021). Diversifying farming systems can support the emergence of a broader, more robust, sustainable, and equitable adaptive capacity. Adaptive capacity that emerges from diversifying farming systems can enable nimble responses to a broad spectrum of possible, even not-yet-imagined, stressors and shocks. Instead of locking farming systems into pernicious cycles that reproduce social and ecological externalities, systemic commitment to social equity and ecological sustainability becomes the self-reinforcing conditions from which adaptive capacity emerges. Greater social equity leads to better balance between the priorities of privileged and vulnerable populations (reducing social externalities), while ecologically sustainable management seeks balance between present and future needs (reducing environmental externalities). Both equity and sustainability bolster the capacity to recognize localized stressors early, enabling more rapid and precise reallocation of resources and expertise to mitigate those stressors before they amplify into systemic shocks. Equity and sustainability are thus both prerequisite conditions for high adaptive capacity in farming systems, and our cases suggest that both qualities can emerge from diversifying processes.

The ability to envision and prioritize multiple goals appears across our cases as an essential condition for diversification. It is critical to value farming system functions beyond maximizing yield and profit to include conserving biodiversity, supporting community vitality, strengthening food security and sovereignty, enhancing justice and equity, and increasing non-commodity ecosystem services to and from agriculture (Maier and Shobayashi, 2001; Van Huylenbroeck et al., 2007; Renting et al., 2009). Agriculture can, and should, support a wide range of benefits. Multifunctional agricultural systems emphasize social and environmental values and components of farming activities beyond private or individual values. Intentionally promoting multiple and accessible functions facilitates the emergence of system properties that underlie adaptive capacity. For instance, when diversifying practices adopted at landscape scales promote soil-based ecosystem services like water capture and storage, agroecosystems can better cope with droughts and more farmers can access this benefit. Furthermore, adaptive capacity may itself be an explicit goal, either as a generic proactive preparation for uncertain future challenges or as a specific catalyzing agent for other goals, such as conserving biodiversity amidst increasingly forceful food safety pressures.

Whether in response to crisis, slow-simmering changes in consciousness, shifting social-ecological circumstances, or other triggers, when decision-makers and farmers recognize and act on multiple goals they often take diversification steps across ecological, social, and economic dimensions (Atwell et al., 2009; Carlisle, 2014; Roesch-McNally et al., 2018). Although structural forces and macro-scale factors still constrain adaptation (Stuart and Gillon, 2013; Roesch-McNally et al., 2018), the resulting diversification creates options and flexibility in responding to stressors and shocks, spreads risks, and increases response diversity—all of which facilitate the emergence of adaptive capacity.

Balancing the pursuit of multiple goals requires an inclusive process for active dialogue and iterative negotiation among diverse and potentially diverging perspectives. For example, dignifying farm labor will not be achieved through ecological diversification alone, but also requires reconfiguring hierarchical relationships among workers, managers, and owners. Conceptualizing social-ecological function to encompass multiple provisioning and service goals simultaneously not only widens the scope of potential solutions and the set of metrics by which to evaluate those solutions (International Panel of Experts on Sustainable Food Systems, 2016), but also allows for redistributing decision-making power, agency, and voices from owners and managers to frontline workers.

The levels at which stressors and shocks first manifest or can best be managed—such as regional labor markets or climates—do not align with a hierarchical understanding of farmers as individual actors, or even actors subject to hierarchical relations of power such as landowner vs. renter, farm manager vs. worker, or individuals vs. states or institutions. For example, drought is often felt at a regional scale while groundwater recharge is best managed at a watershed or landscape scale. Farmworkers may have more direct contact with changing field conditions, but may not have the authority to make timely management decisions. Additionally, rigid hierarchies also manifest in social networks that can be exclusionary, protecting static notions of culture and antiquated management norms (Naranjo, 2012; Bidwell et al., 2013; Davidson, 2016). Rigid hierarchies therefore limit system adaptation to specific levels with little recognition that farming systems are deeply interconnected through biophysical and socioeconomic ties. Instead, farming systems require a high degree of coordination and cooperation across levels in order for adaptive capacity to emerge.

Systemic diversification may provide a pathway to iteratively dismantle the lock-ins that perpetuate simplification and associated hierarchy. Greater biological diversity and access to more diverse and redundant market channels and community connections, for example, equips agricultural actors with management options to nimbly navigate through stressors like droughts or supply chain disruptions from foodborne pathogens or pandemics. Moreover, a dynamic operation may better attract skilled workers, who then add their human capital to further bolster its adaptive capacity. In contrast to the rigid hierarchies characteristic of simplifying pathways, our cases suggest that adaptive capacity can be facilitated by empowering people and enhancing ecosystem functionality to proactively distribute resources and knowledge where needed and nimbly respond to changing circumstances.

Challenges and Opportunities That Arise With Diversification

While our cases highlight a broad range of diversifying opportunities, they also reveal tradeoffs, limitations, complexities, and uncertainties that can manifest through diversifying farming systems, as well as the unexpected opportunities that can be harnessed from change. One important limitation is that enhancing ecological diversity alone is insufficient to promote sustainable and equitable adaptive capacity. Ecological diversification cannot be practiced widely without addressing the environmental, cultural, political, economic, institutional, and technological factors that constrain farmers' capacity to diversify at the farm level. Examples of these factors from our cases include the ubiquity of access to labor, land, and markets in mediating farmers' capacity to increase crop diversity or substitute on-farm ecosystem services for external capital-intensive inputs (e.g., sections Weathering Drought and Farming Marginal Land). Our cases demonstrate how building adaptive capacity requires diversifying processes across all of these dimensions—from diverse institutional arrangements to better manage foodborne pathogens (section Living With Foodborne Pathogens), to increasing land access and tenure for new entry farmers (section Enhancing Land Access and Tenure), to diversifying open-source technologies and broadening who is considered a holder of knowledge to enhance marginal lands and protect cultural landscapes (section Farming Marginal Land).

Diversification as a general strategy across these many dimensions seems to face a "chicken-and-egg" dilemma: no single diversifying step definitively comes first. Rather many steps likely need to be made iteratively and in concert with one another (**Figure 3**); otherwise, diversifying along one dimension may pose a tradeoff with another dimension. For example, increasing crop diversity in the absence of reforms for farmworker protection and compensation may actually further entrench hierarchical and exploitative labor relations (section Dignifying Labor). On the other hand, this lack of clearly ordered steps may open many entry points for farmers and other decision-makers to begin diversifying from any starting point and along any dimension, no matter how simplified their farming system is to begin with (**Figure 3**). We see a pressing need to consider opportunities for structurally diversifying farming systems, such as the nested regulatory system suggested for enhancing adaptive capacity to foodborne pathogens or supply chain flexibility in demand and distribution to better absorb the harmful economic impacts of drought events (**Table 2** and **Appendix 1**). Opportunities for local diversification exists, but will require systemic support from higher levels to reach their full potential to enhance sustainable and equitable adaptive capacity.

Another challenge arises from the complexity that diversification intentionally introduces into agri-food systems. As the complexity of a system increases, so does uncertainty about the specific outcome of any given event (e.g., a decision or disturbance), especially at fine-grained resolution. Furthermore, stochasticity, non-linearities, and surprises increase- despite our increased ability to use complex models- the ability to apply mechanistic thinking decreases (Preiser et al., 2018). Complex adaptive systems also tend toward dynamic equilibria, suggesting that rather than seeking persistent adherence to a desired state, adaptive capacity more humbly seeks ephemeral balance points that are ever-changing or fluctuating (Gunderson and Holling, 2001; Allen et al., 2014). Adaptively managing complex systems also means tolerating uncertainty and accepting a loss of short-term, precise control in favor of long-term, "fuzzy" stability in realizing system goals. Whether a community is responding to unfair labor conditions, a farmer to a severe weather event, or lawmakers to new disease ecologies, every adaptive "solution" is contingent, provisional, and temporary, subject to change as circumstances shift, dialogue progresses, and new insights are learned. When the outcomes of an action are unknown or uncertain, cycles of targeted experimentation, observation, learning, reflecting, and adjustment are key (Allen et al., 2011), as are participatory conversations that more fully represent the range of ways to frame problems and solutions and weigh the potential distribution of burdens and benefits (Jasanoff, 2003).

In our cases, farmers face ongoing challenges to balance on-farm biodiversity with the potential transmission of pathogens onto growing crops (section Living With Foodborne Pathogens), work to improve land and ecosystem resilience on marginal lands while maintaining production that supports livelihoods (section Farming Marginal Land), and manage labor shortages while addressing more labor-intensive practices associated with ecologically based farming practices (section Dignifying Labor). Adaptive capacity therefore does not aim to "fix problems" that farmers face, but to enable self-organization such that farming systems minimize challenges to begin with and can functionally persist, and even improve, through change (Vandermeer and Perfecto, 2017). For example, diversifying pathways for weathering drought —e.g., soil management practices that improve water capture and storage— do not immediately deliver water when a drought occurs, but if these field management practices are in place and supported by secure land tenure, diverse markets, and workers valued for their knowledge, the farm will likely be less negatively impacted by drought and may even be able to harness new cropping or marketing opportunities.

Knowledge Gaps and Future Research Directions

Scholars, practitioners, and policy makers can use the conceptual framework presented in this article (**Figure 3**) to describe, evaluate, and guide efforts to diversify farming systems in order to enhance their adaptive capacity. We identified research needs for each case presented (**Appendix 1**), which articles in this special issue begin to address. We invite future research that utilizes our framework, and tests its utility, in assessing how diversifying farming systems can nurture adaptive capacity to climate change, biodiversity loss, and food insecurity, as well as novel shocks and stressors. The COVID-19 pandemic, for example, is already transforming long-standing challenges such as labor shortages and exacerbating systemic threats like global food insecurity.

Many questions remain about potential institutional, market, and policy reforms that could increase adaptive capacity by diversifying farming systems while also promoting equity and sustainability. Case studies and comparisons of policy instruments, legal tools, and equitable decision-making structures can, for example, help us understand the conditions under which diversification can be pursued or continued on marginal lands without compromising resource-poor farmers or sensitive ecosystem needs (section Farming Marginal Land).

Likewise, exploring how to de-silo the US food safety regulatory regime would enable situating food safety goals within a broader governing framework for just and healthy food systems (section Living With Foodborne Pathogens). Studies that explore how to manage landscapes collectively are crucial to promoting the emergence of adaptive capacity at the landscape and community levels (section Enhancing Land Access and Tenure).

Future work that is situated in specific social and ecological contexts, including research outside the US, can highlight opportunities for public policy and social movements to incentivize and increase adaptive capacity by diversifying farming systems. For example, empirical studies that document how soil improvements made through diversified farming practices affect drought responses will need to consider a myriad of environmental, social, and economic factors to understand context-specific tradeoffs, leverage synergies, and mitigate risks (section Weathering Drought). To assess whether agricultural diversification reduces or enhances adaptive capacity in the case of farm labor, research must grapple with considerable agronomic and socioeconomic heterogeneity (section Dignifying Labor). The challenge of land access and tenure highlights the need for regionally contextualized understandings of who has the power to implement diversifying land management pathways (see Calo, 2020, in this special issue). Geographies outside the US, for example, illustrate a wide variety of socio-legal land tenure contexts, such as communal lands and sovereign sub-territories, that demand analysis beyond landlord-tenant dynamics (section Enhancing Land Access and Tenure).

Diversifying farming systems and building adaptive capacity requires promoting diversity in scholars and scholarship. This "thickening" of the legitimacy of agroecological knowledge must include work conducted by people and in ways not generally recognized as conventional scientists and science (Montenegro de Wit and Iles, 2016). Diverse knowledge bearers and ways of knowing (Haraway, 1988) invite more collaborative, transdisciplinary, and participatory research (Méndez et al., 2013) that can "take better account of the world" in particular times and places (Fortmann, 2008)—a prerequisite for context-specific diversification pathways that can build real-world equitable and sustainable adaptive capacity. Applied, participatory agroecological research is necessary to, for example, identify key drought-related issues, find innovative and appropriate practices, and select species that are context-specific in their ability to tolerate and alleviate drought stress (section Weathering Drought). Employing our conceptual framework, future research on farm labor should, for example, include perspectives from people who work on farms in the process of appropriately scaling and co-designing technologies that support diversification while also protecting farmworkers by alleviating physically dangerous work (section Dignifying Labor).

A diversifying farming systems research agenda must also analyze the institutionalized and entrenched power hierarchies that limit the potential for diversification strategies to benefit food and farm workers as well as environments. This includes interrogating legacies within our own research institutions that have facilitated simplification, from the expropriation of Indigenous land (Lee and Atone, 2020) to ongoing privatization of public resources and knowledge (Hightower, 1972; Kloppenburg, 2005). Research and development that encourages simplifying pathways dominate our public and private institutions, at least in the United States, leaving diversifying pathways largely sidelined and ignored (Miles et al., 2017). Limited resources to conduct sustainable agriculture research remains a significant obstacle to addressing the gaps outlined in this article (DeLonge et al., 2020). Research is needed to understand precisely how to shift resources and energy away from simplifying and toward diversifying farming systems simultaneously in both our research institutions and the broader agri-food system in ways that support actors in the process of transitioning pathways (see, for example, sections Living With Foodborne Pathogens and Weathering Drought).

Our conceptual framework highlights the conditions under which farming systems can cultivate forms of adaptive capacity with qualities congruent with principles of equity and sustainability. Future studies should also consider how diversifying farming systems and emergent adaptive capacity can resist market capture and dilution, learning from past efforts in organics (Guthman, 2004) and fair trade (Jaffee, 2014). Simultaneously, there may be transition opportunities for those who employ diversifying farming systems practices (such as agroforestry or cover cropping) to benefit from market-based funding mechanisms, like payment for ecosystem services programs or market premiums. Identifying inflection points when diversifying farming systems can overcome barriers, even turning them into opportunities, in response to global change will continue to be important for moving on a pathway toward a more just, sustainable, and agroecological food system that benefits a broad range of actors.

Limitations of Framework

We have outlined a descriptive framework to appraise agri-food system challenges, impacts of the Anthropocene triple threat, and the potential for sustainable and equitable adaptive capacity to emerge from diversifying farming systems. This framework can provide the foundation for developing future analytical and intervention-oriented tools to facilitate the emergence of adaptive capacity through diversification. The complexity and heterogeneity that arises from diversification, which is so crucial to adaptive capacity, complicates efforts to model or predict how these systems respond to global threats (Morton, 2007). This framework is not intended to specify precisely what should be analyzed or define the units of analysis. These points will need to be developed for each context-specific study before comparisons, evaluations, experiments, interventions, or other research processes are pursued in future work. While this is certainly a limitation of this paper, it is so by design—diversification and adaptation to context are also foundational processes in the type of reflexive research needed to diversify farming systems.

Most participants in this group conduct research in the United States and other high-income country contexts, and only a few of us have direct experience as farmers or practitioners. Most of our specific cases are situated in particular US contexts, with some resonant global examples used. We invite future work exploring the emergence of adaptive capacity from diversifying farming systems and the applicability of the framework to other contexts. Recognizing that these biases limit the scope of our assessments, we recommend that future work in this area seek to represent a broader and more inclusive range of interdisciplinary knowledge, multi-stakeholder expertise, and social network innovation and exchange within situated local contexts (Iles and Marsh, 2012).

CONCLUSION

Rather than fixed states, diversification and simplification are ongoing and iterative adaptation pathways in farming systems. Building on this conceptual extension, we presented a novel framework that integrates social and ecological systems to analyze the properties of adaptive capacity that emerge through simplifying as opposed to diversifying pathways. Applying our framework to five distinct cases of stressors in the agri-food system, we have found that:

1. Simplifying processes, though high performing under narrow metrics of productivity, lead toward a narrow and brittle adaptive capacity marked by exacerbated long-term vulnerability to the triple threat of climate change, biodiversity loss, and food insecurity.
2. Diversifying processes, though their benefits may not accrue directly or immediately, lead toward broad and nimble adaptive capacity marked by long-term and equitably distributed resilience to the shocks and stresses emanating from the triple threat.
3. Both simplifying and diversifying processes are self-reinforcing, forming "vicious" or "virtuous" cycles, respectively.
4. Diversifying farming systems means pursuing multiple goals in concert across multiple levels and domains simultaneously (ecological, structural, social, cultural, economic, etc.) with attention to potential tradeoffs.
5. Diversifying farming systems offers a pathway to embrace complexity and uncertainty, which is particularly useful in times of global shocks (e.g., pandemic) and change (e.g., long-term droughts).

Diversifying farming systems provides opportunities to increase adaptive capacity in various ways—including via different processes, magnitudes, and rates of adoption—that are ecologically and culturally appropriate to their specific environmental, social, political, and market contexts. The framework we have outlined invites ongoing participation of individuals, communities, and institutions—irrespective of scale or starting point—to evaluate and enhance the qualities of adaptive capacity that emerge from divergent adaptation pathways in farming systems. Understood as an ongoing process, diversifying pathways can be an inclusive space to "call in" producers, policy makers, and other actors to take steps toward enhancing adaptive capacity to pressing global threats. We hope that diversifying farming systems to increase adaptive capacity can help build a more representative coalition of farmers and support emancipatory agroecology movements, adding to the well-developed scholarship and practice of agroecology.

AUTHOR CONTRIBUTIONS

MP-R conceived and presented the idea and wrote the initial draft. MP-R, PB, and TB developed the framework, literature review, theory, co-wrote iterative drafts, edited all cases, coordinated meetings, facilitated discussions, and managed contributions. PB and PS drafted the foodborne pathogen case. MM, YS, SB, JO, and TB drafted the drought case. AG, KD, CK, MP, and MP-R drafted the marginal land case. AI, FC, and AD drafted the labor case. AC, MP-R, KD, and JL drafted the land access and tenure case. AI and CK edited the entire article and assisted in conceptual refinement. MM, KE, and MP provided helpful edits to the full text. AG created **Figure 1** and **Figure 3** with conceptual development by MP-R, PB, and TB. TB and MP-R developed **Figure 2**. PB led the development of **Box 1**, **Tables 1**, **2** with TB, AI, and MP-R. All authors discussed the broad ideas as part of the Diversified Farming Systems Center weekly meetings during Spring 2019 at the University of California, Berkeley, were invited to edit the full manuscript, and approved publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2021.564900/full#supplementary-material

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Puerto Rican Farmers' Obstacles Toward Recovery and Adaptation Strategies After Hurricane Maria: A Mixed-Methods Approach to Understanding Adaptive Capacity

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Farmers across the globe are experiencing compounding shocks that make evident the need to better understand potential drivers and barriers to strengthen adaptive capacity. This is especially true in the context of a disaster, where a disruption in the natural and built environment hinders livelihood strategies and exposes the underlying dynamics that perpetuate vulnerability to natural hazards. As such, the interconnections of structural and individual attributes must be considered when evaluating adaptive capacity. This paper uses a convergent mixed-methods approach to assess Puerto Rican farmers' actual and intended adoption of adaptation practices, in light of the obstacles they faced toward recovery after 2017's category four Hurricane Maria, to contribute to better understanding adaptive capacity. This study uses data from 405 farmers across Puerto Rico (87% response rate), surveyed 8 months after Maria by agricultural agents of the Extension Service of the University of Puerto Rico at Mayagüez. Quantitative data was assessed through negative binomial regressions (actual adoption) and generalized linear models (intended adoption), while qualitative data (reported obstacles) were analyzed through thematic analysis. This study found that almost half of farmers adopted an adaptation practice after Maria, and that in many cases, broader structures, such as systems of governance, farmers' social networks, and infrastructure, affect adaptive capacity more than individual perceptions of capacity. Future adaptation strategies and interventions, especially in the context of disaster, should consider the extent to which structural factors hinder individuals' ability to prepare for, respond, and recover from the impacts of these shocks. Our results show that there might be opportunity to enact new systems in light of catastrophic events, but this does not solely depend on individual actions. The mixed-methods approach used can inform future studies in better assessing adaptive capacity from a standpoint that incorporates individual and structural components.

Keywords: climate change, islands and archipelagos, food systems, farmers' decision-making, disaster and climate risk reduction

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INTRODUCTION

Farmers across the globe are facing multiple compounding shocks, such as devastating hurricanes and severe droughts. These impacts are likely to become more frequent or intense in a changing climate (IPCC, 2014; Zhang et al., 2020), and thus it is increasingly important to understand the barriers and drivers to strengthen farmers' adaptive capacity—the available resources or assets to mitigate, prepare for, counter, and recover from impacts (Brooks and Adger, 2005; Gallopín, 2006; López-Marrero, 2010; Wisner et al., 2012; Cinner et al., 2018; Barnes et al., 2020; Phillips et al., 2020). Since farmers are embedded within social-ecological systems, it is important to recognize that adaptive capacity is comprised of various determinants that may extend beyond the individual to the institutional or systemic levels: political regulations, poverty, vulnerability to extreme events, and others (Reed et al., 2013; Shinbrot et al., 2019; Doran et al., 2020). Hence, when evaluating adaptive capacity, its determinants must be considered across scales (Adger, 2006; López-Marrero, 2010).

This assertion is especially true in the context of a disaster, where a disruption of the built and natural environments, as well as to day-to-day livelihood activities, reflects what resources and abilities people have available to manage the situation (Quarantelli, 1992; Wisner et al., 2004, 2012; Adger, 2006; Clay et al., 2018). Disasters are social-historical products that highlight people's vulnerability to natural hazards, which to a great extent is driven by people's exposure and sensitivity to those shocks, as well as their adaptive capacity (Brooks and Adger, 2005; Gallopín, 2006; Smit and Wandel, 2006; Ribot, 2014). Decreasing exposure and sensitivity to natural hazards—the rate of being subject to impacts, and the degree of change due to impacts, respectively—is difficult in places that due to geophysical and geographical conditions experience a higher prevalence of natural hazards (Gallopín, 2006; Smit and Wandel, 2006; López-Marrero, 2010; López-Marrero and Wisner, 2012). Thus, focusing on strengthening adaptive capacity is a crucial step in decreasing vulnerability.

As such, to better understand the adaptive capacity of farmers in a disaster context from both the individual and societal level, this paper uses a mixed-methods convergent design (Creswell and Plano-Clark, 2018) to examine the intended and actual adoption of adaptation practices and strategies of Puerto Rican farmers in the aftermath of Hurricane Maria, in light of the obstacles they faced to post-hurricane recovery. Hurricane Maria was the strongest category four hurricane to hit Puerto Rico in 89 years, and the first one to make a direct impact in 19 years (Castro Rivera et al., 2018; Bang et al., 2019). The hurricane made landfall in September 20, 2017, and it triggered a disaster that made visible how social and political dynamics in Puerto Rico exacerbate vulnerability to natural hazards (Moulton and Machado, 2019; Bonilla, 2020). Almost 3,000 deaths are attributed to the lack of access to electricity, water, healthcare, and other basic needs after Maria's passage (Santos-Burgoa et al., 2018; Bonilla, 2020).

Prior to Maria, agriculture in Puerto Rico was experiencing advancements in production and its recognition as an important sector in a place where around 85% of food is imported (Comas-Pagán, 2014; Irizarry-Ruiz, 2016; Álvarez-Berríos et al., 2018). Hurricane Maria changed this trajectory; the Puerto Rico Department of Agriculture reported that 80% of agricultural infrastructure and production were lost due to the winds and rains of Hurricane Maria, which made landfall just 2 weeks after category five Hurricane Irma impacted the territory. Both storms were part of the 2017 Atlantic hurricane season, which surpassed meteorological standards and was the costliest season in record (Bang et al., 2019). Both hurricanes affected many islands in the Caribbean, making evident that improving adaptive capacity among island systems is key to furthering adaptation.

Island states and territories are known to face additional vulnerabilities to climate change because of characteristics, such as exposure to sea-level rise and constant shocks, coupled with their small economies and territories, isolation, and dependence on imports (Graham, 2012; IPCC, 2014; Scobie, 2018; Kim and Bui, 2019). The 2017 hurricane season made evident those underlying conditions, including Caribbean governance structures that reflect neocolonial relationships (Bang et al., 2019; Borges-Méndez and Caron, 2019; Bonilla, 2020), and perpetuate the vulnerability of social-ecological systems (Quarantelli, 1992; Ribot, 2014). Given the importance of local agriculture for island food security in the context of response and recovery from shocks, understanding farmers' adaptation to climate change in light of a disaster—the set of decisions and processes that allow them to secure agricultural production while safeguarding their livelihoods (Brooks and Adger, 2005; Jezeer et al., 2019; Shinbrot et al., 2019)–may provide us with a clearer picture of the interplay between individual and structural factors in adaptive capacity.

The adoption of agricultural practices and management strategies, such as farm diversification of products and energy sources, amongst others, allow farmers to offset impacts in a changing climate and are key to adaptations that can support livelihood outcomes such as food security (Caswell et al., 2016; Akhter and Erenstein, 2017; Niles and Salerno, 2018; Fernandez and Méndez, 2019; Anderzén et al., 2020). These actions can allow farming systems to either return to the prior state before the event (i.e., "bounce back"), or to transform into new systems that are better suited to changing climatic circumstances (i.e., "bounce forward") (Payne et al., 2021). Either option, whether incremental (e.g., adopting cover crops) or transformative (e.g., changing from monoculture to diverse farming) is dependent on farmers' adaptive capacity—resources or assets farmers have access to, which play a key role in such decisions (Holt-Giménez, 2002; Caswell et al., 2016; Barnes et al., 2020; Wilson et al., 2020).

Adaptive capacity is multidimensional, and its determinants span from individual attributes, such as gender and financial assets, to material and governmental resources (**Table 1**). Approaching climate change adaptation by acknowledging adaptive capacity as a multidimensional component provides a framework to define what resources are needed to counter and recover from shocks in a given context (López-Marrero, 2010; López-Marrero and Yarnal, 2010; López-Marrero and Wisner, 2012). Understanding the role of how different capacities affect Puerto Rican farmers' actual and intended adoption of agricultural practices and management strategies after Maria, and their obstacles for recovery, can enable a more systematic assessment of the barriers and drivers to strengthen farmers'

TABLE 1 | Determinants of adaptive capacity as delineated by López-Marrero and Yarnal (2010).

Determinant of adaptive capacity	Description
Agricultural resources	Resources available to carry out current or post-hazard agricultural production (e.g., seeds, agricultural inputs, agricultural machinery, etc.).
Economic resources	The economic, and financial resources (e.g., monetary) farmers have (e.g., earned income, savings, credit, pensions, transfers from the state, insurance, etc.), and that are available (e.g., monetary aids) for adaptation or recovery.
Human resources	The skills (e.g., training), knowledge (e.g., formal education), and awareness (e.g., of adaptation options, the nature and evolution of hazards), experience, ability to work, and good health (e.g., food secured) that enable farmers to pursue adaptive strategies before hazards, and afterwards for recovery.
Institutions	The availability of critical institutions that promote and support adaptive strategies amongst farmers, along with the way they operate and are structured (e.g., transparent decision-making, institutional requirement).
Material resources and technology	The infrastructure (e.g., transport, drainage systems, housing, utilities) and the production equipment and materials available for adaptation and recovery; along with technological systems (e.g., communication systems, protective structures) available for adaptation and recovery.
Natural resources	The resources present in the physical environment (e.g., raw materials, biodiversity) and/or the services they provide (e.g., pollination) that are useful for adaptation.
Perception/cognition	The different views of nature people have, perceptions of hazards (e.g., likelihood of occurrence and potential damages), perceived effectiveness of past adaptive actions, perceived alternatives and perceived capacity to undertake them or act upon hazard exposure.
Political resources	Power, right, development of political capabilities or claims farmers can make on the state, institutions, or those more powerful than they are (e.g., unions, lobbying, access to legislature, etc.).
Social resources	The social resources (e.g., informal-horizontal networks, social mobilization, collective actions, and relations of trust, reciprocity, and exchange) upon which farmers can draw for adaptation and recovery.

Language was modified to focus on farmers (e.g., instead of using the word people); "agricultural resources" was added, and "food security" was added to "human resources." Table content is from López-Marrero (2010).

adaptive capacity, further providing us a better picture of how broader structures, beyond individual attributes, effect action (Rodríguez-Cruz and Niles, 2021).

Farmers' decision-making around climate change adaptation has been studied from various disciplinary perspectives (Prokopy et al., 2019; Ranjan et al., 2019; Foguesatto et al., 2020), with mixed conclusions on the extent to which adaptive capacities impact adaptation outcomes. For example, in contrast to studies in low and middle-income countries (e.g., Kassie et al., 2015), adoption of new behaviors among mainland US farmers is not highly dependent on natural and agricultural resources (e.g., land tenure, farm size, etc.) (Prokopy et al., 2019). Furthermore, social, governmental, and institutional resources, such as belonging to farmer networks, subsidies, and regulations, influence the degree to which farmers adopt new practices. For example, access to information sources through social or institutional networks of support that increase farmers' knowledge on what strategies benefit them, and how to carry them out, has been shown to be positive for farmers' adaptation (Dang et al., 2018; Bagagnan et al., 2019; Luu et al., 2019; Raza et al., 2019). Studies have also shown that psychological factors, such as perceived vulnerability and capacity, for example, play a role in farmers' adaptation decisions. Wang et al. (2019) found that perceived vulnerability and severity had an effect on Chinese farmer's intention to adopt pro-environmental practices, and Niles et al. (2016) found that perceived capacity linked to both intended and actual adoption of new practices among New Zealand farmers.

Though these studies have shown the importance of integrating different aspects of adaptive capacity, there is still a gap in triangulating the role of individual and structural aspects of adaptation behaviors (Foguesatto et al., 2020; Wilson et al., 2020; Rodríguez-Cruz and Niles, 2021). Much of the literature has either focused on the intention to adopt or actual adoption, and has not considered both within the same population (Niles et al., 2016). Furthermore, adaptation literature has been limited in intersecting individual (intrapersonal) and societal (interpersonal) components. Here we address these gaps by focusing on Puerto Rican farmers during their recovery period from Hurricane Maria and examine the multiple facets of adaptive capacity and their relationship to intended and actual adaptation practice adoption. This paper intends to contribute to current conversations on how to understand and approach adaptive capacity in a way that analyzes multiple components. We do not aim to assess the efficacy of the practices and strategies toward adaptation. Rather, we aim to provide a different methodological perspective that can improve our assessment of what may be needed to improve adaptive capacity.

As such, we ask the following: (1) What obstacles did farmers experience that thwarted the recovery of their farms after Hurricane Maria, and what determinants of adaptive capacity are reflected within them? (2) What were farmers' actual and intended adoption of agricultural practices and management strategies after Hurricane Maria? (3) What determinants of adaptive capacity explain actual and intended adoption, and how do they compare? (4) How do farmers' reported obstacles to recovery post-Maria compare with intention to adopt and adoption of farm management practices and adaptation strategies??

The literature in the Caribbean and Central America, regions affected by Atlantic hurricanes, has shown that adaptation is highly dependent on structural (social, governmental, and institutional) and income-related factors (economic, material and technological). Thus, we expect that variables reflecting these factors will be significant in both actual and intended adoption (H1). Furthermore, though research has shown that perceptual factors are not pivotal to Puerto Rican farmers' adoption of practices (Rodríguez-Cruz and Niles, 2021), we expect these factors to be related to intended practices (H2). Lastly, it is known that Hurricane Maria caused significant damage in Puerto Rico, and that subsequent recovery efforts failed to safeguard lives and wellbeing (Santos-Burgoa et al., 2018; Bonilla, 2020). Thus, we expect that farmers' self-reported obstacles will reflect the role of broader structures (governmental, institutional, and societal) in the recovery of their farms (H3), as well in the type of practices they actualized, and intended to carry out (H4).

MATERIALS AND METHODS

Survey Sample

A mixed-methods survey, informed by previous studies (Spence et al., 2011; Haden et al., 2012; Niles et al., 2015; Niles and Mueller, 2016; Singh et al., 2017), was developed in English, and translated to Spanish, to capture Puerto Rican farmers' perceptions and opinions around their experience with Hurricane Maria and climate change. The overall objectives of the main project focused on understanding farmers' adaptation and food security outcomes in light of farmers' recovery from Hurricane Maria (Rodríguez-Cruz and Niles, 2020). The study received approval by the Institutional Review Board of the University of Vermont in December 2017.

A pilot with a pool of diverse farmers ($n = 31$) was carried out in February 2018, and results were shared with partners at the Extension Service of the University of Puerto Rico at Mayagüez. The survey received minimal language and structure corrections.

Data used for the present study were a sub-set of survey questions (both closed response and open-ended), concerning farmers' demographics, questions that reflected adaptive capacity resources, adaptation perceptions, actual and intended practices, and reported obstacles. Some variables were converted (e.g., Likert scale to binary) to better group individuals, and because some had concentrated answers in two items (e.g., agree and strongly agree). **Table 2** shows independent variables used in this study, and **Table 3** shows agricultural practices and management strategies asked about in the survey.

The survey was deployed by local agricultural agents of the Extension Service who acted as enumerators of the survey, between May and July 2018, 8 months after Hurricane Maria. To access a diverse and substantial number of farmers, 440 surveys were distributed in the five regions that the Extension Service covers across Puerto Rico: Arecibo, Caguas, Mayagüez, Ponce, and San Juan. One hundred surveys were sent to each of the administrative offices of Caguas, Ponce, and San Juan; 70 were sent to each in Arecibo and Mayagüez, as per recommendation of Extension personnel. Agricultural agents then randomly surveyed farmers that were connected to Extension or had received services from it in municipalities of each region. This approach was recommended by Extension colleagues to access a diverse range of farmers (e.g., mixed, banana, plantain, dairy, poultry, etc.) from across Puerto Rico.

Place and Population

Puerto Rico is the smaller of the Greater Antilles of the Caribbean. It is an unincorporated territory of the United States, with a population of ∼3.3 million people (US Census Bureau, 2020). As most islands in the region, Puerto Rico has seen a decrease in farms and food production since the 1990s due to trade liberalization and unbalances, economic situations, influx of imports, and other external and internal factors that make local food production and access difficult (Weis, 2007; FAO, 2014; Lowitt et al., 2015; Irizarry-Ruiz, 2016). While it produced about 40% of its food needs in the 1980s, Puerto Rico currently only produces around 15% (Carro-Figueroa, 2002; Gould, 2015; Irizarry-Ruiz, 2016; Gould et al., 2017). The territory is also undergoing a social and political crisis due to high debt (Bueno, 2017; Félix and Holt-Giménez, 2017; Bonilla, 2020). Within that context, the agricultural sector was experiencing improvements in production, access to local markets, and other opportunities prior to Maria (Comas-Pagán, 2014; Irizarry-Ruiz, 2016; Gould et al., 2017). Governmental and community-based efforts were focused on supporting current and new farmers before 2017's hurricanes. These efforts were halted by the impacts of both hurricanes Irma and Maria. The Puerto Rico Department of Agriculture (2018) reports that these two hurricanes caused $2 billion in damages, Maria being the most significant of the two in terms of agricultural losses ($228 million in production losses, and $1.8 billion in infrastructural losses). For example, reports indicate that the plantain sector suffered $72 million in damages, while the banana sector suffered $19 million. Other heavily affected sectors were coffee ($18 million), dairy ($14 million), and poultry ($6 million).

Though both impacts decimated Puerto Rico's agricultural sector, farmers have experienced a significant quantity of natural hazards since 2017, such as intense storms, and severe droughts (Gould, 2015; Díaz et al., 2018; López-Marrero and Castro-Rivera, 2018, 2019; Rodríguez-Cruz and Niles, 2021). Even a category one hurricane can easily damage local agriculture. Many of Puerto Rico's high value crops, such as coffee, bananas, and plantains, are very susceptible to temperature change and moderate winds. Moreover, important farmland is located in coastal areas, which is susceptible to erosion, and seawater intrusion to aquifers (Díaz et al., 2018). US Congress' Fourth National Climate Assessment (2018) concluded that rainfall patterns will change, and water availability will likely decline for Puerto Rico, coinciding with rising temperatures that contribute to the occurrence of recurring droughts in the future. Those impacts are occurring simultaneously with stronger storms (Díaz et al., 2018).

Puerto Rico's farmers produce mainly for domestic markets, and work small to medium farms according to the USDA. Subsistence farming is not typical in Puerto Rico. The 2018 census states that most farmers in the territory (or principal operators) have a household income <$20,000, significantly less than the US average, which exceeds $60,000 (USDA ERS, 2020). Puerto Rico's average household income is $20,539 (US

TABLE 2 | Independent variables used in quantitative models.

Determinant	Question/statement	Variable name	Scale	Rationale
Agricultural, natural	What agricultural products have you produced, currently produce or plan to produce in the future on your farm? Check all that apply.	Agricultural production	Aggregated count variable	This variable is a proxy for agricultural diversity, which has been shown to provide benefits (e.g., ecosystem services) that increase farms resistance to impacts, and supports recovery.
Agricultural, natural	How many *cuerdas* do you manage in your farm?	Farm size	Continuous	Farm size has been shown to be related to livelihood and adaptive capacity outcomes across regions.
Human	What is the highest level of education you have completed? Mark one	Education	Binary (1 = Some college or more; 0 = High school diploma or less)	Attaining formal education levels is related to livelihood and adaptive capacity outcomes through increasing household assets (e.g., higher income)
Human	What is your gender?	Gender	Binary (1 = Female; 0 = Male)	Farmers identified as females have been shown to face several constraints in achieving livelihood outcomes, such as food security.
Human	How many years have you been a farmer?	Years as farmer	Continuous	This variable was highly correlated with *age*. This variable was included because years farming may reflect traditional and local knowledge of farming. As well as farmers experiences with past events.
Physical, political, institutional, governmental	In what municipality your farm is located?	Metropolitan	Dummy (1 = Farm in metropolitan municipality; 0 = Not metropolitan)	Puerto Rican municipalities are categorized by the *Junta de Planificación* as metropolitan based on location (near big cities or coast) and population size. Metropolitan municipalities have higher access to physical assets (e.g., roads) and governmental and institutional agencies.
Physical, institutional, governmental	-	Extension region	Categorical (dummy)	A categorical variable based upon reported municipalities where farms are located. This variable was created to group farmers based on the Extension office that gives them service.
Economic, governmental	Are you a "bona fide" farmer of the Department of Agriculture?	Bona fide	Binary (1 = Yes; 0 = No)	To be bona fide, 51% or more of farmers' income must come from agriculture. This certification provides farmers with economic benefits (e.g., exemptions, incentives, etc.) and formal recognition by the Puerto Rico Department of Agriculture.
Economic	What is your approximate household income, including all far and off-farm income?	Household income	Binary (0 = <$20,00; 1 = $20,000 or more)	Household income has been a key variable in reflecting people's adaptive capacity. It is assumed that a higher income relates to access to other assets and higher wellbeing.
Economic	How do you sell your products? Check all that apply	Access to markets	Aggregated count variable	Having a diversity of ways to sell products may be beneficial for farmers' adaptive capacity in that it allows them to have more opportunities in selling their products.
Social, institutional	Which of the following organizations and institutions, if any, have you received information from related to adapting to climate-related impacts?	Contact scale	Aggregated count variable	Farmers were asked about the organizations and institutions that have provided them with information on climate change adaptation. This variable allows us to proxy social networks and access to diverse sources of support, which aid in adaptive capacity.
Perception/cognition	I feel that I have the capacity to change my agricultural practices to adapt to future potential extreme weather events like Hurricane Maria.	Perceived capacity	Binary (1 = Strongly agree, 0 = agree and below)	Perceived capacity composes diverse tested behavioral theories, such as the Theory of Planned Behavior. Individuals' perceived capacity to change a behavior or assume a new one has been shown to preclude actual behavior. Nevertheless, its role on behavior change varies. Furthermore, perceived capacity can reflect access to external assets that may motivate the individual to change or assume new behavior.
Perception/cognition	I believe my farm is vulnerable to future extreme weather events like Hurricane Maria.	Perceived vulnerability	Binary (1 = Strongly agree, 0 = agree and below)	Perceived vulnerability or risk can be a motivator for individuals to enact change that reduces that vulnerability or risk. Moreover, the perception of vulnerability can be useful to understand an individual's social-ecological context.
Vulnerability context	How would you describe the damages, if any, caused by Hurricane Maria to your farm?	Hurricane damage	Binary (1 = Total loss, 0 = Significant loss or other damages)	This variable is used as a proxy that reflects farmers' exposure and sensitivity to Hurricane Maria. Direct damage from a natural hazards can also reflect the severity of the impact.

Each variable is categorized to reflect an adaptive capacity determinant.

TABLE 3 | Categorized agricultural practices and management strategies to adapt to future extreme weather events like Hurricane Maria asked to farmers.

Category	Practice or management strategy	KR-20	α
Market oriented and capital-intensive growth practices and strategies	Acquire new insurance or improve current insurance	0.6471	0.6551
	Acquire solar panels		
	Apply more synthetic inputs		
	Expand agricultural land		
	Improve irrigation systems		
	Increase tillage		
	Seek new agricultural markets		
Ecological transition practices	Crop rotations	0.8005	0.6725
	Decrease tillage		
	Diversify crops		
	Integrated pest management		
	Switch from a perennial to an annual crop		
	Switch from an annual to a perennial crop		
Natural design practices	Collect rainwater for irrigation	0.8338	0.8460
	Contouring		
	Plant native species		
	Plant trees to reduce erosion		
	Use compost		
	Use mulch		

Frequencies and reliability measures are shown: Kuder-Richardson Formula 20 for actual adoption's binary variables (KR-20), and Cronbach's alpha for intended adoption's Likert-scale variables (α).

Census Bureau, 2020). The USDA defines a Puerto Rican farm as a location where $500 or more of agricultural products are produced or sold. Between 2012 and 2018, Puerto Rico saw a decrease in farms (USDA NASS, 2020). There were 13,159 farms in 2012—when the last census was carried out–, with, 8,230 farms reported in the current census (USDA NASS, 2020). Today, most farms are <100 *cuerdas* (Puerto Rico's traditional land measure) or 97 acres (an average of 59.3 *cuerdas*), and are mostly family or individual farms. There are large farms that run extensive monocultures, but many of the small-medium farms produce a diverse array of agricultural products (Álvarez-Febles and Félix, 2020). It is important to note that many other farms, such as community-supported agriculture projects and others, are not counted in the census or are not directly linked to the Puerto Rico Department of Agriculture.

Qualitative Analysis

The survey asked farmers to state at least three obstacles faced during the recovery of that their farming operations. Farmers' responses to this open-ended question were analyzed using double coding through thematic analysis with *a priori* codes (Creswell, 2014, 2016). Responses were translated from Spanish into English by the first author, and then transcribed to a Microsoft Word document, which was uploaded to NVivo v.12 (QSR International, 2019). Given the purposes of this study, an *a priori* coding frame, accompanied by code definitions, was used to categorize the reported obstacles into nine nodes (or themes) following the nine determinants of adaptive capacity adapted from López-Marrero (2010) shown in **Table 1**. Authors LARC and MM (coders) first agreed upon the codes and coded the first 10 responses together. In order to establish transparency and consistency within the coding (O'Connor and Joffe, 2020), intercoder reliability (ICR) was evaluated following a first round of coding by quantifying the degree of consensus using percentage of agreement. More than 90% is considered highly reliable (Lavrakas, 2008). Two nodes, *agricultural resources* and *economic resources*, did not score more than 90% agreement after the first round of coding. Thus, after the coders discussed divergences and reached consensus, a second round of coding was undertaken. The second assessment successfully achieved more than 90% ICR in all nodes (**Supplementary Table 1**). Codes with the highest frequency within each theme, and quantification of such themes to identify coverage and percentages, were evaluated using Nvivo 12's hierarchy chart wizard and word cloud function.

Quantitative Analysis

The survey asked farmers to state the agricultural practices and management strategies that they were considering adopting in the future or that they had adopted since Hurricane Maria (∼8 months prior to the survey) in a close-ended question with pre-coded responses. The responses included a list of practices which list of practices was developed based upon conversations and recommendations from colleagues at the Extension Service and at the University of Vermont with expertise in agriculture and climate change. The survey asked, "Which of these agricultural practices and management strategies, if any, might you adopt in the near future to adapt to future extreme events like Hurricane Maria?" (**Table 3**). Hereafter, these practices and strategies will be referred to as "adaptation practices." The 22 practices were

TABLE 4 | Descriptive statistics of independent variables.

Variable	Scale	Frequency (%)	Mean ± SD	n
Contact scale	Continuous	–	2.3 ± 2.1	398
Agricultural production	Continuous	–	2.2 ± 2.0	402
Bona fide	Yes	210 (52.8)	–	398
	No	188 (47.2)		
Education	High school diploma or less	131 (32.7)	–	401
	Some college or more	270 (63.3)		
Farm size	Continuous	–	58.1 ± 98.5	383
Gender	Female	55 (14.0)	–	395
	Male	340 (86.0)		
Income	<$20,000	138 (36.4)		379
	More than $20,000	241 (63.6)		
Markets	Continuous	–	1.2 ± 1.0	401
Metropolitan	Metropolitan	229 (58.0)	–	398
	Non-metropolitan	169 (42.0)		
Perceived capacity	High perceived capacity	192 (50.4)	–	381
	Low perceived capacity	189 (49.6)		
Perceived vulnerability	High perceived vulnerability	264 (66.5)	–	397
	Low perceived vulnerability	133 (33.5)		
Region	Arecibo	57 (14.3)	–	398
	Caguas	88 (22.1)		
	Mayagüez	76 (19.1)		
	Ponce	92 (23.1)		
	San Juan	85 (21.4)		
Damage	Total loss	229 (57.4)	–	399
	Significant loss	170 (42.6)		
Years farming	Continuous	–	20.5 ± 15.3	392

Frequency and percentages of responses, as well as mean and standard deviation (SD) are included.

included as a list in a table. The first column stated, "Currently in use" (binary, yes or no), and the subsequent columns represented a 5-point Likert scale for adoption (from very unlikely to very likely). The binary column was used to assess actual adoption, and the Likert scale to assess intended adoption. The list was also intended to include practices recommended for adaptation or for conservation of natural resources, as well as other conventional or common practices in Puerto Rico and the contiguous United States. It is important to note that the objective was to assess what practices or strategies farmers understand to be helpful for adaptation, and not to evaluate if those decisions are adaptive or maladaptive. Variables that had $n < 20$ were excluded from the analysis (e.g., "stop farming" and "forage conservation"), since they represented <5% of total respondents.

Actual Adoption

Actual adoption of agricultural and management strategies following Hurricane Maria was assessed through binary variables where farmers indicated "currently in use," as noted above. We used Kuder-Richardson-20 Reliability Tests in Stata 15.1, which test internal consistency or scale reliability of binary items (Kuder and Richardson, 1937) and range from 0 to 1 in ascending reliability (**Table 4**). This test is similar to assessing Cronbach's alpha, which evaluates internal consistency of scale variables (Nunnally, 1978). Two categories had KR-20s > 0.80, which are acceptable determining internal consistency or reliability of a group of items, and one had a coefficient > 0.60, which is reasonable (Nunnally, 1978). Each groups' variables were summed to create three new aggregate count variables for analysis: (1) Market oriented and capital-intensive growth, (2) Ecological transition practices, and (3) Natural design, as well as fourth aggregate count variable of all practices combined (**Table 3**). The first group had seven practices; the second and third groups were composed of six.

Model development considered distribution of the count variables. We implemented a Poisson regression to test the model, but model assumptions were not met on multiple factors (Likelihood ratio test showed over dispersed data: LR test of alpha = 0: chibar2(01) = 649.22, Prob ≥ chibar2 = 0.000, and the Poisson regression assumption of identical means and variances was not met). Instead, we used a negative binomial regression (nbreg) to fit over-dispersed data. We developed four nbreg models, with clustered errors for municipalities, utilizing Stata 15.1.

Intended Adoption

Intended adoption was evaluated through scale variables (5-point Likert) of the list of practices and management strategies. Three

TABLE 5 | Coding coverage—the percentage of content coded-for each set of adaptive capacity themes.

Theme	Coverage	Most prevalent references
Institutions	27.0%	Issues with the government, such as frustration with government bureaucracy, insufficient support from governmental institutions, and lack of general aid
Economic resources	26.2%	Loss of income, delayed insurance payment, cost of workforce, and financial assets (e.g., available cash)
Material resources and technology	26.1%	Access to electricity, machinery, and water; physical access to farms (e.g., because of fallen trees and landslides).
Agricultural resources	22.8%	Lack of seeds
Human resources	7.9%	Lack of laborers or human assistance to help with post-hurricane cleanup and reconstruction.
Political resources	5.5%	Negotiating with governmental agencies
Natural resources	4.3%	Pests, lack of flowers, erosion
Social resources	2.8%	Focused advice from specialized advisors.
Perception/cognition	0.1%	Feelings of abandonment

scale variables were created to understand likelihood of intended adoption, using the same categorization of actual adoption scales, with Cronbach alpha acceptable at >0.65 (Nunnally, 1978). As with all of the actual adoption variables, a single variable was created with all the intended adoption practices (alpha = 0.86). We used generalized linear models with clustered errors around municipalities to account for spatial correlation (Nichols and Schaffer, 2007). Distribution of scale variables show similitude to both Gaussian and Gamma distributions. In order to choose the best family distribution to build the models, Akaike information criterion (AIC) and the Bayesian information criterion (BIC) were used to compare across generalized linear models with either Gaussian or Gamma distributions. The models yielded better fit with Gaussian distributions (**Supplementary Table 2**). Thus, the generalized linear models used Gaussian as the family choice, and "Identity" as the link choice.

RESULTS

Descriptive Statistics

A total of 405 farmers answered the survey through Extension enumerators, resulting in an 87% response rate. Farmer and farm characteristics, which were categorized under different adaptive capacity resources, varied across respondents (**Table 4**). On average, farmers surveyed had 58 *cuerdas* (56.3 acres or 23 hectares), were 54 years old, and had been farming for 20 years; results that align with census data for Puerto Rican farmers (USDA NASS, 2020). The majority of respondents were male (86%) and reported a household income of $20,000 or more (64%), which also aligns with recent census data. Nevertheless, we had an overrepresentation of bona fide farmers (53%). Farmers reported being connected to an average of two organizations or institutions (min = 0, max = 11), that provide them support around climate change adaptation, and reported selling their products to one of the listed venues, on average (min = 1, max = 5).

Qualitative Analysis Results

While 345 farmers (90%) responded that they had faced significant obstacles toward recovery, only 333 provided responses to the open-ended question. Farmers identified many obstacles to their recovery related to their adaptive capacity (agricultural resources, economic resources, human resources, institutions, material resources and technology, natural resources, perception/cognition, political resources and social resources) (**Table 1**). **Table 5** shows the coverage percentages of each of the identified themes, and also displays the most prevalent references within each theme. Obstacles most mentioned by farmers were related to institutions and institutional support (27.0%), material resources and technology (26.1%), economic resources (26.2%), and agricultural resources (24.3%). Obstacles that fell under themes of perception/cognition (0.07%), natural resources (2.6%), and social resources (4.3%) were the least mentioned. Within each theme, we also evaluated the most prevalent references within a theme.

In addition, from the cloud analysis, we can see that the top 10 words mentioned in farmer responses to our open-ended question regarding their top obstacles to recovery overall were: "lack," "insurance," "seeds," "agricultural," "electrical," "energy," "economic," "aid," "laborers," and "water." Many of these words reflected resources or structural components of a system (e.g., electricity, energy, aid) that are often related to institutional support.

Quantitative Analyses Results

Actual Adoption Results

Figure 1 shows farmers' reported actual adoption and management practices. Overall, the top five practices implemented after Hurricane Maria were: integrated pest management ($n = 97$, 24.4%), crop rotation ($n = 84$, 21.2%), crop diversification ($n = 78$, 19.6%), contouring ($n = 68$, 17.1%), and composting ($n = 65$, 16.2%). We find that 49% of farmers adopted any new practices after Hurricane Maria.

Table 6 shows results with significance for the four models that evaluated the relationship between actual adoption outcomes and adaptive capacity resources or assets. **Supplementary Table 3** shows full model results. In the first model (actual adoption of all practices) we found that farmers with higher levels of education ($\beta = 0.5780$, IRR = 1.7824)

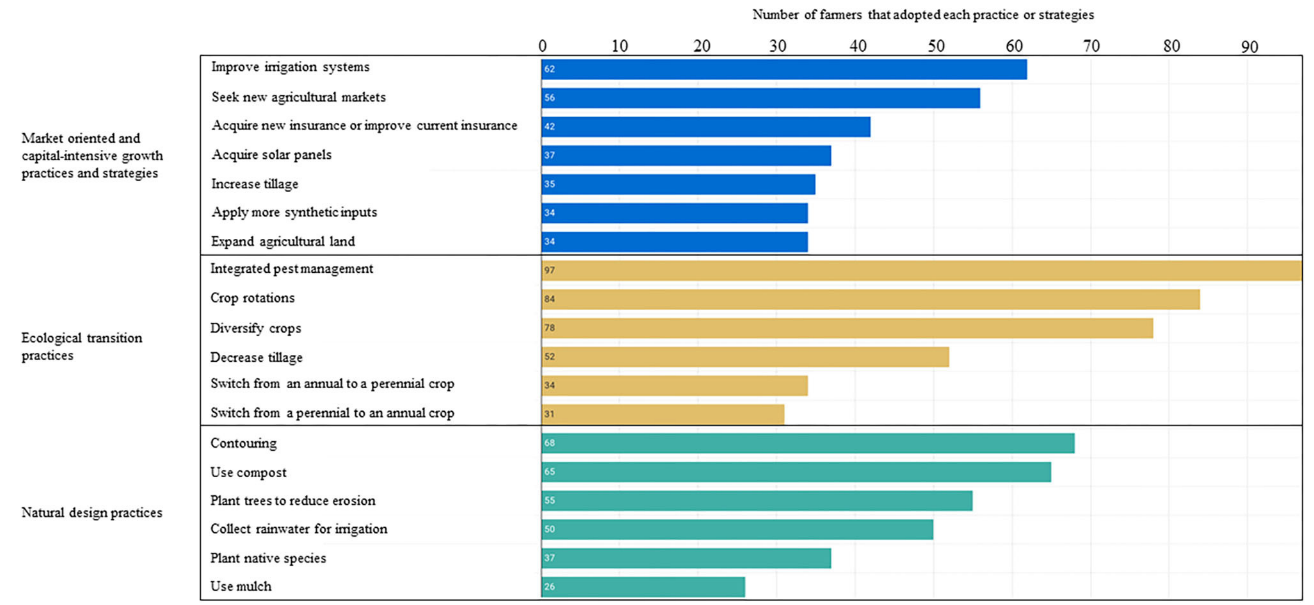

FIGURE 1 | Farmers' reported actual adoption of adaptation practices and strategies after Hurricane Maria.

TABLE 6 | Significant results for four separate negative binomial regression models predicting actual adoption of farm and management practices.

Models and dependent variables	Independent variables	β	p
Model 1: Actual adoption of all practices	Education*	0.5780	0.003
	Damages*	0.6665	0.001
Model 2: Actual adoption of market oriented and capital-intensive growth practices and strategies	Damages*	0.7343	0.000
Model 3: Actual adoption of ecological transition practices	Agricultural production*	0.1207	0.014
	Education*	0.5483	0.001
	Damages*	0.6467	0.002
Model 4: Actual adoption of natural design practices	Contact scale*	0.1453	0.013
	Education*	0.9138	0.000
	Farm size*	−0.0041	0.038

Supplementary Table 3 *shows full results, including estimates (β), robust standard errors (SE), significance (p), 95% confidence intervals (CI), and incident rate ratios (IRR).*
**p < 0.050.*

and those that reported total loss of their farms (infrastructure and production) due to hurricane winds, rain, and landslides ($\beta = 0.6665$, IRR = 1.9474), were more likely to report higher number of practices adopted after Hurricane Maria (**Table 6**). We expected that variables around structural and financial assets, such as social, agricultural, economic, and material resources would be significant. These results were not aligned with that expectation (H1).

In examining the specific types of market oriented and capital-intensive growth practices and strategies adopted (Model 2), we found that total farm loss (damages) was the only significant variable ($\beta = 0.7343$, IRR = 2.0840, $p < 0.05$) (**Table 6**). This result is counterintuitive because no variable related to economic or material resources was found significant (H1). In model 3, actual adoption of ecological transition practices, we found that the number of agricultural products produced ($\beta = 0.1207$, IRR = 1.1282), farmers' levels of formal education ($\beta = 0.5483$, IRR = 1.7303), and reporting total loss of farms ($\beta = 0.6467$, IRR = 1.9093) were significant predictors of adoption ($p < 0.05$) (**Table 6**).

Finally, we found in the fourth model (Actual adoption of natural design practices) that the number of reported organization or institutions that have provided services to farmers (contact scale) ($\beta = 0.1453$, IRR = 1.1564), education ($\beta = 0.9138$, IRR = 2.4936), and farm size ($\beta = -0.0041$, IRR = 1.0000), were significantly correlated ($p < 0.05$) with conservation practice adoption after Hurricane Maria (**Table 6**). These results demonstrate that farms with higher number of contacts and higher education (or greater social and human

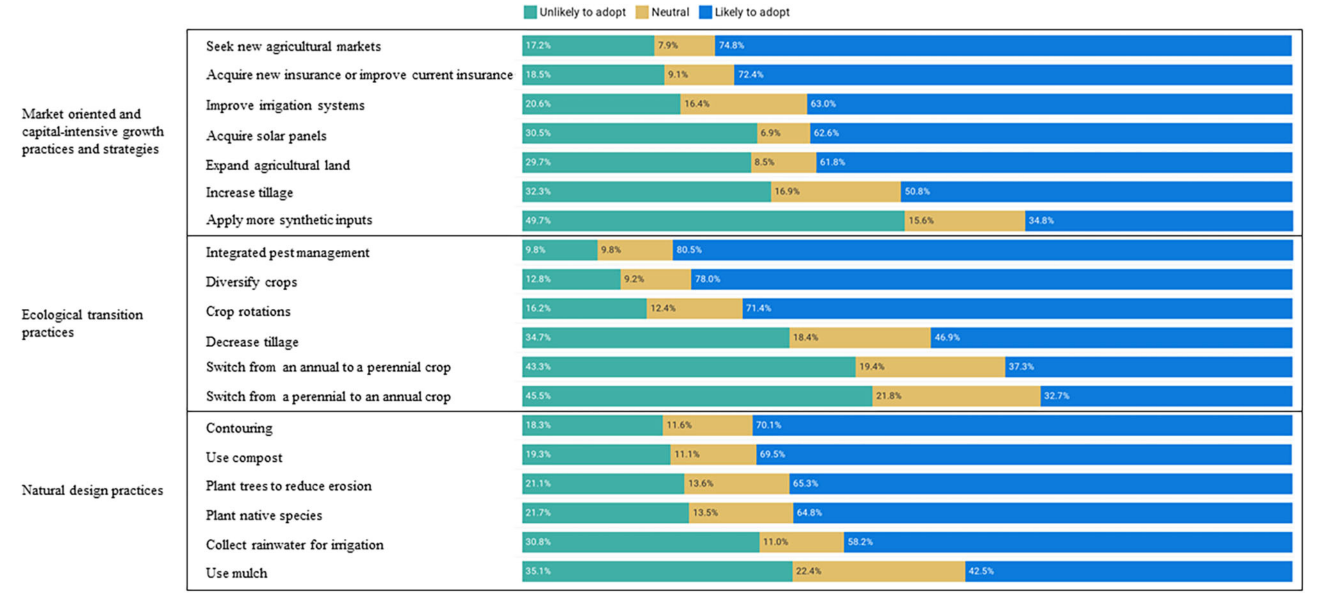

FIGURE 2 | Farmers' intended adoption of adaptation practices and management strategies after Hurricane Maria.

capital) were more linked to adoption, while larger farms were less likely to have adopted conservation practices.

Intended Adoption Results

Figure 2 shows farmers' intended adoption practices and strategies, and **Supplementary Table 4** shows the tabular results. Respondents' top reported practices and management strategies to adopt in the future (likely and very likely to adopt) were: integrated pest management (80.5%), diversification of crops (78.0%), seeking new agricultural markets (74.8%), acquiring new insurance or improving current insurance (72.4%), and crop rotations (71.4%). In general, intention to adopt results contrasted with those of actual adoption.

Four generalized linear models were carried out to assess intended adoption outcomes (**Table 7**). Full model results are shown in **Supplementary Table 5**, while significant results are shown in **Table 7**. We found no significant variables in Model 5 (Intended adoption across all practices and management strategies). In Model 6 (Intention to adopt market oriented and capital-intensive growth practices and strategies), perceived capacity significant ($\beta = 0.2343$, $p < 0.05$) was correlated with higher adoption intention (**Table 7**). In the 7th model (Intended adoption of ecological transition practices) being a bona fide farmer ($\beta = -0.3243$, $p < 0.05$) was negatively correlated with intention to adopt, meaning that those that reported being part of that program of the Puerto Rico Department of Agriculture, had lower intention rates to adopt such practices (**Table 7**). In the last model (8), exploring intended adoption of natural design, two variables were significant ($p < 0.05$). Producing a higher number of agricultural products (Agricultural production) was correlated with higher intention to adopt conservation practices ($\beta = 0.0902$) while reporting a total loss (damages) ($\beta = -0.2663$) was negatively correlated with intention to adopt those practices.

DISCUSSION

This paper assessed Puerto Rican farmers' actual and intended adoption of adaptation practices and management strategies in light of the obstacles they faced toward recovery after Hurricane Maria. It aimed to understand potential barriers and drivers for strengthening adaptive capacity through a mixed-methods approach, in order to provide a new approach to understanding adaptive capacity in a disaster context. We find that drivers for actual adoption vary from factors related to intended adoption of adaptation practices, and that almost half of all farmers in our survey had actually adopted a practice or strategy in the 8 months since Hurricane Maria. Furthermore, we find that the majority of farmers faced significant obstacles in their recovery, especially with institutional support, economic resources, and access to material resources and technology. Combining quantitative and qualitative data provided a richer understanding of how individual and structural factors intersect and reflect adaptive capacity.

Contrary to our expectations (H1), variables related to governmental, institutional, social, economic, and material resources were not the principal drivers for both actual and intended adoption. Instead, facing a total loss, and having a higher level of formal education were most related to actual adoption of adaptation practices. Furthermore, we did not find that perception factors were significantly related to intended adoption, rejecting H2. Instead, intention to adopt had varying factors across the different categories of practices and management strategies. Although variables used around the aforementioned resources were not significant, qualitative data analysis suggests that lack of broader structures of support, such as expected and timely aid, insurance payments, and access to services and supplies should be considered in farmers'

TABLE 7 | Significant results of generalized linear regression models for farmers intended adoption.

Dependent variables	Independent variables	β	*p*
Model 5: Intended adoption of all practices[a]			
Model 6: Intended adoption of market oriented and capital-intensive growth practices and strategies	Perceived capacity*	0.2343	0.048
Model 7: Intended adoption of ecological transition practices	Bona fide*	−0.3243	0.014
Intended adoption of natural design practices	Agricultural production*	0.0902	0.004
	Damages*	−0.2663	0.028

*Full results are shown in **Supplementary Table 5**, including estimates (β), robust standard errors (SE), and 95% confidence intervals (CI) are shown.*
$^*p < 0.05$.
[a]No variable was found to be significants.

decision-making around adaptation and recovery (H3 and H4). These findings reflect how we might include other questions in future surveys exploring disaster recovery. Future studies should consider how variables often used to assess the determinants of adaptive capacity, such as the ones used in this study, might not provide the nuanced information specific to a disaster context.

One of the major drivers for actual adoption in all models, except for natural design practices (Model 4), was "total loss." This contrasts with research in Central America with tropical agriculture farmers subjected to Atlantic hurricanes, where farmers' adoption of new practices was not significantly driven by damage experiences from extreme weather events, likely because of pre-existing low adaptive capacity (Harvey et al., 2018). Nevertheless, experiencing risk or climate-related impacts has been found to be a precursor of adaptation (Salerno et al., 2019). Farmers in our study that reported total loss were more likely to report a higher number of practices adopted overall, and in the adoption of market oriented and capital-intensive growth practices and strategies, as well as higher adoption of ecological transition practices. These findings are critically important for considering opportunities to rethink agricultural systems, and provide evidence that farmers may be willing to reconsider transforming their farming systems after experiencing significant damages that change their farming landscapes.

The finding that higher levels of formal education were linked to all actual adoption models, except market oriented and capital-intensive growth practices and strategies (Model 2), suggests that human assets may open doors to access other resources important for recovery and adaptation, likely through enabling formation of ties that increase social and structural support (López-Marrero and Wisner, 2012; Kassie et al., 2015; Caswell et al., 2016; Shah et al., 2019). For example, Model 4 showed that education was positively linked to adoption of natural design practices, as well as access to information sources. These findings align with Caribbean research suggesting that farmers who have external support are likely to adopt practices that support the environment, while sustaining their production (Lowitt et al., 2015; Saint Ville et al., 2016; Paul et al., 2017). Research outside the Caribbean supports this as well (Bagagnan et al., 2019; Žurovec and Vedeld, 2019).

These results suggest that being able to adopt practices to adapt to climate change or re-envision agricultural systems may occur among farmers with higher levels of human capacity. Future studies should focus on farm recovery processes after a significant shock that alters the working landscape to understand decision-making processes in rebuilding the system. Taken together, this suggests that total loss events, while catastrophic, do present opportunities for reinvention, if people have access to the necessary resources. These results further highlight the need for institutional support and capacity for farmers without formalized education, or social networks.

Our results also reflect other research showing that actual and intended adoption may not be driven by the same variables (Niles et al., 2016). While we expected (H2) to see perceived capacity be a notable factor predicting intended adoption as reflected in the Theory of Planned Behavior (Ajzen, 1991), as well as perceived vulnerability, we only find this significant in intention to adopt ecological transition practices (Model 6). These results further support that perceptual factors around climate change may not be pivotal in advancing adaptation in places where shocks are consistently experienced (Rodríguez-Cruz and Niles, 2021). On the other hand, being a "bona fide" farmer decreased likelihood of intending to adopt ecological transition practices (Model 7), which may help in increasing farmers likelihood of recovering their farms after a hurricane (Holt-Giménez, 2002; Rosset et al., 2011). This finding was counterintuitive since bona fide farmers are recognized officially by the government, which often provides them access to governmental and institutional resources. However, given the large number of farmers who reported institutional obstacles for recovery, it is possible that bona fide farmers did not receive benefits that might otherwise have been available.

Results from the qualitative analysis highlighted that most of the obstacles reported by farmers stemmed from mechanisms of support (e.g., insurance payments, governmental aid, etc.) that were not available for a significant period of time following Hurricane Maria, varying from several months up to a year. Farmers across all regions of Puerto Rico voiced the challenges they experienced when attempting to access governmental agencies, services, and materials and supplies needed to repair their farms and recover, physically and financially, from the effects of the hurricane. These were exacerbated by Puerto Rico experiencing the longest blackout in the modern history of the United States; communications were downed, and many regions of Puerto Rico only received restored power and water months after the hurricanes' landfall (Masters and Houser, 2017; Bonilla, 2020). Farmers also noted these obstacles as the third most common. Furthermore, our analysis suggests that farmers also faced challenges when accessing social networks of support, likely further challenged by the lack of material resources and technology. Perfecto et al. (2019) assessed how coffee farms' management, whether incorporating agroforestry or performing an intensive style, related to the degree of damages experienced and farm recovery after Maria. The study found that coffee farmers' recovery after Maria was potentiated by assistance from their social and community networks of support. And that management style may come secondary in a catastrophic context (Perfecto et al., 2019). Nevertheless, our qualitative analysis suggests that farmers may have been constrained in accessing such networks, which likely affected their overall capacity for recovery. Furthermore, the analysis reflected farmers disappointment on the state, and unmet expectations regarding aid in a catastrophic event.

The lack of institutional, economic, and social support likely affected the way that farmers perceived the practices necessary to adopt to overcome future challenges. We found that one of the top intended future practices was to acquire new agricultural insurance or improve current insurance. Agricultural insurance is one key risk management tool that farmers in the US use to offset climate-related impacts (Claassen et al., 2017; Reyes et al., 2020), but this insurance may not be aligned with needs of farmers, the impacts they face, or their farming systems. On the other hand, it could be that insurance dynamics (e.g., making payments, answering claims, etc.) might not be adequately equipped to deal with emergencies such as the one triggered by Hurricane Maria. Mainland US research has demonstrated regional differences in the role of insurance as a risk management tool (i.e., important in Midwest, less so in New England) (Mase et al., 2017; White et al., 2018). In Puerto Rico, agricultural insurance is mostly managed by the *Corporación de Seguros Agrícolas* (CSA) of the local Department of Agriculture. Obtaining payments from this insurance was specifically mentioned numerous times as an obstacle in our analysis. For example, one farmer noted, "I had about an acre of *yautía* insured, and the insurance paid after seven months. I could not recover any of the *yautía*." These results indicate that while insurance likely would have enabled increased capacity for farm recovery, it too faced many barriers, which prevented farmers from receiving the money from their insurance claims in a timely manner. This finding also suggests that in the aftermath of Maria, or in the context of disaster, where "normal" means of communication and accessing resources are disrupted, bureaucracy may not have the capacity to manage these challenges. Thus, there is an important need to improve agricultural insurance delivery in future disaster contexts, especially if more farmers intend to invest in these services.

Our study further suggests that broader structures, such as systems of governance, farmers' social networks, in relation to infrastructure, policy, and public health, play a significant role in farmers' adaptive capacity. Hurricane Maria, as a disaster, made evident that Puerto Rico's political and social characteristics must be taken into account when aiming to understand adaptive capacity.

We note several limitations of our study, all of which are important for future research. First, we did not ask farmers if they had insurance prior to Maria, so we do not know if their insurance adoption is new or additional. Nevertheless, the survey did ask farmers if they had insurance at the time (8 months after landfall), and 53% stated that they did. Most of them reported that their insurance was with the Corporation of Agricultural Insurance of the Puerto Rico Department of Agriculture. Much of this paper's qualitative data indicated that farmers had insurance at the time of Maria, and show the difficulties experienced in assessing the funds. Future research could look more deeply to the extent to which agricultural insurance in Puerto Rico relates to adaptive capacity outcomes. Second, we are assuming that reported actual adoption was indeed only adopted after Maria, and not just a continuation of practices prior to the hurricane. The table in which they reported actual adoption practices specified "currently in use," though we did not ask about pre-hurricane adoption. Nevertheless, the survey question asked about new practices for future adoption ("Which of these agricultural practices and management strategies, if any, might you adopt in the near future to adapt to future extreme events like Hurricane Maria?"). Thus, we assumed those reported practices were only actualized after Maria. Third, we did not include in the models the type of farming system (e.g., dairy, mixed, coffee, etc.), and instead use a proxy for diversity (number of products). This was done because many production systems in Puerto Rico are already diversified, making it challenging to assign farmers to a specific category. Furthermore, even within some categories (e.g., fruit/vegetable farmer), systems can vary significantly from annual to perennial. Lastly, we had an overrepresentation of bona fide farmers-—53% in our study, while 24% overall in Puerto Rico as reported by past Secretary of Agriculture in 2019, Carlos Flores (Díaz Rolón, 2019)—, despite most other demographics consistent with census data. This may be the result of selection bias through Cooperative Extension, which may have stronger connections with bona fide farmer networks.

CONCLUSION

This study assessed how various determinants of adaptive capacity reflect on Puerto Rican farmers' actual and intended adoption of adaptation practices, in light of the obstacles they faced toward recovery after 2017's Hurricane Maria. Our results suggest that, in many cases, broader structures, such as systems of governance, farmers' social networks, and infrastructure, affect adaptive capacity more than individual perceptions or capacity assets. We find that experiencing a total loss, appears to provide a window of opportunity for reinventing agricultural systems, as evidenced by the fact that farmers who faced a total loss adopted the most actual adaptation practices. Importantly, farmers with higher education were also more likely to adopt more adaptation practices, suggesting that capacity to change farming systems after a total loss is related to human capital. These results suggest that catastrophic events like Hurricane Maria, while devastating, do provide opportunities for change and resilience; but being able to take advantage of those opportunities is related not only to the human capital of an individual farmer, and their social networks, but the institutional and infrastructure capacities that are in place for recovery. Absent either, agricultural resilience may be challenging to achieve, or slow at best. Thus, working to improve both individual and structural factors that affect adaptive capacity are both likely to lend themselves toward greater adoption of adaptation practices, which would, in turn, improve resilience of farm systems under future shocks. Lastly, our study shows that a mixed-methods approach into understanding adaptive capacity provides nuanced information that might not be captured in quantitative model assessments alone. Future studies should further integrate qualitative and quantitative data to improve our understanding on the role of broader structural components in individual adaptive capacity outcomes.

DATA AVAILABILITY STATEMENT

The data sub-set used for this study can be available upon request to the corresponding author.

ETHICS STATEMENT

This study was approved on December 2017 by the Committees on Human Subjects Serving the University of Vermont and the UVM Medical Center at the Research Protections Office approved our study on 04-Dec- 2017. The study (CHRBSS: 18-0258) received Exemption Category 2. Consent was included in survey booklet, and was facilitated orally through enumerators. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

LR-C: wrote manuscript draft, conceptualized study, quantitative analysis, qualitative analysis, and project liaison. MM: reviewed and edited manuscript, conceptualized study, and qualitative analysis. MTN: reviewed and edited manuscript, conceptualized study, quantitative analysis, and funding allocation. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2021.662918/full#supplementary-material

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A Regional, Honey Bee-Centered Approach Is Needed to Incentivize Grower Adoption of Bee-Friendly Practices in the Almond Industry

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Managed and wild bee populations contribute over $15 billion in pollination services to US agriculture, yet both are declining or becoming increasingly vulnerable to parasites and disease. The loss of healthy and diverse forage is a key driver in bee declines, so incentivizing land managers to adopt diversified bee-friendly management practices such as forage plantings and reduced pesticide use can directly increase food security, pollinator health, and farmer adaptive capacity. To better understand what might incentivize growers to adopt bee-friendly practices, we conducted a survey of California almond growers, whose orchards are entirely dependent on bee pollination and draw nearly 88% of US bee colonies each February to pollinate almond bloom. We asked 329 respondents across all major almond growing regions of CA about their adoption rate and incentives for planting cover crops, pollinator habitat, and practicing the recommended and legally required bee-friendly best management practices, as well as their interest in bee-friendly certification programs. Using a model selection framework, we evaluated which geographic, social, operational, and pollination-service related factors were predictive of bee-friendly practice adoption. We found that no single factor was a statistically significant predictor of adoption across all models, suggesting there is no silver bullet determining bee-friendly practice adoption. However, we discovered that region and concerns about future pollination services consistently emerged as important factors related to all the practices we investigated, except the adoption of legally required BMPs. These findings suggest that a regionally flexible pollinator conservation strategy focused on supporting honey bee colonies might have the highest likelihood of grower participation and adoption.

Keywords: pollinators, adaptive capacity, diversified farming, almond (*Prunus dulcis* Mill.), honey bees (*Apis mellifera*), agriculture, apiculture, wild bees

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INTRODUCTION

Pollinators are an essential component of functioning and sustainable agricultural systems and play a central role in food security (Potts et al., 2010b). Managed and wild bees add an estimated $15 billion in pollination services to nearly 70% of all major food crops in the United States (Pollinator Health Task Force, 2016; Kulhanek et al., 2017). Despite their critical economic and ecological role, the current state of wild and managed bee populations is precarious. Between April 2019 and

2020, US beekeepers lost an estimated 43.7% of their colonies (Bruckner et al., 2020), and research suggests that estimated wild bee abundance declined 23% between 2008 and 2013 in the US (Koh et al., 2016), part of a pattern of widespread loss of pollinator diversity and abundance (Potts et al., 2010a, 2017). Research indicates that these losses are due to a nexus of stressors including parasites, pathogens and disease, pesticides, and the loss of the habitat and floral resources necessary for pollinator survival—all of which negatively impact bee health (Goulson et al., 2015). Though there is considerable scientific support for practices that promote wild and managed bees in agriculture (Winfree et al., 2007; Klein et al., 2012; Di Pasquale et al., 2013; Kremen and M'Gonigle, 2015; M'Gonigle et al., 2015; Evans et al., 2018; Kremen et al., 2018, 2019), less is known about what factors lead to farmer adoption of bee-friendly management practices.

A key strategy to support native and managed bees is through diversified farming practices such as planting and maintaining seasonal and permanent pollinator habitat (Winfree et al., 2007; Kremen et al., 2012, 2019). Strips of semi-natural vegetation in fields (e.g., flowering cover crops) or along margins (e.g., hedgerows, wildflower strips) can increase wild pollinator diversity and visitation (Klein et al., 2012; Kremen and M'Gonigle, 2015; M'Gonigle et al., 2015; Williams et al., 2015; Ponisio et al., 2016; Nicholson et al., 2017; Kratschmer et al., 2019; Kremen et al., 2019), support natural enemies for pest control (Landis et al., 2000; Gontijo et al., 2013; Holland, 2019), and provide a consistent supply of high quality floral resources that strengthen honey bee health (Huang, 2012; Di Pasquale et al., 2013, 2016). A diverse bee community can boost crop yield and yield stability (Garibaldi et al., 2011, 2013) and increase the pollination efficiency of honey bees (Greenleaf and Kremen, 2006; Brittain et al., 2013), which suggests growers could reduce the cost of honey bee importation while increasing yields if populations of wild bees were restored in agricultural areas. Within fields, temporary cover crops also have a number of non-pollinator friendly benefits, such as weed and nematode suppression and improvements in soil structure and water infiltration (Dabney et al., 2001; Marahatta et al., 2010; Crézé et al., 2018).

Given the current state of honey bee health and the decline of native pollinators, the adoption of bee-friendly practices on agricultural land can play a key role in stabilizing pollinator health and populations and also create greater on-farm adaptive capacity (Engle, 2011). Adaptive capacity refers to a farm operation's ability to prepare for stresses and changes in advance, or adjust and respond to the effects caused by those stresses (Smit et al., 2001; Engle, 2011; Petersen-Rockney et al., 2021), such as climate change or declines in biodiversity (e.g., bee populations). One approach to on-farm adaptive capacity is through diversification (Petersen-Rockney et al., 2021), such as through the addition of pollinator habitat and forage that could support wild bees so growers are not completely reliant on managed pollinators. While adopting pollinator-friendly practices is just one step toward becoming a diversified farming system, transformations toward diversified farming are likely to proceed incrementally (Petersen-Rockney et al., 2021).

Incentivizing Bee-Friendly Practice Adoption and Adaptive Capacity

Varied strategies incentivize farmers to adopt diversified bee-friendly practices. Farmers might plant cover crops and pollinator habitat with funding from government and private cost-share programs, such as the US Department of Agriculture's Environmental Quality Incentive Program (EQIP) and the non-profit Project Apis m.'s "Seeds for Bees" cost-share program for cover crop planting (Project Apism, 2020). To reduce pesticide use during bloom, pesticide-related bee-friendly management practices have been formulated and promoted by specialty crop groups, extension specialists, and regulatory agencies, such as the Honey Bee Best Management Practices (hereafter HB BMPs) formulated by California's Department of Pesticide Regulation (CDPR), the Almond Board of California and other stakeholders (CDPR, 2018a; Almond Board, 2019). These practices aim to provide a healthy environment for bees (primarily honey bees) during bloom, increase communication between different stakeholders, and reduce pesticide exposure to bee-toxic pesticides while managed bees are in. almond orchards.

Several organizations have developed bee-friendly certification programs requiring the installation of permanent pollinator habitat and cover crops, and the adoption of pesticide-use restrictions such as the HB BMPs. These programs, which include the Xerces Society and Oregon Tilth's Bee Better Certification (Xerces Society, 2020) and the Pollinator Partnership's Bee Friendly Farming certification (Pollinator Partnership, 2020), aim to create market-based demand for the adoption of pollinator beneficial practices. If enough of a market develops for bee-friendly crops, processors or distributers may incentivize their growers to adopt bee-friendly practices, so they can provide almonds for companies like Kind Snacks, which became the first snack company to commit to using only bee-friendly almonds in their snack products (PR Newswire, 2020).

Farmers are highly influenced by their social networks and tend to adopt new practices, such as perennial crops and vegetation (e.g., cover crops, pasture, riparian buffers, and restored wetlands), based on the advice of trusted peers and experts (Brodt et al., 2004a; Atwell et al., 2009). In-person communication from private and government conservation organizations can have a strong effect on the adoption of perennial vegetation (Atwell et al., 2009), in part by educating land managers about the varied programs available to help fund conservation efforts (Gaines-Day and Gratton, 2017). Pest control advisors (PCAs) also play an important role in shaping growers' pest management practices (Brodt et al., 2005), which can include the installation of forage and habitat to attract beneficial insects. Bee-reliant farmers interact with beekeepers and bee brokers during their crop's bloom period (Goodrich, 2017; Durant, 2019), who may influence growers to adopt bee-friendly practices as well. For example, social pressure to support bees may be strong for growers that are surrounded by year-round beekeepers like the honey bee queen breeders in California's Northern Central Valley (Schiff and Sheppard, 1996).

Beekeepers can also incentivize growers by providing discounts on honey bee colony pollination services, though this practice is not yet widespread (Durant, unpublished data). For example, almonds (*Prunus dulcis*) are a pollinator-dependent crop in California's Central Valley that blooms mid-February (Connell, 2000), earlier than most bee-dependent crops in the US. Since 2004, the average per-colony fees for almond pollination have risen by about 180%, from 7% of the total operating costs in 1998 to 15% in 2019 (Hendricks et al., 1998; Duncan et al., 2019; Goodrich, 2019). Other crops have also seen increases, but none so substantial as that experienced by the almond industry and other crops that overlap with the almond industry's bloom period, such as early-blooming cherries and plums in California (Ferrier et al., 2018; Goodrich, 2019). These fee increases are linked to growers' demand for stronger colonies, in other words, colonies with more frames of active honey bees (Goodrich, 2019). To meet this demand, beekeepers began taking measures to increase colony strength through increased disease monitoring, nutritional supplements, and storing colonies in winter warehouses, all of which has added to the cost of colony production (Durant, 2019; Goodrich, 2019). However, if growers have cover crops or pollinator habitat flowering during bloom or commit to reducing the use of bee-toxic pesticides while bees are in their orchards, some beekeepers may offer a discounted rate on colony rental fees because of the benefit these practices can provide their colonies.

Other incentives might be operationally or regionally determined. For example, installation costs of planting cover crops and permanent habitat can be high (Brodt et al., 2008; Cruz et al., 2013; Morandin et al., 2016), since growers need to pay for equipment, water, and labor expenses to establish as well as maintain cover crops or habitat. This may mean that larger operations might be better positioned to adopt bee-friendly practices. Growers with abundant winter and/or summer rainfall may be more incentivized to adopt cover crops and pollinator habitat as well. Depending on the rental arrangement, land tenure can be another factor affecting bee-friendly practice adoption. Research indicates that absentee landlords can block conservation projects (Brodt et al., 2008) and renters are less interested in long-term diversified conservation practices (Soule et al., 2000), so owner operators might be more incentivized to adopt bee-friendly management practices. Finally, growers may be incentivized to adopt bee-friendly practices to increase their ability to attract high quality beekeepers or out of a desire to support native pollinators (Hanes et al., 2015; Gaines-Day and Gratton, 2017; Park et al., 2020).

While existing research offers key insights to California growers' adoption of bee-friendly practices (Brodt et al., 2004a, 2005, 2006, 2008), much of this research took place before honey bee declines became a serious issue in 2006 with CCD (Underwood and VanEngelsdorp, 2007) and evidence about precipitous declines in wild populations emerged (Koh et al., 2016; Kopec and Burd, 2017). Honey bee and native bee declines are now a concern at federal, state, and county levels (The White House, 2014; CDPR, 2018b). As such, growers and commodity marketing boards may have a stronger impetus to support wild and managed pollinators to secure stable pollination for their crops. Also, while previous studies have largely focused on a specific region within a state, many crops are grown across state regions (e.g., California's North and South Central Valley), and thus a cross-regional analysis could help identify regional factors that shape adoption.

To investigate the factors leading to the interest in and adoption of bee-friendly practices, we conducted a survey of almond growers across the major almond producing regions in California. We focused on factors that might influence grower adoption and/or interest in adopting bee-friendly practices such as cover crops, permanent pollinator habitat, reducing pesticides through adopting the HB BMPs, as well as interest in bee-friendly certification programs. Using a model selection framework, we employed the survey data to identify regional, operational, social, and pollination-related factors that predicted grower adoption and interest in bee-friendly practices. Though survey-based studies have investigated the drivers for adopting practices to support native bees in fruit crops (Hanes et al., 2015; Gaines-Day and Gratton, 2017; Park et al., 2020), this study is the first to evaluate the adoption of multiple bee-friendly practices across multiple California regions. Additionally, our survey offered an incentive, which likely helped increase survey response (Ryu et al., 2006), particularly those who might be adopting fewer bee-friendly practices. The factors incentivizing grower adoption of bee-friendly practices play a critical but understudied role in determining successful pollinator conservation and restoration on agricultural lands (Brodt et al., 2004b). Results from this study can thus increase our understanding of factors that incentivize farm diversification and help inform pollination conservation strategies on agricultural lands, particularly for farmers who produce crops that rely on bee pollination.

MATERIALS AND METHODS

Survey Methods

To understand the factors affecting grower adoption of bee-friendly practices, we conducted an online survey of almond growers, both hired farm managers and owners/owner operators. We selected the almond industry for this study because almonds are one of the most bee-intensive crops in California. California's almond industry produces 80% of the world's almonds and was the state's second most valuable crop in 2019 (CDFA, 2020). Over the past two decades, the almond industry expanded from 595,000 acres to over 1.39 million acres (Tippett et al., 2001; CDFA, 2019), which has led to a corresponding demand for more managed bees. Currently, two colonies per acre are recommended by crop experts (USDA and FCIC, 2018), which means that around two million honey bee colonies are shipped to California each February to pollinate almonds—nearly 88% of all managed colonies in the United States (Goodrich and Durant, 2020). Given the high number of blooming flowers and managed honey bees pollinating almonds each spring, the management practices almond growers adopt can potentially have a large impact on wild and managed bees.

The survey ran between December 2019 through February 2020. We included questions that focused on five key areas: (1) information about the almond operation and the people who were influential in decision making regarding bee-related management practices; (2) adoption of cover crops and other pollinator beneficial habitat; (3) adoption of the HB BMPs; (4) interest in a bee-friendly certification programs; and (5) satisfaction with various aspects of their 2019 almond pollination experience as well as their concerns about future pollination services. For the remainder of the paper, we refer to the respondents as growers rather than farmers, as this is the term used by the almond industry.

To distribute the survey, we advertised the survey through the Almond Board at their annual Almond Board Conference. We then mailed postcards with a QR code and link to the online survey to over 3,248 growers in seven representative counties using addresses obtained from each county agricultural commissioner's pesticide permit data (public data). The counties were Butte, Colusa, and Glenn counties in the north; Stanislaus and Merced in North San Joaquin Valley; and Fresno and Kern county in South San Joaquin Valley. We also attended three Almond Board pollination workshops to promote the survey. Finally, several industry stakeholders sent emails to their members about the survey to help increase participation. All respondents were offered a $20 gift certificate incentive for completion of the survey, to increase response (Ryu et al., 2006).

TABLE 1 | Honey Bee Best Management Practices (HB BMPs) and their legal status.

Recommended	Cover water sources for pollinator bees before pesticide applications (or replace water after)
	Avoid applying pesticides during bloom with label cautions stating: "highly toxic to bees" or "toxic to bees"
	Avoid applying pesticides during bloom with label cautions stating "residual times" or "extended residual toxicity"
	Only apply fungicides in the late afternoon or evening, when bees are not present
	Avoided applying all insecticides (except B.t.) during bloom
	Avoided tank-mixing insecticides (except B.t.) with fungicides during bloom
Legally required	If labeled bee-toxic pesticides are applied, provide 48-h advance notice to all beekeepers within one-mile radius
	Ensure that bee colonies are never sprayed directly with any pesticides
	Read the pesticide label's protocols before applying any agrochemical for the first time

Growers' Self-Selected Incentives

For each bee management practice section (cover crops, permanent habitat, HB BMPs, and interest in certification), we asked growers to select different variables that might have incentivized or would potentially incentivize them to adopt the practice. To identify these incentives before administering the survey, we conducted informal interviews and piloted the questions with multiple stakeholders to determine the most likely incentives for almond growers, and then used those as options. In the survey, growers were asked to identify which incentives might encourage them to adopt a given bee-friendly practice; they could choose all options that applied.

In the survey, we defined cover crops as "a variety of species planted intentionally and temporarily between tree rows" and permanent pollinator habitat (hereafter pollinator habitat) as including "year-round herbaceous and/or woody plant species (e.g., hedgerows, perennial or re-seeding wildflower strips, riparian forests, filter strips) that are maintained along at least some of the edges of the orchard." Growers were asked if they had planted cover crops in the past 5 years or had pollinator habitat of any kind surrounding the almond acreage they farmed in 2019, and could respond *yes* or *no*. For the HB BMPs, we listed each one and included an informative link in the survey for more information about the practice. Growers were asked to select if they *sometimes, always,* or *never* adopted the recommended HB BMPs (Almond Board, 2020) or *made an effort*, *usually*, or *always* adopted the legally required HB BMPs (CDPR, 2018a) (see **Table 1** for full list of HB BMPs). Since we were collecting emails (i.e., identifying data), we did not have *never* as an option for the legally required HB BMPs. We defined bee certification as a voluntary bee-friendly certification program that would require growers to adopt "some level of bee-friendly management practices on farm to meet the standards such as: practicing most or all of the HB BMPs, planting annual cover crops, or planting and maintaining permanent pollinator habitat." Growers were asked if they were interested in a bee certification program and could respond *yes, no*, or *not sure*.

Finally, we asked growers about their level of concern about the following factors that may affect future almond pollination: the cost of bee colonies, declining bee health, lack of available bee colonies, lack of skilled beekeepers, and loss of native pollinators. Growers could respond with *not a concern*, *moderate concern*, or a *strong concern*.

Quality Criteria for Cleaning Data

The survey received 447 responses in total. To prepare for analysis, we cleaned the data according to the following quality criteria. We first removed any incomplete or notably inaccurate responses, such as growers who responded that they managed over 40,000 acres of almonds (more than the largest operation in California). We also deleted any responses completed under 2.5 min, and those using the same IP address because of concerns about duplication (particularly since we offered compensation). Lastly, if respondents selected "no" and "prefer not to answer" for most of the questions or if they only marked the first answer choice in each question, the entire response was flagged, reviewed, and then deleted if the result was determined unreliable. After this data cleaning, we had a total of 329 responses for analysis.

Factors Affecting Growers' Implementation of Bee Friendly Practices

To determine which factors influenced grower adoption of bee-friendly practices, we used the following three factors as binary response variables in generalized linear models (GLMs, binominal error): (1) whether a grower reported that they had planted cover crops, (2) whether they had planted pollinator habitat, and (3) whether they were interested in participating in a certification program. Because certifications are relatively new or in the process of being established, we only modeled interest, and not participation in, certification. In our analysis, we also explored the factors that determined whether growers adopted: (1) all six recommended HB BMPs and (2) all three legally obligated HB BMPs (**Table 1**). Growers had to have responded "always" to all obligated or recommended HB BMP criteria to be considered as having adopted this practice. After selecting our five adoption variables, we selected explanatory variables that matched our hypotheses about the regional, operational, social, and bee-related concerns that may influence grower adoption and interest in bee-friendly practices. We detail each of these below.

Region

We hypothesized that region would play a strong role in the adoption of bee-friendly practices and interest in certification. Almond growers operate in distinct geographic regions in California's Central Valley, influenced by different rainfall patterns, seasonal temperatures, water districts, water rights, and social communities. The five highest almond-producing counties are in the central and southern San Joaquin Valley (Kern, Stanislaus, Fresno, Madera, and Merced) (USDA NASS, 2019), and rely heavily on out-of-state beekeepers (Goodrich, 2017). Sacramento Valley, however, is where a large portion of the nation's honey bee queens are reared (Schiff and Sheppard, 1996; Cobey et al., 2016), so almond growers are immersed in a strong community of involved beekeepers who might influence growers' adoption of bee-friendly management practices.

Another key difference between regions is annual rainfall. Growing regions in the San Joaquin Basin receive much lower rainfall (∼5–15 inches) than counties in the Sacramento Valley which can receive 15–25 inches a year (National Weather Service, 2020). Water costs also vary greatly between Sacramento Valley and North and South San Joaquin Valley. For example, the cost per cubic foot of surface water (CCF) in Sacramento Valley was $1.76 in 2020 (in the Chico-Hamilton Tariff Area), while in North San Joaquin Valley (Stockton Tarriff Area) it was $3.42 per CCF. In South San Joaquin Valley, the cost was $13.5 per CCF in the Kern River Valley, nearly seven times as expensive as Sacramento Valley (California Water Service Company, 2020). Groundwater, on the other hand, was largely unregulated until the passage of the Sustainable Groundwater Management Act (SGMA) in 2014 (Chappelle et al., 2017). SGMA's implementation may mean that groundwater in San Joaquin Valley, which has high rates of new well installation and high levels of groundwater overdraft (Krieger, 2014; Hanak et al., 2015), will be more expensive and less available than in the past.

To better understand the role that region plays in the adoption of bee-friendly practices, we assigned each county to one of the following regions: Sacramento Valley, North San Joaquin Valley, and South San Joaquin Valley (**Appendix Table A**). We used USGS water basin designations to guide which counties went into which regions (**Appendix Table A**). We assigned counties in the Sacramento Valley Basin to Sacramento Valley, counties in the San Joaquin Basin to North San Joaquin Valley, and counties in the Tulare Basin to South San Joaquin Valley (USGS, 2021a,b,c). For countries in the center of the San Joaquin Valley (e.g., Madera) we assigned them to the North or South based on their primary watershed affiliation. We validated that our results did not change qualitatively based on the north vs. south assignment of these counties. Given the proximity to year-round beekeepers, higher rainfall, and less expensive water than in the North and South San Joaquin Valley, we hypothesized that growers in Sacramento Valley would be more likely to adopt cover crops and pollinator habitat than growers in North and South San Joaquin Valley as a result.

Operational Characteristics

Given the potential expenses associated with reducing pesticide use and planting cover crops and permanent pollinator habitat (Brodt et al., 2008; Cruz et al., 2013; Morandin et al., 2016), we hypothesized that larger operations (i.e., those who manage more acres) would be more likely to adopt these practices. Larger operations that are also processors and distributers (referred to as "handlers") often market their products directly to consumers, while small and mid-sized farmers deliver their almonds to third-party handlers after harvest (Durant, 2019). Thus, larger operations could be more interested in the potential for an increased price point from a bee-friendly certification as well. Land tenure can also affect bee-friendly practice adoption (Brodt et al., 2008), so we hypothesized that growers who owned the majority of the land they farm on would be more likely to adopt bee-friendly practices and more interested in certification. Lastly, because we hypothesized that growers who have already adopted cover crops and installed permanent habitat would be more likely to express interest in certification programs, we also included those variables as explanatory in the certification interest model.

Social

To determine which actors on almond operations were influential in determining bee-friendly practice adoption, we included the following actors that growers identified in the survey as either "influential" or "not influential" in determining pollinator management practices: pesticide control advisors (PCAs), beekeepers, and bee brokers (growers, beekeepers, and full-time bee brokers who connect almond growers with beekeepers and colonies). In the almond industry, over 97% of growers rely on PCAs (Brodt et al., 2005), and most almond growers hire beekeepers or bee brokers to meet their pollination needs. We hypothesized that growers who stated that a beekeeper or bee broker played an influential role in pollinator decisions would be more likely to adopt bee-friendly practices and would be more interested in participation in a certification program. We hypothesized that those with a PCA might have lower rates of HB BMP adoption.

Pollination-Related Concerns

We also examined the effect of factors related to growers' concerns about the future of almond pollination services. Given that growers are concerned about the price and strength of their bee colonies, particularly because of the 2020 dip in almond prices (Goodrich and Durant, 2020), we wanted to use growers' satisfaction with the strength and price of bee colonies in 2019 as variables. In the survey, growers were asked to rate their satisfaction with the price and strength of their colonies, and we used these rankings as variables. We hypothesized that growers who were satisfied with the price and strength of their colonies would be more likely to adopt bee-friendly practices and be interested in certification. Lastly, we examined growers' concerns about future pollination, including the cost of bee colonies, the lack of availability of future bee colonies, declining bee health, a potential lack of skilled beekeepers, and the loss of native pollinators. We hypothesized that growers who expressed strong concern about the cost of rented bee colonies would be less likely to adopt bee-friendly practices, given that adopting some practices may require extra labor and material expenses. We also hypothesized that growers who expressed strong concern about the rest of the concerns would be more likely to adopt bee-friendly practices and consider certification.

Model Selection

We then tested our hypotheses on the adoption and interest in bee-friendly practices and certifications data using a model selection framework (Johnson and Omland, 2004). We performed multi-model inference based on the corrected Akaike's Information Criterion (AICc) using the dredge function in the MuMIn R package (Burnham and Anderson, 2002; Johnson and Omland, 2004; Bartoń, 2020). Because model selection can be biased by collinearity (Cade, 2015), we used variance inflation factors (VIF) (Fox and Weisberg, 2019) to identify colinear variables and exclude them from being included together in a candidate model. We found that two bee concern-related variables (lack of skilled beekeepers and lack of available colonies) and two bee satisfaction-related-variables (honey bee colony and strength satisfaction) were colinear (VIF > 2) (Zuur et al., 2010). We therefore specified that these variable pairs could not be included in the same candidate model before running the model selection procedure. The model including all explanatory variables was fit using the glm function (logit link function). All the explanatory variables were categorical except acreage, which was standardized by subtracting the mean and dividing by the SD of the data. We selected the top model set as the models within 2 AICc of the minimum AIC. Using the top model set, we then computed the conditional model average (Bartoń, 2020). We used standard model assessment techniques to determine whether the top model met all the assumptions of a GLM (Zuur et al., 2009). All analyses were conducted in R v 4.0.0 (R Core Team, 2018).

RESULTS

In this section, we report on the results of the raw survey data, followed by the model selection analysis. The survey data are reported in the following order: demographics and information about 2019 pollination, adoption rates of the bee-friendly practices (the dependent variables in our model selection analysis), and finally, the results of our operational details and bee satisfaction/concerns (the independent variables in our model selection analysis).

Demographic and 2019 Pollination Season Details

The 329 responses to our survey represented a total of ∼212,000 almond acres (**Table 2**). Most respondents were male (84%) and fell within the 25–34 and 55–64-year-old age ranges, though our distribution was fairly representative across growers from 25 to 74 years of age. The majority of respondents were owner/operators of their almond orchard (56%) and the largest response was from operations that managed between 1 and 49 acres (40%). For regional representation, comparing our data to that of the 2017 Agricultural Census indicates that our data is representational of growers from each region (**Appendix Table A**), and that our results report on practices that apply to ∼17% of the total almond acreage in the Central Valley (**Appendix Table A**). Comparing our respondents' acreage ranges to the 2017 Agricultural Census data, our results slightly overrepresent larger operations (250+ acres) by ∼12% and underrepresent smaller operations (1–49 acres) by ∼13%, while our representation of mid-size growers (50–249 acres) is consistent with census estimates of the proportion of growers managing that acreage range (**Appendix Table B**).

We asked several questions about the 2019 pollination period (February through March). Most respondents (72%) rented all their bee colonies, while around 20% supplied some or all their own bee pollination, and 5.5% of respondents had some portion of their orchards that were not mature enough for pollination at the time of the survey (**Table 2**). Of those who rented, the majority rented directly from a beekeeper (64%), while around 24% used a bee broker, and 4% relied on another grower to broker their colonies. About 25% of respondents obtained bee colonies from either their county or a neighboring county, 22.2% obtained colonies from another county in California, and another 41% were obtained from out of state. When analyzed by region, 42% percent of respondents in Sacramento Valley obtained their colonies from either the same county or a neighboring county, compared to 22% of growers in the North and South San Joaquin Valley regions who obtained their colonies from a nearby location.

Adoption of Bee-Friendly Practices

Our survey results indicated that growers are more interested in growing cover crops than pollinator habitat (**Table 3**). Thirty-five percent of respondents said they had grown a cover crop in the last 5 years, and an additional 16% said they were interested in growing a cover crop in the future, bringing the total number of survey respondents that were either growing or interested in growing cover crops to 51%. Growers had less interest in adopting permanent pollinator habitat. Nineteen percent of growers said they already maintained permanent pollinator habitat in 2019, and the same number were interested in potentially adding pollinator habitat in the future, bringing the total number of growers interested in or already maintaining pollinator habitat to 38% of respondents. In general, growers were more satisfied with cover crops than with pollinator habitat (**Figure 1**), with an equal number somewhat or very satisfied with cover crops (46% for each), while most respondents were

TABLE 2 | Demographic and pollination operation details, listed by percent of total survey respondents.

DEMOGRAPHIC INFORMATION	
Gender	**%**
Male	84
Female	12
Other	1
Prefer not to answer	2
Age range	**%**
18–24	2
25–34	23
35–44	19
45–54	19
55–64	22
65–74	11
75+	3
Role on orchard	**%**
Owner	15
Owner/operator	56
Farm manager	29
Acreage range	**%**
1–49	40
50–99	17
100–249	16
250–999	15
1000+	12
Total acres: 212,416 (bearing and non-bearing)	17
POLLINATION INFORMATION	
How obtained pollination services	**%**
Rented all bee colonies	72.3
Rented some colonies supplied some	11.3
Supplied all bee pollination	8.5
Orchards not mature enough for pollination	5.5
Prefer not to answer	2.4
Who supplied bee colonies	**%**
Beekeeper	64.4
Bee broker	24.3
Another grower	4.0
Other	0.3
Prefer not to answer	2.1
Not applicable	4.9
Colony origin	**%**
Near orchard (same county or neighboring)	25.5
From California (not a neighboring county)	22.2
Out of state	40.7
Prefer not to answer	4.0
Not applicable	7.6
OPERATIONS THAT RENTED COLONIES FROM NEAR THE ORCHARD (% IN THAT REGION)	
Sacramento Valley	42.3
North San Joaquin	22.2
South San Joaquin	22.7

"Operations that rented colonies from near the orchard" is listed by percent of total respondents in that region. The "Total acreage" percentage in the Acreage Range section is the percent of all almond acreage (bearing and non-bearing) in California in 2017.

TABLE 3 | Adoption of bee friendly practices, listed by percent of total survey respondents.

Adoption of cover crops	**%**
Already growing CCs	35
Interested in growing	16
Not Sure	28
Not interested	21
Adoption of HB BMPs	**%**
Always all recommended	29
Always all legal	60
Adoption of pollinator habitat	**%**
Grew pollinator habitat in 2019	19
Interested	19
Not sure	29
Not interested	32
Interest in bee certification	**%**
Strong interest	27
Moderate interest	47
No interest	21
Prefer not to answer	4

somewhat satisfied with pollinator habitat (64%), and 25% were very satisfied.

The data also indicated a low to moderate rate of consistent HB BMP adoption, with 60% of growers always practicing all the three legally required HB BMPs, and 29% always practicing all of the six recommended HB BMPs. Lastly, most growers were interested in participating in a bee certification, with 47% expressing moderate interest and 27% a strong interest (**Table 3**).

Operational Details, Colony Satisfaction, and Pollination Concerns

Eighty-five percent of survey respondents managed orchards that were majority owned, i.e., their operation owned $\geq$ 50% of the acreage they managed in 2019 (**Table 4**). The median total acreage that respondents managed (yielding and non-yielding acreage) was 65 acres. We had a higher response rate from North and South San Joaquin Valley regions (57 and 26%, respectively) than Sacramento Valley (16%), which corresponds with acreage grown in California (USDA-NASS, 2019; **Appendix Table A**). The choropleth map (**Figure 2**) demonstrates our response rate by county. When asked which individuals were influential in pollinator management decision making (**Table 4**), forty percent of respondents selected that their PCA was influential, followed by their beekeeper (32%); while about 8% selected their bee broker (the other options were the owner and the hired manager, which we excluded from analysis). The majority of operations had not planted any cover crops or pollinator habitat (56%), though a sizable number had (44%).

A portion of the survey asked about growers' pollination practices in 2019 and general concerns about future pollination (**Table 4**). When considering price satisfaction, most growers (40%) felt their HBC price was fair, though 34% thought it

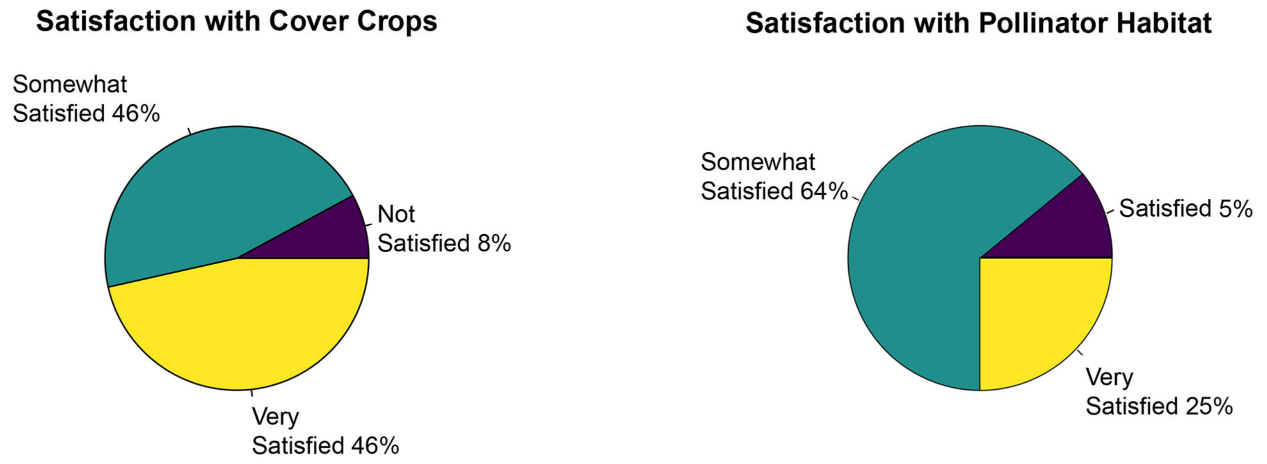

FIGURE 1 | Satisfaction with cover crops (left) vs. pollinator habitat (right).

was too expensive. The majority of growers were also either very satisfied (46%) or somewhat satisfied (34%) with honey bee colony strength, and only a small fraction of growers were unsatisfied (2%). When considering future almond pollination, most growers expressed strong concern about all the variables except the loss of native pollinators (**Table 4**). The greatest concern was the future cost of bee colonies, and most felt it was a strong concern (67%). Declining bee health and the lack of future bee colonies were also strong concerns, with 63 and 58% strongly concerned about bee health and the lack of colonies respectively. A future lack of skilled beekeepers and loss of native pollinators were less concerning, with most growers strongly concerned about skilled beekeepers (47%) and moderately concerned about the loss of native pollinators (43%). Around 15% of growers were not concerned about beekeepers or native pollinators.

Growers' Self-Selected Incentives

Figure 3 highlights the incentives to bee-friendly practices and interest in a bee certification program that respondents selected in the survey. Notably, across every single bee-friendly practice, increasing the strength of bee colonies during bloom was the number one incentive. For cover crops, the second top incentive was the non-pollination associated benefits from having cover crops, such as nitrogen fixing and water sequestration. Access to equipment and decreased rental fees as well as private and federal cost-share programs were all mid-level concerns, while decreased rental fees was the second-highest incentive for pollinator habitat, followed by federal cost-share programs and supporting native pollinators. Fourteen percent of growers said there were no incentives that would encourage them to plant cover crops, while 21% stated that there were no incentives that would encourage them to plant permanent pollinator habitat.

For the HB BMP incentives, after increasing the strength of colonies, growers seemed most incentivized by a decreased rental fee, followed by the ability to attract high-quality beekeepers, and a "handler" (processor or distributer) request to implement the HB BMPs was a less influential incentive. Nearly half of all respondents responded that they were already practicing most or all the HB BMPs. Finally, growers' second highest incentive for participating in a bee certification program was a decreased rental fee for managed bee colonies. Mid-range incentives included potential price premiums from the certification, cost-share programs to support associated costs with adopting bee-friendly practices, and the ability to attract high-quality beekeepers. A third of the respondents were incentivized by supporting native pollinators and the ability to better market their product through a bee-friendly label.

Factors Affecting Bee-Friendly Practice Adoption

In this section we review the results of the model selection analysis, which determined the variables that played an important role in shaping growers' adoption of bee-friendly practices. **Table 5** provides an overview of the statistically significant variables and those with *P*-values < 0.10. We also report differences in the probability of adoption by taking the exponent of the logit coefficient estimates that represent differences between levels of the explanatory variables.

Growers Who Grew Cover Crops in Past 5 Years

There were three statistically significant variables that influenced cover crop adoption: region, concern about the future cost of bee colonies, and potential lack of available colonies (**Figure 4**, **Table 5**, **Appendix Table C**). Respondents in the Sacramento Valley were statistically significantly more likely (by an average of ∼41%) to have grown cover crops than those in North and South San Joaquin Valley. Regarding cost of bee colonies, if a respondent was concerned about the future price of bee colonies, they were statistically significantly less likely to adopt cover crops than those who expressed no concern by an average of ∼20% for those with a strong concern, and ∼17% for those with a moderate concern. Respondents concerned about a future lack of available colonies were statistically significantly more likely

TABLE 4 | Operational details, colony satisfaction, and pollination concern variables, listed by percent of total survey respondents.

OPERATION INFORMATION	
Majority own	**%**
Yes	85.4
No	14.6
Total acreage	**%**
Median acreage	65.0
Maximum acreage	38,000.0
Minimum acreage	3.0
Region name	**%**
Sacramento Valley	15.8
North San Joaquin	57.5
South San Joaquin	26.8
Influential people	**%**
Pesticide Control Advisor	39.5
Beekeeper	31.6
Bee Broker	7.9
Planted Forage	**%**
Planted any forage	44.0
Have not planted forage	56.0
COLONY SATISFACTION	
HBC price satisfaction	**%**
Inexpensive	3.3
A fair price	40.4
Too expensive	34.0
Prefer not to answer	22.2
HBC Strength Satisfaction	**%**
Unsatisfied	2.1
Somewhat satisfied	34.0
Very satisfied	45.6
Prefer not to answer	18.2
POLLINATION CONCERNS	
Bee colony cost	**%**
Not a concern	5.8
Moderate concern	26.8
Strong concern	67.5
Declining bee health	**%**
Not a concern	4.6
Moderate concern	31.6
Strong concern	63.8
Lack of available colonies	**%**
Not a concern	8.2
Moderate concern	33.7
Strong concern	58.1
Lack of skilled beekeepers	**%**
Not a concern	15.8
Moderate concern	37.4
Strong concern	46.8
Loss of native pollinators	**%**
Not a concern	14.3
Moderate Concern	43.5
Strong concern	42.3

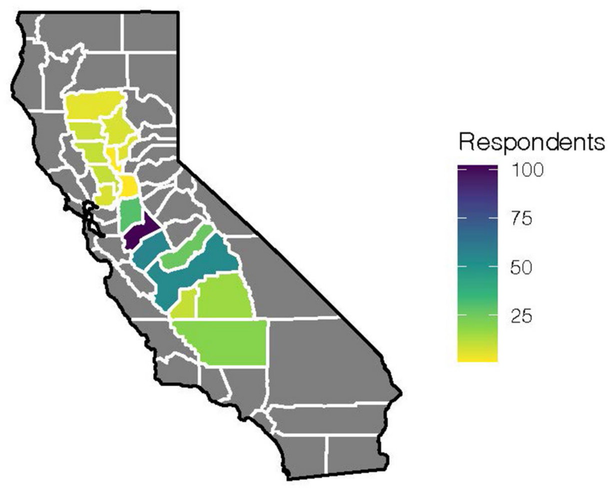

FIGURE 2 | Map of response rate by California county.

to adopt cover crops than those who expressed no concern, by 38% for those with a strong concern, and 34% for those with a moderate concern.

Several non-significant variables were present in the top averaged model set (**Figure 4**, **Appendix Table C**), including total acreage (slightly more likely to adopt with higher acreage), PCAs and beekeepers who were influential in making pollinator management decisions (slightly less likely with a PCA and more likely with a beekeeper), concern about the loss of native pollinators (slightly more likely if a strong concern than not a concern, and less likely if a moderate concern than not a concern), and whether growers owned a majority of the acreage they farmed (slightly less likely to adopt cover crops if they did).

Growers Who Grew Pollinator Habitat in 2019

There were three statistically significant variables that influenced pollinator habitat adoption: satisfaction with bee colony strength, region, and acreage (**Figure 5**, **Table 5**, **Appendix Table D**). If growers were very satisfied with the strength of their colonies, they were statistically significantly less likely to grow pollinator habitat than those who were unsatisfied by 5%. Growers farming more acres were more likely to adopt pollinator habitat than those with less acres by 3%. If growers were in Sacramento Valley, they were statistically significantly more likely to have been growing permanent pollinator habitat in 2019 than respondents in South San Joaquin Valley by 13%, and ∼7.5% more likely than those in North San Joaquin Valley.

Several non-significant variables were present in the top averaged model set (**Figure 5**, **Appendix Table D**), including the cost of honey bee colonies (more likely with a moderate or strong concern), having a bee broker or PCA who was influential in pollinator management decisions (more likely with a bee broker and PCA), loss of native pollinators (more likely if a moderate or strong concern), declining bee health (more likely

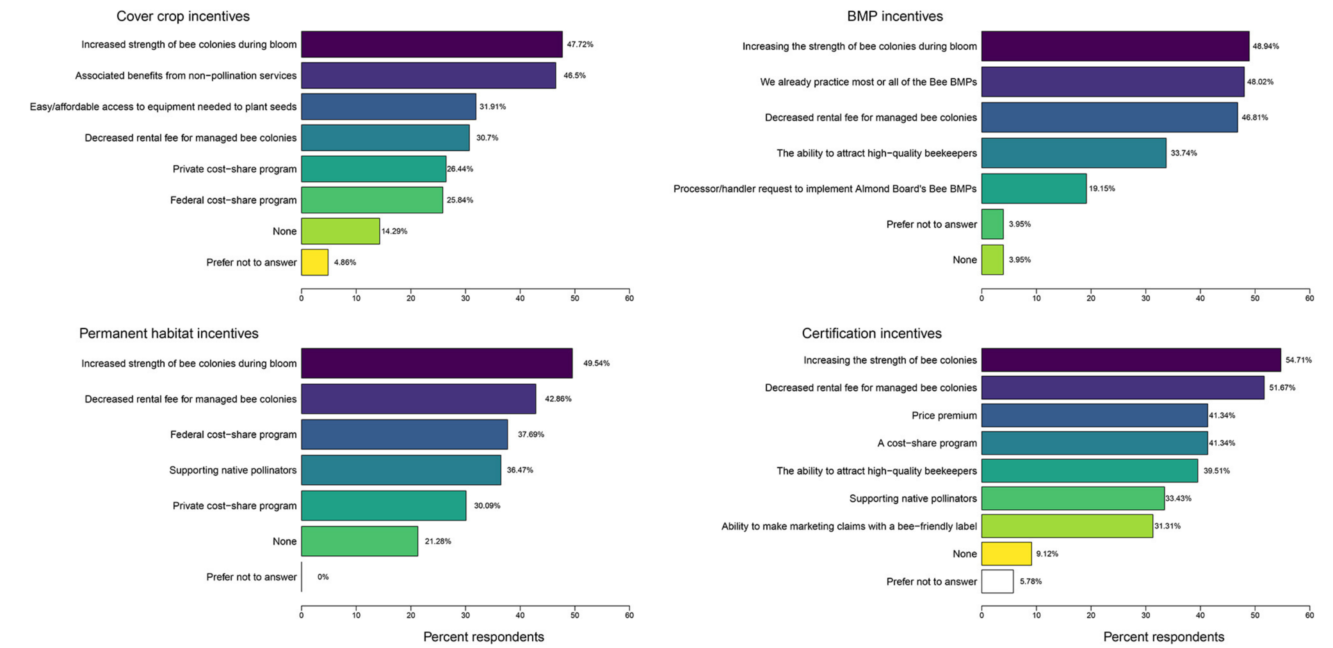

FIGURE 3 | Respondents' self-selected incentives to adopt bee-friendly practices, listed by percent of total survey respondents.

if a moderate or strong concern), or if the operation owned the majority of the acres they managed (slightly more likely if owned).

Grower Adoption of Recommended HB BMPs

Two statistically significant variables shaped growers' likelihood of always following the recommended HB BMPs: concern about declining bee health and region (**Figure 6**, **Table 5**, **Appendix Table E**). Growers who were moderately concerned about declining bee health were statistically significantly less likely (by 39%) to always adopt the recommended HB BMPs than those who thought it was not a concern. Respondents in Sacramento Valley were also statistically significantly less likely to adopt all the recommended HB BMPs than those in South San Joaquin Valley by 27%, and those in North San Joaquin were less likely to adopt than South San Joaquin Valley respondents by ∼12%, but these results were not statistically significant.

Several non-significant variables were present in the top averaged model set (**Figure 6**, **Appendix Table E**), including total acreage (slightly less likely with higher acreage), and whether they had a beekeeper or PCA who was influential in pollinator management decisions (slightly more likely if did).

Grower Adoption of Legally Obligated HB BMPs

Three statistically significant variables influenced growers' likelihood of always following the legally required HB BMPs: satisfaction with colony strength and having a PCA or bee broker who is influential in making pollinator decisions (**Figure 7**, **Table 5**, **Appendix Table F**). Growers who were satisfied with the strength of their colonies were statistically significantly more likely to adopt all the legally required HB BMPs, by 27% for those who were very satisfied and 19.5% for those who were somewhat satisfied. Respondents who listed a PCA or bee broker as influential in pollinator management decisions were statistically significantly less likely to adopt all the legally required HB BMPs than those who did not, by about 15% for those with a PCA, and 21.5% by those with a bee broker.

Several non-significant variables were present in the top averaged model set (**Figure 7**, **Appendix Table F**), including region (more likely to always adopt if in North San Joaquin Valley or Sacramento Valley than in Southern San Joaquin Valley), total acreage (slightly less likely with higher acreage), and concerns about lack of skilled beekeepers and lack of available colonies (slightly less likely with a moderate or strong concern).

Interest in Participating in a Bee Certification Program

We had three statistically significant variables influencing growers' interest in participating in a bee-friendly certification program: whether they planted any cover crops or pollinator habitat, satisfaction with the price of bee colonies, and region (**Figure 8**, **Table 5**, **Appendix Table G**). If respondents had planted cover crops or pollinator habitat, they were statistically significantly more likely by 17% to want to participate in a bee certification program than those who had not. If growers thought their colonies were inexpensive in 2019, they were about 41% less likely to want to participate in a certification program than if they thought they were too expensive. In other words, growers who thought their colonies were expensive were most likely to want to participate in a bee certification program. Finally, if growers were in Sacramento Valley, they were about 24% more likely to be

TABLE 5 | For each model of bee friendly practice adoption, the explanatory variables that are both included in the top model set and statistically significant ($P < 0.05$) are highlighted in gray.

	Majority own	Acreage	Region	Influential PCA	Influential bee broker	Influential beekeeper	HBC price satisfaction	HBC strength satisfaction	Cost bee colonies	Lack available colonies	Declining bee health a concern	Loss native pollinators	Planted forage
Cover crops grown in past 5 years			More likely if in Sac Valley than South SJV ($P = 0.000$)						Less likely if a strong concern ($P = 0.004$) or moderate concern ($P = 0.043$)	More likely if a strong concern ($P = 0.01$) or a moderate concern ($P = 0.025$)			N/A
Pollinator habitat grown in 2019		More likely with higher acreage ($P = 0.043$)	More likely in Sac Valley than South SJV ($P = 0.027$)		More likely if selected ($P = 0.065$)			More likely if very satisfied ($P = 0.012$)				More likely if a strong concern than not a concern ($P = 0.067$)	N/A
Always practiced all recommended BMPs			Less likely in Sac Valley than South SJV ($P = 0.013$) Less likely in North SJV than South SJV ($P = 0.099$)								Less likely if a moderate concern than if not concerned ($P = 0.005$)		N/A
Always practiced all legally required BMPs				Less likely if selected ($P = 0.011$)	Less likely if selected ($P = 0.033$)			More likely if very satisfied ($P = 0.000$) or somewhat satisfied ($P = 0.014$) than if not satisfied					N/A
Interested in bee certification		More likely with higher acreage ($P = 0.079$)	More likely in Sac Valley than in South SJV ($P = 0.044$)				Less likely if they found it inexpensive than if they found it too expensive ($P = 0.014$)			More likely if a strong concern than not a concern ($P = 0.074$)			More likely if they planted cover crops or pollinator habitat ($P = 0.014$)

SJV = San Joaquin Valley.

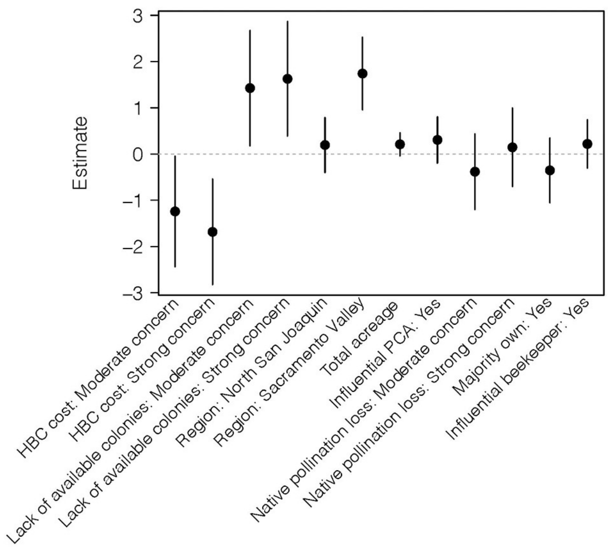

FIGURE 4 | Cover Crops: Coefficient estimates (points) and 95% confidence intervals for explanatory variables included in the top model set for cover crop (CC) adoption. Coefficients are on a logit scale. Because most of the variables are categorical, the coefficients estimate the difference between each category and the intercept. The intercept is set as "No concern" for bee concern variables, "Not influential" for each influential person, and South San Joaquin for region. Estimates above zero represent an increase in the probability of cover crop adoption relative to the intercept, and below zero represent a decrease in cover crop adoption relative to the intercept. Total acreage is the only continuous variable, and therefore the coefficient is the estimate of the slope of cover crop adoption for every unit increase in operation size (log).

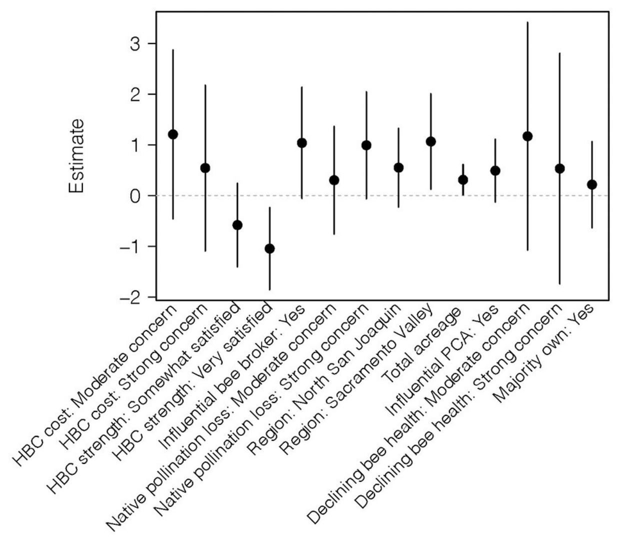

FIGURE 5 | Permanent pollinator habitat: Coefficient estimates (points) and 95% confidence intervals for explanatory variables included in the top model set for permanent pollinator habitat adoption. Coefficients are on a logit scale. The intercept is set as "No concern" for bee concern variables, "Unsatisfied/Prefer not to answer" for honey bee colony (HBC) strength satisfaction, "Not influential" for each influential person, and South San Joaquin for region.

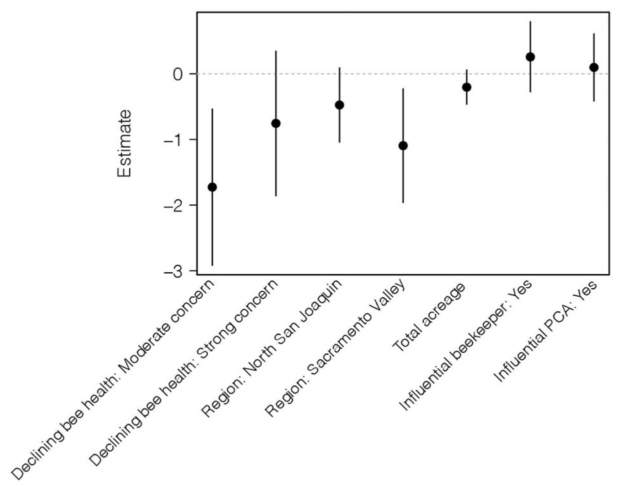

FIGURE 6 | Recommended Honey bee BMPs: Coefficient estimates (points) and 95% confidence intervals for explanatory variables included in the top model set for Almond Board's recommended Best Management Practice (BMP) adoption. Coefficients are on a logit scale. The intercept is set as "No concern" for bee concern variables, "Not influential" for each influential person, and South San Joaquin for region.

interested in a bee certification program than growers in South San Joaquin Valley, and about 31% more likely than growers in North San Joaquin Valley.

Several other important variables in the model were important but not statistically significant (**Figure 8**, **Appendix Table G**), including whether respondents had a bee broker (more likely if selected) or PCA influencing pollinator decisions (slightly more likely if selected), total acreage (slightly higher with higher acreage), and native pollination loss (more likely if a strong or moderate concern).

DISCUSSION

Adjusting pesticide use and planting pollinator forage and habitat are important practices that can support bee populations and mitigate honey bee vulnerability (Brittain et al., 2012; Huang, 2012; M'Gonigle et al., 2015; Di Pasquale et al., 2016; Kremen et al., 2019), and understanding which incentives motivate growers to adopt these practices may help increase their rate of adoption. Our survey data indicated that across every bee-friendly practice, growers' primary self-selected incentive was to strengthen their honey bee colonies, followed by decreasing the rental fee for managed bee colonies. This underscores the major role that pollination concerns and expenses play in incentivizing the adoption of bee-friendly practices. Our data also indicate that California almond growers are more interested in planting or have already planted cover crops than permanent pollinator habitat types. Their primary incentives to plant cover crops were to strengthen honey bee colonies or take advantage of the non-pollination benefits of cover crops such as water retention,

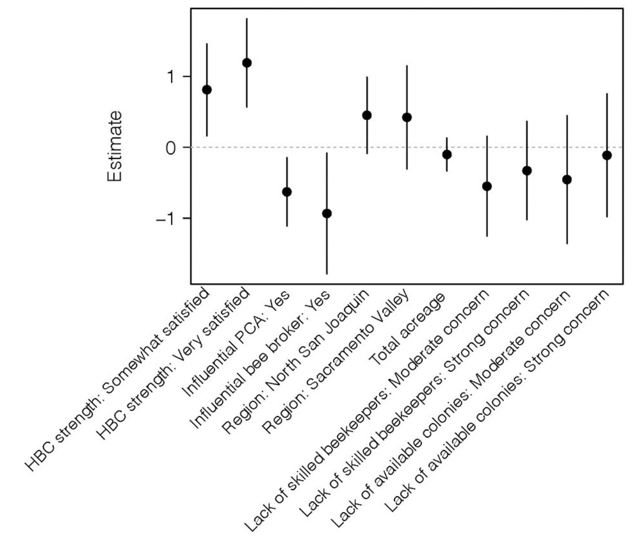

FIGURE 7 | Legally obligated Honey bee BMPs: Coefficient estimates (points) and 95% confidence intervals for explanatory variables included in the top model set for legally obligated BMP adoption. Coefficients are on a logit scale. The intercept is set as "No concern" for bee concern variables, "Unsatisfied/Prefer not to answer" for honey bee colony (HBC) strength satisfaction, "Not influential" for each influential person, and South San Joaquin for region.

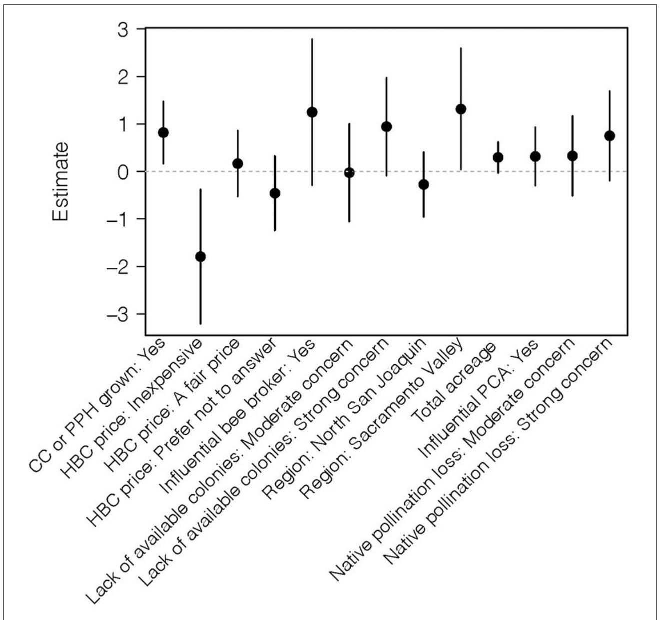

FIGURE 8 | Interest in a Bee-Friendly Certification: Coefficient estimates (points) and 95% confidence intervals for explanatory variables included in the top model set for bee-friendly certification interest. Coefficients are on a logit scale. The intercept is set as "No concern" for bee concern variables, "Too expensive" for honey bee colony (HBC) price satisfaction, "Not influential" for each influential person, and South San Joaquin for region.

pest control, weed and nematode suppression, and nitrogen fixing (Dabney et al., 2001; Marahatta et al., 2010; Crézé et al., 2018). Cover crops may also be more popular because they are perceived to require less water, attention, installation and labor and maintenance costs than permanent pollinator habitat (Brodt et al., 2008; Morandin et al., 2016).

Regarding pollinator habitat, growers were largely motivated to adopt pollinator habitat because of honey bees rather than native bees. Federal cost share programs were the third-highest motivation, suggesting that further outreach could be conducted to educate growers about the federal programs they can take advantage of to adopt pollinator habitat. As for bee-friendly certification programs, native bees were also a lesser incentive for participation, while price premiums, cost-share programs, and the ability to attract high-quality beekeepers were mid-level incentives after increasing colony strength and receiving a colony price reduction. This suggests that the beekeeping industry might be best positioned to encourage pollinator habitat to strengthen their colonies.

We further found that, in addition to pollination concerns, region played a consistent and statistically significant role in shaping growers' adoption of bee-friendly practices across all but one practice (the legally required HB BMPs). Growers in Sacramento Valley were more likely to have planted cover crops and pollinator habitat than those in the North and South San Joaquin Valley regions, confirming our hypothesis. Sacramento Valley growers were also statistically significantly more interested in a bee certification program. We suggest that the higher adoption and interest in certification (which necessitates forage/habitat installation) in the Sacramento Valley may be due to a combination of higher winter rainfall (National Weather Service, 2020), less expensive surface water (California Water Service Company, 2020), and the presence of year-round beekeepers (Schiff and Sheppard, 1996). Our data showed that 42% of growers in Sacramento Valley obtained their bees from their county or a neighboring county, while 22% of growers in North and South San Joaquin Valley obtained their colonies from a nearby location (**Table 2**). However, a grower identifying a beekeeper as influential in pollination management did not play a statistically significant role in the adoption of any bee-friendly practices. This suggests that though growers may not identify a beekeeper as influential, diffuse and informal social interactions with beekeepers, potentially those in their communities, may still be important (Thomas et al., 2020). These results did not change qualitatively with the assignment of central San Joaquin countries in the N or S San Joaquin categories.

Conversely, Sacramento Valley growers were less likely to adopt the recommended HB BMPs (which aim to reduce pesticide use during bloom) than growers in North and South San Joaquin Valley, quite possibly because higher rainfall may require heavier fungicide use during winter to prevent "Shot hole" a common fungal disease (*Wilsonomyces carpophilus*) affecting almonds (Adaskaveg et al., 2008). Indeed, our data show that 37% of growers in South San Joaquin Valley and 28% of those in North San Joaquin Valley always adopted the recommended HB BMPs, compared to 19% in Sacramento Valley. More research would be needed to confirm exactly which HB BMPs Sacramento

Valley growers are less likely to always adopt, and the drivers that determine this.

Pollination satisfaction had statistically significant relationships with adoption, generally in line with growers' self-selected responses. Colony price and strength satisfaction did not play as important of a role as we expected, given growers' self-selected incentives. Growers who felt the price of their bee colonies was inexpensive were less interested in a bee certification program. We suggest this is because, given their lower colony costs, they did not feel the need for the price premium a certification could provide. Regarding satisfaction with colony strength, growers who were satisfied with the strength of their colonies were less likely to adopt pollinator habitat than those who were unsatisfied or did not answer the question, presumably because growers would likely only want to adopt practices that would strengthen their colonies, and growers satisfied with their colonies already perceived they were strong. Conversely, growers who were happy with the strength of their colonies were *more* likely to practice all the legal HB BMPs, which also supported our hypothesis, since growers would probably want to protect strong bee colonies from any pesticide-related harm during bloom. Interestingly, this dynamic, where growers are less likely to adopt cover crops and pollinator habitat if their colonies are strong, inadvertently penalizes beekeepers who bring strong colonies but would like to have access to diverse forage during crop bloom.

Growers' concerns about future pollination also played a statistically significant role in shaping the likelihood of bee-friendly practice adoption. We suspected that growers who were concerned about the cost of bee colonies, the potential lack of available colonies, declining bee health, and the loss of native beekeepers would all be more likely to adopt some or all the bee-friendly practices. We found that the cost of bee colonies and lack of available colonies were both statistically significant in shaping the adoption of cover crops, and the loss of native pollinators was somewhat influential in shaping the adoption of pollinator habitat, though not statistically significantly. Growers were more likely to practice the recommended HB BMPs (which focus on reducing pesticide use) if they were concerned about declining bee health, and also more likely to be interested in a bee certification if they felt the lack of available colonies was a strong concern.

Most notable was that the loss of native pollinators played no statistically significant role in determining grower adoption of bee-friendly practices. This is likely because almond pollination is primarily dependent on managed honey bees (Connell, 2000) and growers seemed less concerned about native bee populations as a result (**Table 4**). Several survey-based research studies provide some context about growers' obstacles to increasing the utilization of native pollinators in pollinator-dependent fruit crops, specifically apples, lowbush blueberries, and cranberries (Hanes et al., 2015; Gaines-Day and Gratton, 2017; Park et al., 2020). Some of these obstacles include uncertainty of native pollinators' contribution to their crop yield, the difficulty of monitoring native pollinators' population size (to determine if there are enough to pollinate an entire crop), a lack of awareness of cost-share programs to support native pollinators, and an existing reliance on honey bees. Continued research could help explain why growers are not more invested in wild bee populations when research indicates that native bees can increase the efficacy of honey bee pollination in almond orchards (Brittain et al., 2013), increase pollination services on large agricultural fields (Carvalheiro et al., 2012), and increase yields on almond varieties that were originally considered self-pollinating, but may actually benefit from some bee pollination (Sáez et al., 2020).

We expected acreage to be significant across all HB BMPs, given that larger operations might have the financial capital to invest in the labor, seeds, plants, and water involved in cover crops and pollinator habitat, and the labor capacity to practice some of the more labor-intensive aspects of the HB BMPs (such as multiple passes through an orchard to minimize tank mixing). However, acreage was only statistically significant in whether growers adopted pollinator habitat, perhaps because some larger operations keep bee colonies year-round, might want to cite bee-friendly practices in their marketing, or may have the financial capital or extra acreage to grow pollinator forage and habitat.

Social actors played a less consistent role in shaping growers' adoption of bee-friendly practices than we expected. There was high variability in the survey responses identifying which people were influential in pollination decisions. We hypothesized that growers with influential bee brokers or beekeepers would be more likely to adopt bee-friendly practices while growers with influential PCAs would be less likely to adopt the recommended HB BMPs, given their frequent affiliation with agrochemical companies (a 2004 study showed that two-thirds of all PCAs were affiliated; Brodt et al., 2005). Counter to our hypothesis, however, beekeepers did not play a statistically significant role in the adoption of bee-friendly practices or interest in a bee certification. This result was a surprise, as we expected that growers might be influenced by beekeepers requesting that certain bee-friendly practices be adopted or simply educating growers about different practices, as mentioned above. It may be that other social actors we did not include in our survey—such as growers in the respondents' network, extension specialists, or affiliates of the Almond Board or processing facilities—might have a greater influence on the adoption of practices, but further research is needed to determine this.

Bee brokers and PCAs, however, did play a role in the adoption of HB BMPs. Contrary to our expectations, our results indicate that growers were *less* likely to always practice the legally required HB BMPs if they had an influential PCA or bee broker. This result runs counter to the generally positive influence of bee brokers, beekeepers, and PCAs: we found that across every other practice these groups were associated with slightly higher adoption rates, though the results were not statistically significant (see **Figures 4–6**, **8**). This result may be due to some other factor we did not have a hypothesis for and thus did not measure that is colinear with influential beekeepers and PCAs. Further research could better contextualize these findings and the information sharing among these groups and growers.

Finally, an operation's land tenure, i.e., whether they owned the majority of the land they farm, was not statistically significant in any of the models, though it was associated with slightly lower adoption rate of cover crops and slightly higher adoption of pollinator habitat (**Figures 4**, **5**). This adds further complexity to debates around how land tenure shapes the adoption of diversified farming practices and conservation practices, which generally find that ownership incentivizes adoption of long-term conservation practices (Soule et al., 2000; Varble et al., 2016; Carlisle et al., 2019; Petersen-Rockney et al., 2021). The results from our analysis may indicate that crop type can mediate the adoption of bee-friendly practices more than land tenure. Further, our findings suggest that the bulk of bee-friendly practices are likely to be adopted regardless of land tenure in the almond industry, possibly due to the Almond Board's concerted efforts to support honey bees during bloom, such as the dissemination of the HB BMPs and promotion of cover crops and other bee-friendly practices.

Implications for Strengthening On-Farm Adaptive Capacity

Our study did not determine a single method that drives grower adoption of bee-friendly practices. Indeed, we found that none of the social, operational or bee concern-related factors included in our analysis identified a "silver bullet" that consistently predicted grower adoption of bee-friendly practices. However, our results did indicate that region plays an important role in determining which bee-friendly practices growers adopt, underscoring a general principle of farm diversification: a uniform approach to supporting pollinator health (and diversification more broadly) will likely not be as successful as a context-sensitive strategy (Kremen et al., 2012).

Just as over-simplification can create vulnerable farm agroecosystems (Petersen-Rockney et al., 2021), over-simplified pollinator strategies may weaken actual adoption or lower the desire to participate in such programs, and thus close potential pathways to diversification. Thus, organizations, regulators, and other stakeholders seeking to bolster rates of adoption may be better served by using a variety of incentivization tools rather than relying on just one and may need to recognize that these factors need to be adapted regionally based on differences in climate, social connections, and the economic context of the growers being targeted. For example, our research indicates that growers in arid regions with expensive water (such as North and South San Joaquin Valley) may be limited in their ability to grow cover crops or adopt permanent pollinator habitat but might be able to reduce pesticide applications (particularly fungicides) given their low winter rainfall (National Weather Service, 2020). Conversely, growers in regions with higher rainfall may find it challenging to lower fungicide applications during the rainy season but may be able to plant pollinator forage and habitat.

Colony strength and price, as well as growers' concerns about future pollination services, may be powerful levers to encourage the adoption of bee-friendly practices. Given that over a third of growers found their colonies too expensive (**Table 3**), and that decreased colony rental was the second most popular incentive across most bee-friendly practices (**Figure 3**), one of the most obvious incentives to adopt bee-friendly practices might be a reduction in colony price from beekeepers. The beekeeping community and Almond Board might also consider increasing communication about why the colony price has risen more sharply for almonds compared to other industries following almond bloom (Ferrier et al., 2018; Goodrich, 2019) so growers understand why these costs have risen. Alternatively, beekeepers could find ways to better demonstrate colony strength so growers can feel more satisfied with the strength they have received for the price. Other incentives might include greater outreach about funding available from existing federal or private cost-share programs that help with the installation of cover crops and pollinator habitat, or price premiums for bee-friendly growers from distributors.

Finally, further outreach may also be needed to communicate the secondary ecosystem service benefits provided by pollinator habitat enhancement to farms such as pest population reduction, protecting soil and water quality by mitigating runoff and soil erosion (Dabney et al., 2001; Marahatta et al., 2010; Crézé et al., 2018). Combining increased research and outreach with a specialized, regional, honey-bee centered pollinator approach may increase the likelihood that growers will adopt bee-friendly practices that make economic sense, strengthen their operation's adaptive capacity, and support managed and wild pollinators in turn.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author/s.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Institutional Review Board at University of California, Riverside. The participants provided their informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

JD and LP designed the study and prepared data for analysis. JD designed the survey, with input from LP, collected and processed the survey responses, and wrote the first draft of the manuscript. LP analyzed the data with input from JD and contributed to revisions. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fsufs.2021.628802/full#supplementary-material

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