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**ARTICLE**

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# Preserving local biodiversity through crop diversification

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**Abstract**

Using the case study of birds and food crops, we investigate whether diversifying crop production can enhance preservation of local biodiversity. To this end we combine annual bird survey data, high resolution land use data, and phylogenetic trees to create a landscape level panel data set covering the conterminous United States for over a decade. Our econometric analysis shows that greater local food crop heterogeneity increases local avian diversity, although this is spatially limited. Supplementary county level data provides evidence that more food crop diversity is unlikely to be at the cost of lower revenues.

**KEYWORDS**

biodiversity, birds, food crops, phylogenetic diversity

**JEL CLASSIFICATION**

Q10, Q57

## 1 | INTRODUCTION

It is widely recognized that agricultural expansion and intensification due to rising demands for food from growing populations has been the primary driver of the loss of biodiversity globally (Díaz et al., 2020; Lee & Goodale, 2018; Tollefson, 2019; Wilson et al., 2017). Worryingly, demand for agricultural commodities is predicted to further increase by 70% to 100% by 2050 (Zabel et al., 2019), leading to continued pressure for land conversion and increases in agricultural yield. As a matter of fact, reconciling growing global food demand with the preservation of biodiversity has been argued to be one of the greatest challenges of the 21st century (Dudley & Alexander, 2017). However, identifying an optimum strategy to maximize biodiversity to the extent possible within a working agricultural landscape that can also achieve food security has proven a difficult task (Wilson et al., 2017). In this regard, conservationists have often framed the array of choices in terms of the land sparing

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versus land sharing debate, where the former refers to focusing on high-yielding cultivation on a small land footprint leaving more room for natural habitat, and the latter promotes more low-yielding, wild-life friendly agriculture practiced on a larger area (Green et al., 2005). The empirical evidence of which of these approaches is more likely to promote biodiversity has, however, been rather mixed; see Kremen (2015) for a review. Moreover, with further agricultural intensification, that is, land sparing, opportunities to maintain or restore semi-natural/natural features may not only be limited due to large scale monoculture crops allowing little habitat fragmentation (Lee & Goodale, 2018), but yields may fall substantially (Lark et al., 2020).

More recently it has been recognized that the land sparing versus land sharing debate is perhaps too simplistic in terms of identifying biodiversity preserving agricultural management practices (Kremen, 2015). In particular, as argued by Frei et al. (2018), agricultural landscapes differ considerably in their impact on the local environment, their capacity to produce food, and their ability to support biodiversity. Thus, the potential to support biodiversity is likely to depend on not only agricultural intensity but also on the agricultural spatial arrangement and agricultural composition. As a matter of fact, current evidence suggests that agricultural intensity and composition may be the most important drivers of biodiversity in agricultural landscapes (Newbold et al., 2015). In this regard, crop diversification has been proposed as an alternative biodiversity preservation strategy that, by providing a variety of complementary habitats and resources, could reduce the negative effect of agriculture without necessarily reducing agricultural productivity goals (Lee & Goodale, 2018; Redlich et al., 2018). Nevertheless, conclusions regarding whether there are biodiversity benefits to landscape-level agricultural diversity are still ambiguous (Redlich et al., 2018).

In this paper we set out to investigate whether cropland diversity can help preserve biodiversity by studying the local effect of food crop diversity on avian diversity across the entire conterminous United States (US) for over 10 years. Arguably, birds and food crops in the US serve as an excellent context within which to investigate whether diversifying food crops can help preserve biodiversity. First, although the area under cropland in the US fell substantially from the late 1970s until the early 2000s, Aguilar et al. (2015) show that this happened alongside decreasing crop diversity. Moreover, even the subsequent cropland expansion that commenced in 2007 appears to have been mostly due to the growth of a few crops and to the detriment of wildlife habitat, without any gain in yields (Lark et al., 2020).<sup>1</sup> At the same time, the US has also seen considerable declines in both bird numbers and the number of bird species since the 1970s (Rosenberg et al., 2019), which by some researchers has been attributed to farmland expansion and related crop management practices (Lark & Schelly, 2018; Li et al., 2020; Stanton et al., 2018).

There is a relatively large literature that has related bird diversity to the intensity of agricultural land use (Cannon et al., 2019; Edwards et al., 2015; Frei et al., 2018; Jeliazkov et al., 2016; Kehoe et al., 2017; Santana et al., 2017; Zabel et al., 2019), with a general agreement on the existence of a negative relationship, at least after some tipping point (Sasaki et al., 2020). In contrast, there are only a handful of fairly recent studies, in a variety of different contexts, that have tried to link avian diversity to crop diversity producing overall a rather mixed set of results.<sup>2</sup> For example, in a bivariate regression analysis Frishkoff et al. (2014) found that in Costa Rica diversified agricultural systems were associated with greater bird diversity than intensive monocultures, whereas Hendershot et al. (2020) did not find any difference when examining trends of endemic and International Union for Conservation of Nature (IUCN) Red List species over a longer period using similar data for the same country. For China, Lee and Goodale (2018) estimated that the probability of a bird species occupying an area depended in part positively on the degree of crop heterogeneity, although only in

<sup>1</sup>The expansion was as a response to high commodity prices, increased demand for bio fuels, and reductions in federal land conservation programs (Lark et al., 2020).

<sup>2</sup>Birds have been the primary target for examining the question of the role of crop heterogeneity in species diversity, likely because of the relative abundance of data due to many ongoing bird counts across the globe. One exception are bees, where St. Clair et al. (2020) found no effect on honey and wild bee diversity on a sample of Iowan farms. In contrast, Alvarado et al. (2018) found that cropland heterogeneity is an important predictor of dung beetles diversity.

the winter. In contrast, for the case of Germany, Redlich et al. (2018) in a cross-sectional regression analysis covering 14 landscapes across winter fields uncovered no evidence of crop diversity influencing avian diversity, whereas Gottschalk et al. (2010) found a positive impact across 304 sampling sites in a catchment of the German region of Hesse. Finally, Josefsson et al. (2017), using Swedish farm data, show that bird species richness does not respond to crop type heterogeneity but rather to the crop distinction by management practices and vegetation structure.

Our study here makes a number of contributions to the existing literature in determining whether diversifying crops might help preserve biodiversity. First, the existing literature has treated the land sparing-sharing trade-off and the potential benefits of crop diversity for local biodiversity as two separate issues. However, these aspects of crop management are not mutually independent in that there may be substantial (dis)economies to scale and yield gains or losses due to crop diversification (Bellora et al., 2017; Klasen et al., 2016; Mzyece & Ng'ombe, 2020). We thus here examine the role of crop diversity while also taking account of the degree of land sparing and sharing. Related to this, one should note that in the current literature specifically focusing on crop heterogeneity almost all current studies have been limited to cross-sectional analysis, with the inclusion of a limited number of possibly confounding variables, usually just other landscape feature controls, in their regression analysis (Gottschalk et al., 2010; Hendershot et al., 2020; Lee & Goodale, 2018; Martin et al., 2020; Redlich et al., 2018; Wilson et al., 2017). Of the three studies that used panel data, Hendershot et al. (2020) only examined differences in trends, whereas Frishkoff et al. (2014) or Noack et al. (2021) in their regression analysis ran a pooled cross-section without accounting for location specific effects and using few controls.<sup>3</sup> However, as noted by (Redlich et al., 2018), there are likely to be a number of possibly confounding factors other than landscape features that will determine both birds and cropland choices, such as geography, agricultural management practices, and climate, to name a few. Here we attempt to take account of these through the use of panel fixed effects methods and a set of relevant controls.

Our second contribution comes from our measurement of both bird and crop heterogeneity in terms of the evolutionary distinctiveness of species, that is, phylogenetic diversity (PD). More specifically, compared to other more traditional measures that rely on the differentiation of species taxonomically or via their functional traits, PD arguably best captures the biological diversity among species and can be considered a superior tool for prioritizing and ranking conservation scenarios (Lean & Maclaurin, 2016; Weitzman, 1992). It also has been shown to be a good predictor of predictor of local ecosystems' functions and stability (Cadotte et al., 2009; Cadotte et al., 2012; Srivastava et al., 2012). However, of the existing literature cited above only Frishkoff et al. (2014) calculated PD to capture avian diversity, whereas all others used the number of species (Hendershot et al., 2020; Redlich et al., 2018; Wilson et al., 2017), diversity indices based on the number of species (Frei et al., 2018; Noack et al., 2021), or the incidence of species (Lee & Goodale, 2018). Moreover, no current study has also simultaneously used a phylogenetic measure for crop heterogeneity to capture its potential role in determining avian diversity. Rather, the existing literature has been limited to distinguishing just between a few crops and treating each crop as equally different to each other (Lee & Goodale, 2018; Wilson et al., 2017), using a simple distinction between intensive and diversified agriculture (Frishkoff et al., 2014; Hendershot et al., 2020), or grouping crops according to structural and vegetation similarity (Josefsson et al., 2017; Redlich et al., 2018), thus arguably oversimplifying the degree of diversity of crops.

The analysis in our study crucially rests on the assembly of four different rich data sets. First, we use the North American Bird Breeding Survey (BBS), which provides an annually conducted count of bird numbers and species along predetermined geographically representative routes across the continental United States. We combine these with the high resolution (30 m) Cropland Data Layer (CDL) data, which allows us to determine the distribution of crop types, along with other land uses,

<sup>3</sup>Noack et al. (2021) are mainly interested in the effect of farm size on bird diversity and in a novel approach use differences between the former eastern and western German borders to isolate a causal effect for this, while controlling for crop diversity.

around the BBS routes. In order to construct local phylogenetic diversity measures along the bird routes, we resort to the phylogenetic trees generated for birds and food crops by Jetz et al. (2012) and Milla (2020), respectively. These four information sources allow us to generate a panel data set of bird and food crop phylogenetic diversity at over 2500 location over 11 years (2008–2018), covering the entire conterminous US. Our econometric analysis using these data shows that food crop heterogeneity plays a substantial role in preserving local avian biodiversity, although the spatial extent of this appears to be limited. Employing county level data, we also provide evidence that once one controls for differences in input intensity, diversifying food crop production phylogenetically appears not to lead to revenue losses.

The remainder of the paper is organized as follows. In the next section we define the diversity indices used in the analysis. Section 2.1 describes the data and provides summary statistics. In Section 3 we provide the results of the econometric estimation of the impact of phylogenetic crop on phylogenetic bird diversity. Section 4 explores whether greater phylogenetic crop diversity reduces crop revenues using county level data. The final section concludes.

## 2 | DIVERSITY INDICES

Traditionally biodiversity metrics have been based on taxonomic classifications of species (Magurran & McGill, 2011). However, although taxonomic metrics are easy to construct, they ignore differences between species in their function and form, which have been shown to be particularly important for local ecosystem functions and stability (Cadotte & Tucker, 2018), and a better representation of biodiversity (Srivastava et al., 2012). This has spurred some researchers to create biodiversity indices based on functional traits, defined as morpho-physio-phenological traits that impact the fitness of individual species via their effects on growth, reproduction, and survival; but gathering, classifying, and quantifying these traits is generally difficult (Cadotte et al., 2012; Gravel et al., 2012; Staab et al., 2021). An increasingly more popular alternative is phylogenetic measures, which are based on molecular data and capture the extent to which species differ in their evolutionary history (Magurran & McGill, 2011). Importantly, phylogenetic measures of diversity have not only been shown to capture differences in functional traits among species (Cadotte & Tucker, 2018; Gravel et al., 2012) but also a reasonable predictor of local ecosystems' functions and stability (Cadotte et al., 2009; Cadotte et al., 2012; Srivastava et al., 2012).

In the context here, we are particularly interested in how greater food crop diversity may serve to provide a greater variety of habitats and food sources for birds and thus encourage greater local bird diversity. With regard to the former, Xie et al. (2018) showed that plant phylogenetic diversity is a good predictor of differences in plant height, leaf area, and dry leaf matter, and much more so than taxonomic diversity. In terms of the provision of food sources, apart from more local crop diversity obviously being able to provide a greater variety avian herbivores, it has also been shown that phylogenetic rather than taxonomic diversity is a good representation of the diversity and number of types of herbivores and predatory arthropods (Dinnage, 2013; Dinnage et al., 2012; Egorov et al., 2017; Wang et al., 2020), which may serve as food for carnivorous birds.

### 2.1 | Phylogenetic diversity index

DNA-sequencing data allow one to construct the phylogeny, that is, the history of the evolution of a species. The phylogenetic distance between any two species then represents the estimated amount of time since the most recent common ancestor. It can thus serve as a quantifiable metric for the extent of phenotypic differences, that is, differences in observable characteristics and traits, between them. Although a number of earlier measures of phylogenetic diversity only considered the presence or

absence of a species in a phylogenetic tree, Cadotte et al. (2010) argues that the ecological interactions within communities are determined by both the evolutionary histories and the abundance of species. Additionally, Hillebrand et al. (2008), in a review of the empirical evidence on the importance of presence–absence (richness) versus the relative abundance (evenness) of species in ecosystems, point out that that species' evenness has important consequences for ecosystems long before any species is driven to extinction due to population dynamics and stability. More specifically, the authors conclude that “to be able to predict the consequences of human impact, it is imperative to consider the conclusion of all diversity elements [richness and evenness]” (p. 1517).

The proxy of phylogenetic diversity we use in this study incorporates both genetic richness and evenness, and is based on the community diversity measure by Cadotte et al. (2010). Consider a set of species  $i = 1, \dots, I$ , along a local area, that is, community,  $j$ , each with abundance  $n_i$ . Consider also an evolutionary tree,  $T$ , the root to which each species  $i$  is connected to through a series of branches  $b$ . One can then define abundance weighted evolutionary distinctiveness (AED) of species  $i$  as:

$$AED_{ijT} = \sum_{b \in \mathbf{B}(ijT)} \lambda_b \frac{n_{iT}}{N_{jT}} \quad (1)$$

where  $\mathbf{B}$  is the set of branches that connect species  $i$  to the root of the tree,  $\lambda_b$  is the length of each of these branches  $b$  in terms of evolutionary years,  $n_{ij}$  is the abundance of species  $i$ , and  $N_j$  is the total abundance of species that share the same set of branches  $B$ . In essence Equation (1) divides the evolutionary information of branches among all species that share these branches according to their relative abundance in the community.

The entropy measure of the abundance weighted evolutionary content of species  $i$  from community  $j$  for tree  $T$  that we use here is then defined as:

$$HAED_{jT} = - \sum_{i=1}^j \frac{AED_{ijT}}{\sum_{i=1}^j AED_{ijT}} \ln \left( \frac{AED_{ijT}}{\sum_{i=1}^j AED_{ijT}} \right) \quad (2)$$

where a higher value of  $HAED$  indicates higher phylogenetic diversity of community  $j$  for tree  $T$ . One should also note that  $HAED$  satisfies the replication principle for species neutral diversity, according to which if there are  $N$  equally large and diverse groups with no species in common, then the diversity of the pooled group must be  $N$  times that of a single group.

## 2.2 | Taxonomic diversity indices

A widely used taxonomic diversity measure in the biodiversity literature is the Shannon-Wiener index (Magurran & McGill, 2011):

$$SHAN = - \sum_{i=1}^R p_i \times \ln(p_i) \quad (3)$$

where  $p_i$  is the proportion of individuals that belong to species  $i$  and  $R$  is the number of species. One should note that a higher  $SHANNON$  implies higher biodiversity. In addition to the Shannon-Weiner index we also, as is common, use the number of species to capture taxonomic diversity.

### 3 | DATA AND SUMMARY STATISTICS

#### 3.1 | North American breeding bird survey

The North American Breeding Bird Survey (BBS) is the primary information source for the estimation of bird population trends and the modeling of the potential impact of its possible stressors in the US. First implemented at the continental level in 1968, the BBS is conducted annually in the summer along randomly selected secondary roads that are representative of the entire region, where each route is roughly 39.4 km (24.5 miles) long. Importantly the surveying approach employed in the BBS has remained unchanged since its inception, in that a voluntary observer selects a morning and, commencing 30 min before sunrise, will at 50 predetermined stops, positioned 800 m apart, exit his/her vehicle and conduct a 3 min count, recording the different bird species and their numbers seen or heard within a 400 m radius area. One may want to note that in order to ensure consistency of sampling the same stops are surveyed each year. Although there are currently about 3900 active routes in the BBS, because the actual surveying depends on the availability of volunteers, not all routes are covered in every year. For the sample period used here (2008–2018), a total of 3202 routes were covered with on average 8 out of 11 possible annual observations.

Birds in the BBS are classified according to the American Ornithologists' Union (AOU) categorization. There are currently 914 naturally occurring bird species in the US, and after dropping unusable observations and birds that are non-native to the US, we were left with a total of 700 species in our sample period. One may want to note that by focusing only on native species, we remove the possibility of avian diversity increasing purely because of the spread or introduction of non-native species, considered by conservationists to be an undesirable outcome.

A number of other survey specific variables are collated in the BBS. More specifically, each observer is given a unique identification number, allowing one to construct measure of observer experience, as well as if an assistant was present during the survey. Information is additionally collected in terms of the number of cars seen, the number of stops at which there was noise present, as well as the temperature, clarity of the sky in terms of the Weather Bureau scale, and the observed wind speed according to the Beaufort scale at the start and end time of the survey morning. We use all of these factors as route level controls in our analysis.

#### 3.2 | Definition of relevant landscape

We define a “community,” or landscape, around a BBS route as the area that is within a 1 km buffer of route. One should note that the choice of 1 km as our benchmark definition of “community” is to allow for some measurement error in the surveyor's gauge of 400 m distances. Moreover, using a distance greater than 400 m will better capture edge effects (Pidgeon et al., 2007), that is, changes in species abundance and community structure in the ecotone between two contrasting habitats (Zurita et al., 2012). In this regard one should note that most studies exploring the relationship between avian and crop diversity focus on fairly small localities; see, for instance, Redlich et al. (2018); Martin et al. (2020). We nevertheless in robustness checks experiment with smaller and greater landscape definitions.

#### 3.3 | Cropland layer

The US Department of Agriculture (USDA) NASS Cropland Data Layer (CDL) product is an annual raster formatted, geo-referenced, crop-specific land cover map based on satellite imagery and extensive agricultural ground truth data (Boryan et al., 2011). Originally available in 1997 only for North Dakota, it was subsequently gradually expanded to additional states, and since 2008 it has covered



the entire conterminous United States. As such it provides land cover types, including 77 major food crops, at a 30-m spatial resolution. We extract the grid cells in the CDL that are within the 1 km buffer around the BBS routes to generate land cover type classifications of each route's proximity. The CDL data are also used to determine the share of non-food crops (tobacco, cotton, hay, and other crops), forest, wetlands, fallow cropland, shrubland, barren land, pasture, high developed areas, medium developed areas, low developed areas, and open developed areas.<sup>4</sup> For food crops we also calculate the mean size of "patches," that is, the average size of contiguous areas of land occupied by these within the route 1 km proximity.

### 3.4 | Avian phylogenetic trees

Phylogenetic (evolutionary) trees map out how species evolved from a series of common ancestors and thus represent their evolutionary relationships. We use the phylogenetic trees generated by Jetz et al. (2012) to construct measures of avian phylogenetic diversity for each bird route at each point in time. To construct these evolutionary trees for all 9993 extant avian species, Jetz et al. (2012) combine available molecular data for 6663 species and combine these with taxonomic constraints for the remaining 3300 data-deficient species to construct a set of possible phylogenetic trees.<sup>5</sup> More specifically, using backbone phylogenies each species was assigned to one of 158 clades (branches). Then 10,000 possible relaxed-clock trees<sup>6</sup> for the clades were constructed combining species with and without genetic data, where those without were placed within their clade using constraint structures that were consistent with taxonomic information, branching times sampled from a pure birth model of diversification,<sup>7</sup> and the backbone trees constructed from the molecular data.

One should note that whereas Jetz et al. (2012) used the Birdlife V3 World List to identify species, the AOU distinction of species used in the BBS does not completely match that of the modified Birdlife V3 World List. In particular, in the AOU list bird types are on occasion divided into more taxa than in the Birdlife V3 World List. To match bird species in the AOU to the phylogenetic trees, we combined a total of 95 AOU species into a common taxa in line with the Birdlife V3 World List, resulting in a total of 631 different species on the BBS routes. For each route-year observation, the species sighted allow the extraction of the sub-tree that represents all sighted species of each of the 10,000 complete avian evolutionary trees. This allows us to calculate HAED in Equation (2) for each of the possible 10,000 phylogenetic trees. Because there is no reason to prefer one tree over another, the mean HAED across all trees is used as the abundance weighted phylogenetic avian diversity measure.<sup>8</sup>

### 3.5 | Food crop phylogenetic trees

To construct phylogenetic trees for the bird routes in terms of the local food crops, we resort to the Phylo Food v.1.0 database (Milla, 2020). The Phylo Food database provides a phylogeny for 866 food crops and is based on the time-calibrated molecular phylogeny of 74,533 plant species as constructed in Jin and Qian (2019), which is an update and combination of information from Zanne et al. (2014) and Smith and Brown (2018). Using the backbone tree from Jin and Qian (2019), Milla (2020) are

<sup>4</sup>The developed areas are defined mainly by the percentage of impervious surfaces, where these constitute in open, low, medium, and high have less than 20%, 20%–49%, 50%–79%, and 80%+, respectively.

<sup>5</sup>The lack of data for some of the species meant that only a set of possible, rather than one exact, tree could be constructed.

<sup>6</sup>Correctly estimating the rate at which mutations accumulate in a lineage is essential for phylogenetic analysis, and under a relaxed clock model lineages are assumed to be independent on each other rather than to evolve at the same rate. The output of these models are known as relaxed clock trees (Fourment & Darling, 2018).

<sup>7</sup>Under a pure birth rate model every lineage is assumed to have the same mutation rate.

<sup>8</sup>Using the median produced the same qualitative and nearly identical quantitative results in our empirical analysis.



able to prune 661 crop binomials. For the remaining 205 food crop taxa the author used other published studies to construct the phylogenies.

In matching the food crop categories in the CDL with those in the Phylo Food phylogenetic tree database there were two challenges. First, some CDL cells were classified as consisting of two crops. For these we for simplicity sake assumed that half of the 30 m grid cell was allocated to each crop type. Second, the species for some crops in Phylo Food database were more aggregated in categorization than the CDL crop breakdown. Thus, for some categories of crops in the CDL, we necessarily assigned the same genetic category despite there likely being some genetic differences (e.g., broccoli and cauliflower, or corn and sweet corn), albeit probably small. Overall, this was done for 18 out of the total of 77 major food crops in the CDL, thus leaving us with a total 67 different genetic classification groups.

### 3.6 | Other route specific variables

#### 3.6.1 | Climate controls

A number of studies have shown that climate affects bird populations in the US and this may be linked to the presence of agricultural land use (Correia et al., 2020; Ferger et al., 2017; Karp et al., 2018; Vollstädt et al., 2017). To take account of this we generate controls for mean monthly precipitation and temperature around the bird routes. The source for these is the Precipitation Elevation on Independent Slopes Model (PRISM), that is, the official spatial climate data sets of the US Department of Agriculture. The PRISM data provide, among other climate variables, monthly values of precipitation and average temperature, which are constructed from weather station data taking elevation into account. The gridded data output is available at an approximately 800 m resolution (Daly et al., 2008). To isolate the climatic data relevant to our BBS routes, we mask those cells within 1 km of the routes and then calculate out for each route the average monthly precipitation and temperature of 12 months period prior to the survey date.

#### 3.6.2 | Crop yield

Because crop yield is an important component of the land sharing versus land sparing trade off one ideally would like to have a local reported measure of crop yield. As this is not available at the BBS route level, we instead use a satellite derived measure of gross primary production (GPP), that is, the amount of carbon fixed during photosynthesis by all producers in the ecosystem. In this regard, He et al. (2018) have shown that for the US GPP captures well characteristic local cropland productivity patterns and seasonal variations, and also corresponds favorably with reported county-level crop production data in the US. To measure GPP along the BBS routes we use the Landsat Gross Primary Production CONUS 30 m resolution derived product. More precisely, we masked the 16-day measures of GPP with the annual food crop areas derived from the CDL within 1 km buffers of the BBS routes to create monthly mean series. For each route we then used the date of the interview to determine the previous 12 months of GPP along the route in food crop areas to calculate a mean monthly route specific value.

#### 3.6.3 | Neonicotinoids and other pesticides

In a review of the literature on the determinants of farmland bird decline in the US Stanton et al. (2018) noted that 42% of all 122 studies surveyed identified pesticides as the most prominent negative driver, where in particular neonicotinoids, that is, systemic agricultural insecticides, are likely

to be culprits (Li et al., 2020). In order to construct estimates of neonicotinoids use at the route level, we use state level information on individual pesticides use as gathered from the Pesticide National Synthesis Project (PNSP), in combination with crop land use at the route level. More specifically, data on pesticide use for the PNSP was compiled from proprietary surveys of farm operations, except for California, where this information was obtained from annual Department of Pesticide Regulation-Pesticide Use Report; see Baker and Stone (2014) for details. At the crop level, the data provides state level annual estimates of each pesticide type by 10 crop groups. We use these with state level crop land use areas calculated from the CDL to determine the annual state level per km<sup>2</sup> per crop group use of the neonicotinoids (imidacloprid, thiamethoxam, clothianidin, acetamiprid, dinotefuran, and thiacloprid) in the data base. Land use for each of the 10 crop groups from the CDL at the route level then provides us with an estimate of the quantity used of each of the six neonicotinoids around each BBS route. As in Li et al. (2020), we weight the individual neonicotinoid by their relative toxicity to arrive at annual estimates of route level toxicity weighted total neonicotinoid use. For all other pesticides we use the same procedure except we do not toxicity weight these because such weights are currently unavailable in the literature.

### 3.6.4 | Fertilizer

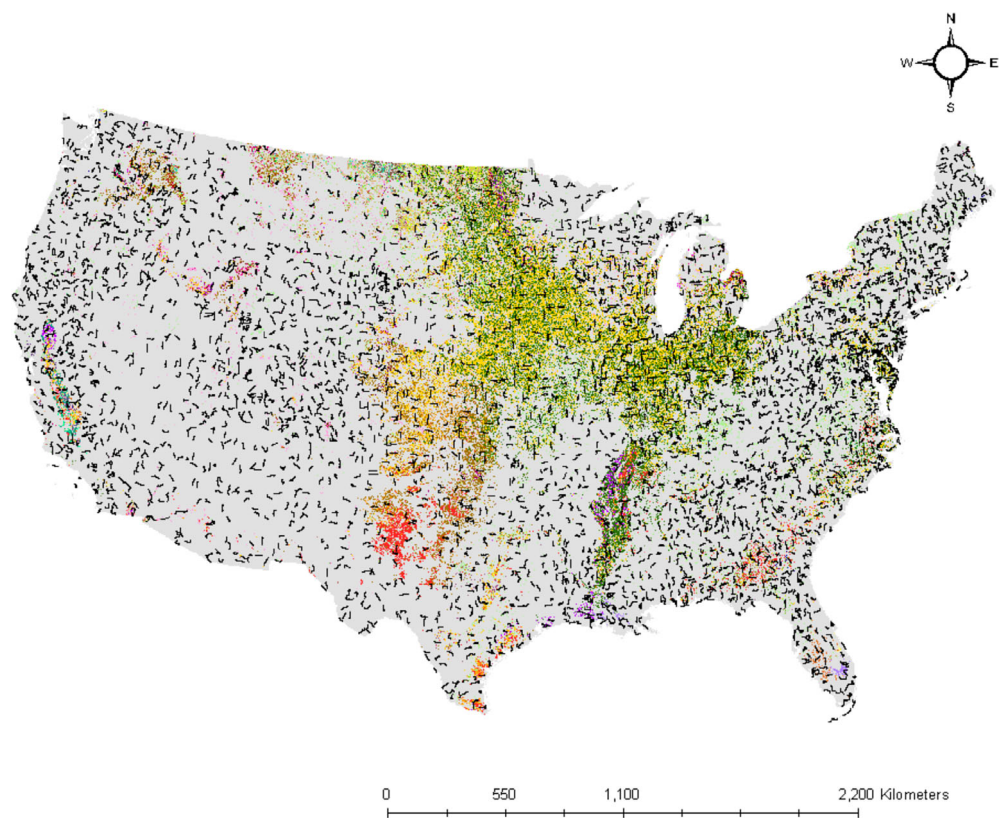
To proxy the localized use of fertilizer we avail of the spatial data set created by Cao et al. (2018). These data provide annual fertilizer use at the 5 km resolution and were constructed for the period 1850 to 2015 using state-level crop-specific fertilizer use and geographical land cover data. The units are given in gNm<sup>-2</sup>.

### 3.6.5 | Irrigation

In order to measure local irrigation, we resort to the Moderate Resolution Imaging Spectroradiometer (MODIS) Irrigated Agriculture Dataset for the United States (MIrAD-US), which was developed by the U.S. Geological Survey (USGS) Earth Resources Observation and Science (EROS) Center. More specifically, using information from the Agricultural Census and MODIS satellite imagery, the MIrAD-US is a comprehensive and consistently processed geospatial dataset of the percentage of land of irrigated agriculture across the conterminous U.S at a 500 m resolution using the method of Pervez and Brown (2010). As the Agricultural Census is only conducted every 5 years, the MIrAD-US is available only for the years 2002, 2007, 2012, and 2017. In order to be able to couple these data with as many years as are available from the CDL, we use the data for 2007, 2012, and 2017 to proxy the irrigated agriculture around the BBS routes for the years 2008, 2013, and 2018, respectively. Thus the measurement error of using the previous year's irrigation extent for the current year is equal across the three time periods.

### 3.6.6 | Critical habitat designation

When a species is listed under the Endangered Species Act, federal law requires that the U.S. Fish and Wildlife Service (USFWS) identifies areas that contain the physical or biological features essential to its conservation. These critical habitats provide key protection for listed species by prohibiting federal agencies from permitting, funding, or carrying out actions that adversely modify designated areas. As of this date, there are critical habitats defined for 31 avian species, and these may potentially affect within and nearby bird diversity. In order to determine which are within the vicinity of the BBS routes, we masked the avian critical habitat polygons provided by the USFWS ECOSEnvironmental Conservation Online System within the 1 km buffers around the routes and,



**FIGURE 1** Food crop and BBS route distribution (2018)  
*Notes:* (i) Black lines indicate BBS routes; (ii) colored (non-gray) areas are food crops, where different colors indicate different crops

using their publication date, created a time varying measure of the share of the area within the vicinity of each BBS route designated as critical habitat.

### 3.6.7 | County level crop variables

The U.S. Bureau of Economic Analysis publishes annual data on county-level farm receipts and expenses accruing to proprietorship farms, corporations, and hired farm labor as part of their local area personal income statistics. In terms of receipts it provides a breakdown of that due to crop and non-crop sources, and we use the former noting that it does not allow us to isolate those specifically due to food crops. Production expenses are in terms of total (crop and non-crop combined) spending on fertilizer, seeds, petroleum products, and labor. Additionally we resort to the county level total pesticide use estimates from the PNSP. All monetary series (crop receipts and expenses) were deflated to be in 2018 prices and were normalized by area (km<sup>2</sup>) of total (food and non-food) cropland use in a county as calculated from the CDL.

### 3.7 | Summary statistics

We provide a map of the conterminous United States along with the BBS routes (black lines) and the food crop areas (in color) in Figure 1. As can be seen, the BBS routes are fairly evenly geographically

distributed and thus arguably spatially representative of the US. This stands in contrast to the location of food crops where there is a clear visual agglomeration in the mid-west. If one considers the crop diversity within areas, as depicted by the range of colors, one can notice that although some smaller areas have considerable food crop variety, much of the mid-western region is visually dominated by a few large scale crops.

After dropping observations with missing values for any of our route specific variables, except for irrigation and fertilizer use, our sample consists of an unbalanced annual panel of 3202 units with a total of 25,490 observations. Summary statistics of all route specific variables used in the analysis are provided in Table 1. Accordingly, the mean HAED for avian species along the BBS routes is on average much larger than that of food crops within 1 km of the surveyed routes, standing at 3.15 versus 1.56. We also show kernel density distributions of these measures in Figure 2. From this and the summary statistics it is clear that the observed phylogenetic diversity is very much un-aligned across birds and the food crop within the habitat of the bird sightings. More specifically, nearly half (49.8%) of route-year observations have genetic crop diversity smaller than the minimum observed avian diversity (1.7). In contrast, more than three quarters (75.3%) of all route-year observations in our sample have higher genetic diversity than the maximum observed genetic crop diversity found. One may also want to note that the distribution for food crops is much flatter than that of avian species.

In Figure 3 we depict the mean phylogenetic indices over time. As can be seen, food crop HAED is generally on a downward trend from 2009 onward, except for 2018 when there was a relatively large jump in value, possibly due to the fall in soybean price in that year. In contrast, there is little discernible temporal pattern in avian HAED, although one may want to note that the fall at the end of the sample period coincided with the corresponding rise in its food crop counterpart, suggesting a negative relationship. We also explored comparing temporal trends within routes by regression both the avian and the food crop HAED separately on a time trend for each route that had at least six observations. A scatter plot of the coefficients on the time trends for the avian versus food crop phylogenetic diversity measure for each route, along with a fitted regression line across all of the estimated coefficients, is shown in Figure 4. Accordingly, there is a large spread of trends across the two indices, although the regression line tentatively suggests a positive relationship, implying that on average trends for both indices tend to have a positive association within routes.

Table 1 also shows the two alternative, more traditional, diversity indices. Comparing the two Shannon indices of diversity similarly as for the phylogenetic counter parts demonstrates that diversity is larger for birds than food crops along the BBS routes, where the mean of the former (3.22) is over four times that of the latter (0.87). In terms of the number of species for birds a mean of 54 is observed, whereas for food crops there are about on average eight different types present along the 1 km vicinity of the routes.

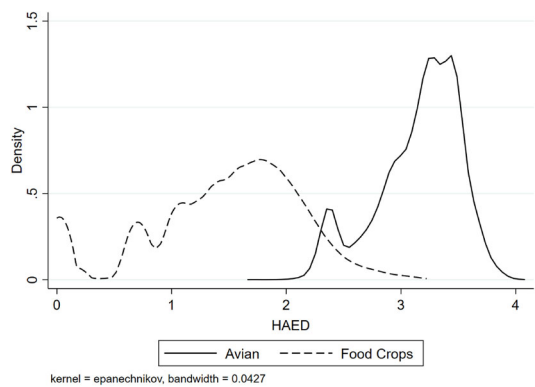
Examining the decomposition of land use shows that food crops occupy on average 17% of the landscape, where each patch of contiguous food crop covers roughly 0.32 km<sup>2</sup>. For other land use types around the BBS routes, one can see that the highest incidence is forested land (33%), followed by pasture (17%), and then shrubland (13%). The average total share of urban areas is 8%, of which seven percentage points are classified as medium and low developed areas. The mean monthly GPP in the food crop areas is 0.05 kg\*C/m<sup>2</sup>/16-day, but is as large as 0.13 along some routes. Food crop patch size is on average 0.32 km<sup>2</sup> with a similar standard deviation. Nevertheless patch size was as large as 0.92 km<sup>2</sup> over our sample period.

Our estimated toxicity weighted use of neonicotinoids along a route is a little around 11, but with considerable variation. For non-neonicotinoids the mean and standard deviation are 0.61 and 1.51, although these are not strictly comparable as they simply represent kg/km<sup>2</sup> and are not toxicity weighted. One should note that the area designated as critical habitat for specific bird species is relatively minor, only constituting on average 0.01 km<sup>2</sup>. The average use of fertilizer is

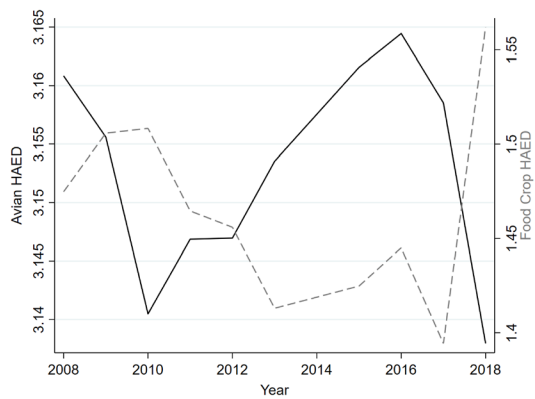
**TABLE 1** Summary statistics—route level data

Variable:	Defn. (unit)	Avg.	Std.	Min.	Max.
HAED[A]	Route avian HAED	3.15	0.37	1.7	4
HAED[F]	Food crop HAED	1.56	0.67	0	2.9
SHAN[A]	Avian Shannon index	3.22	0.43	0.42	4.88
SHAN[F]	Food crop Shannon index	0.87	0.48	0	3.29
SHAN[C]	All crop Shannon index	0.84	0.49	0	3.2
NRA	Bird species (#)	54	15	6	267
NRF	Food crop (#)	7.52	4.93	0	41
FCROP	Food crop (%)	0.17	0.23	0.00	0.94
GPP	Gross primary Pr. (kg*C/m <sup>2</sup> /16 – day)	0.05	0.02	0	0.13
PATCH	Food crop patch (km <sup>2</sup> )	0.32	0.32	0.00	0.92
OCROP	Other crop (%)	0.01	0.04	0.00	0.77
TOBACCO	Tobacco (%)	0.0001	0.001	0.00	0.06
COTTON	Cotton (%)	0.007	0.04	0.00	0.77
HAY	Hay (%)	0.02	0.04	0.00	0.65
URBANH	High urban (%)	0.001	0.01	0.00	0.26
URBANM	Medium urban (%)	0.01	0.02	0.00	0.42
URBANL	Low urban (%)	0.02	0.03	0.00	0.42
URBANO	Open urban (%)	0.05	0.04	0.00	0.39
WETL	Wetland (%)	0.07	0.14	0	0.97
FOREST	Forest (%)	0.33	0.29	0	0.98
FALLOW	Fallow crop (%)	0.01	0.04	0	0.72
SHRUBL	Shrubland (%)	0.13	0.24	0	0.99
BARREN	Barren (%)	0.00	0.02	0	0.48
PASTURE	Pasture (%)	0.17	0.20	0	0.99
CEREAL	Cereal (%)	0.49	0.39	0	1
FRVGNT	Fruits, veg., & nuts (%)	0.21	0.31	0	1
CHABIT	Critical habitat (%)	0.01	0.05	0.00	1.00
NEON	Neonicides (tox. wght. kg/km <sup>2</sup> )	10.95	37.20	0	1706.70
NNEON	Non-neonicides (kg/km <sup>2</sup> )	0.61	1.51	0	53.94
FERT	Fertilizer (gNm <sup>-2</sup> )	0.29	4.30	0.00	325.81
IRRIG	Irrigated (%)	0.03	0.10	0.00	0.91
TEMP	Monthly temperature (°C)	11.83	5.03	−1.35	25.95
RAIN	Monthly rainfall (mm)	80.79	37.64	1.08	335.81
OBEXP	Obs. experience (years)	10.63	12.73	1	151
CARS	Cars (#)	57.23	94.88	0	2969
NOISE	Noise (#stops noise)	2.46	4.11	0	50
ASSIST	Assistant	0.27	0.44	0	1
STSKY	Start sky (scale)	0.89	1.16	0	8
ENDSKY	End sky (scale)	0.84	1.06	0	8
STTEMP	Start temperature (°F)	15.68	6.05	−18	294
ENDTEMP	End temperature (°F)	22.96	5.74	−18	83
STWIND	Start wind (°F)	0.74	0.97	0.00	5.00
ENDWIND	End wind (°F)	1.62	1.15	0.00	6.00
STTIME	Start time (hrmin)	0509	0047	0401	1300
ETTIME	End time (hrmin)	1407	0079	1300	1730

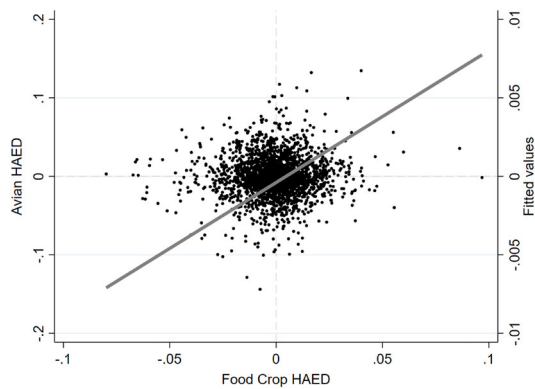
Note: (a) Statistics based on 25,490 observations; (b) data sources: BBS, CDL, and author's own calculations.



**FIGURE 2** Distribution of avian & food crops HAED  
*Notes:* Kernel density distribution of route level values of avian and food crop HAED using optimal bandwidth



**FIGURE 3** Average avian & food crops HAED  
*Notes:* Annual trend in average route level value of avian and food crop HAED



**FIGURE 4** Trends in BBS route avian versus food crops HAED  
*Notes:* Scatter plot of estimated coefficients of regressing alternatively avian and food crop HAED on a time trend for each route

**TABLE 2** Summary statistics—county level data

Variable:	Defn. (unit)	Avg.	Std.	Min.	Max.
<i>CROPR</i>	'000s \$/km <sup>2</sup>	3.86	142.15	0	20647.27
<i>FCROP</i>	Food crop (%)	0.19	0.24	0.00	0.91
<i>GPP</i>	Gross primary pr. (kg*C/m <sup>2</sup> /16 – day)	0.06	0.01	0	0.14
<i>PATCH</i>	km <sup>2</sup>	0.50	0.29	0.001	0.91
<i>HAED[F]</i>	Food crop HAED	2.08	0.47	0.00	3.43
<i>TOBACCO</i>	Tobacco (%)	0.0001	0.001	0.00	0.32
<i>COTTON</i>	Cotton (%)	0.008	0.038	0.00	0.69
<i>HAY</i>	Hay (%)	0.02	0.03	0.00	0.48
<i>PASTURE</i>	Pasture (%)	0.01	0.03	0.00	0.41
<i>OTHCROP</i>	Other crop (%)	0.0008	0.007	0.00	0.32
<i>PEST</i>	Pesticides (kg/km <sup>2</sup> )	16.061	922.072	0	996,629
<i>FERT</i>	Fertilizer (\$'000s/km <sup>2</sup> )	0.66	12.21	0	1183.04
<i>SEED</i>	Seeds (\$'000s/km <sup>2</sup> )	0.249	5.49	0	749.45
<i>PET</i>	Petroleum (\$'000s/km <sup>2</sup> )	1.19	20.18	0	1872.37
<i>LEXP</i>	Labor expenses (\$'000s/km <sup>2</sup> )	2.36	71.38	0	10082.74
<i>OTHER</i>	Other expenses (\$'000s/km <sup>2</sup> )	10.91	188.27	0	18833.77
<i>TEMP</i>	Monthly temperature (°C)	12.70	4.67	0	25.33
<i>RAIN</i>	Monthly rainfall (mm)	86.52	33.18	0	309.47

Note: (a) Statistics based on 33,089 observations; (b) data sources: BBS, CDL, BEA, and author's own calculations.

0.29 gNm<sup>-2</sup>, whereas the share of irrigated land is 3%, although one has to keep in mind that these variables are not available for the full 11 years of data. There is considerable climatic variation across routes. For example, some routes have experienced essentially no rainfall in the past year, whereas others were subjected to mean monthly precipitation of 336 mm. Similarly, mean temperature varied from -1.35 to 25.95°C.

In terms of the other route specific characteristics as derived from the BBS survey, one should note that the average observer experience is close to 11 years, and close to a third of times these are assisted by another person. Start time was around 5 a.m. with temperatures at 15°F, and this rose to 23°F by the end of the survey effort. On average, skies at the beginning and end were around 0.8 on the Beaufort scale, where zero corresponds to a clear sky and the value of one to a partly cloudy sky.<sup>9</sup> In any survey round about 57 cars are observed, and for roughly two of the 50 stops the observer reported noise.

Examining the county level variables in Table 2, the percentage of food crops at the county level is two percentage points higher than at the route level and with a slightly higher yield. Unsurprisingly, given that the route buffers will tend to slice some larger patches that extend beyond the 1 km<sup>2</sup> landscape definition, the average patch size is substantially larger than at the route level. Counties receive on average 3860 USD per km<sup>2</sup> for crop production. In terms of inputs, they use about 551 kg of pesticides per km<sup>2</sup>, but this also differs widely across the sample. For the other inputs, which are for total crop and non-crop production, the largest component is the *OTHER* category, which will also include pesticide use, standing at nearly 11,000 USD. This is followed by spending on labor (23,600), petroleum (1190), fertilizer (660), and seeds (249).

<sup>9</sup>The scale goes up to the value of eight, which corresponds to showers.



## 4 | IMPACT OF FOOD CROP ON AVIAN DIVERSITY

### 4.1 | Econometric specification

In order to explore whether food crop phylogenetic diversity has led to bird phylogenetic diversity along the BBS routes our main econometric specification is as follows:

$$\begin{aligned}
 HAED[A]_{jt} = & \beta_0 + \beta_{FCROP} FCROP_{jt} \\
 & + \beta_{HAED[F]} HAED_{jt} + \beta_{HAED[F] \times FCROP} HAED_{jt} \times FCROP_{jt} \\
 & + \beta_{LSS} LSS_{jt} + \beta_{LSS \times FCROP} LSS_{jt} \times FCROP_{jt} \\
 & + \beta_X X_{jt} + \beta_C C_{jt} + \beta_Z Z_{jt} + \beta_{TREND} TREND_{bt} + \beta_{TREND^2} TREND_{bt}^2 \\
 & + \beta_\pi \pi_t + \mu_j + \varepsilon_{jt}
 \end{aligned} \tag{4}$$

where the subscript  $j$  refers to the community around a BBS route and  $t$  to the year of observation.  $HAED[A]$  and  $HAED[F]$  are the avian and food crop diversity indices as calculated using phylogenetic data in Equation (2).  $FCROP$  captures the share of total area around the BBS routes that is being used for food crops, where as a starting point this is defined relative to all other land use types. Note that we interact  $HAED[F]$  with  $FCROP$  to allow for scale effects (Merlos & Hijmans, 2020), that is, that the effect of food crop diversity is very likely dependent on the extent of local food crops.  $LSS$  is a vector of proxies intended to capture the extent of land sparing and land sharing. In this regard, although the land sparing versus land sharing trade-off is generally discussed as a dichotomous choice for heuristic reasons, in reality it encompasses a large spectrum of potentially complex options between the two conservation strategies (Phalan et al., 2011). In its very basic form land sparing is achieved by the reduction in area devoted to crops by increasing the yield on the land used. We capture such land sparing by using our route specific measure of GPP on food crop areas and interacting it with  $FCROP$ . In contrast, a land sharing approach entails farming in a complex land matrix that contains wildlife-friendly and natural habitat (Tello et al., 2020). To roughly proxy this, we use our measure of the average size of contiguous food cropland patches ( $PATCH$ ) around routes, where the underlying logic is that smaller patches enable a more complex, and possibly more wildlife friendly, landscape matrix. One should note that interacting  $PATCH$  with  $FCROP$  not only allows us to control for scale effects in land sharing but also for the possibility that habitat fragmentation may affect the extent to which land sparing is effective in preserving biodiversity (Lamb et al., 2016).

Of the remaining controls the vector  $C$  captures climate (rainfall and temperature) factors, whereas  $Z$  constitutes a vector of the survey specific controls, including observer experience, presence of an assistant, weather factors at the start and end of the survey, noise and car sightings, and indicator variables for the month and weekday of the survey. We also experiment with including other route specific controls  $X$  that may be correlated with crop choices and avian diversity, namely observer experience, presence of an assistant, noise presence, number of cars seen, temperature, clarity of sky, and observed wind speed.  $TREND$  and its value squared are bird conservation region (BCR) time trends, where the conterminous US contains or intersects 30 such BCRs. These are ecologically distinct regions in North America with similar bird communities, habitats, and resource management issues. Their time trends thus capture possibly non-linear changes in these aspects across the BCRs. Finally,  $\pi$  and  $\mu$  are yearly dummies and route specific effects, and  $\varepsilon$  is the error term. One should note that we purge the route specific effects  $\mu$  by using a fixed effects estimator and that standard errors are computed using the methodology by Driscoll (1998) in order to allow for possible spatial dependence and serial correlation in the error term. All coefficients are reported

as beta (standardized) coefficients in order to make quantitative comparisons across variables more straightforward.

## 4.2 | Main results

We provide the results of estimating Equation (4) in the first column of Table 3, where we include, as a starting point, only the share of food crops, climate controls, survey specific characteristics, year dummies, and BCR time trends, and their value squared. As can be seen, the share of food crops along a route does not have a significant impact on avian phylogenetic diversity. We next include our proxy of crop yield, GPP, along with its interaction term with the local food crop share. This now produces a significant negative coefficient on *FCROP*, as well as a significant positive effect on the interaction term, although there is no independent effect of *GPP*. Thus the greater the presence of food crops the lower is local avian diversity, in line with previous studies (Kehoe et al., 2017; Scholtz et al., 2017). However, as suggested by the positive coefficient on the interaction term with *GPP*, for a given level of food crops this effect can be counteracted by having higher crop yields. In principle this findings provides some support for a land sparing as a conservation strategy in that higher yields for a given food crop area increases the diversity of birds if the difference between these two approaches is defined simply as the trade-off between the extent of cultivated land and yield as in, for example, Kamp et al. (2015); Williams et al. (2017). Nevertheless it should be noted that this has been argued to be a passive form of land sparing in that whereas cropland is minimized by increasing yields, it does not necessarily ensure that there are purposeful efforts to ensure that “saved” land is wildlife friendly (Phalan, 2018).

The third column shows the results of controlling for the mean food crop patch size as well as its interaction with food crop share. As can be seen, although the coefficient on *PATCH* is not statistically significant, its interaction with *FCROP* is. These variables also nullify the direct negative effect of the share of food crop. In other words, the negative impact of increasing areas devoted to food crop depends on the size of the individual food crop patches. The fact that the coefficient is negative suggests that for a given presence of food crops, larger contiguous cultivated areas cause biodiversity to further reduce. If one interprets increased patch size as a hindrance to greater landscape mix of other habitat types, then this finding would be supportive of the biodiversity enhancing effect of land sharing. At the same time it also suggests that, in net genetic terms, habitat specialists benefit more from fragmentation than habitat generalists.<sup>10</sup>

The results for our main variable of interest, that is, the phylogenetic food crop diversity index *HAED* along with its interaction term with the share of food crops, are presented in the fourth column of Table 3. Allowing for crop diversity slightly lowers the estimated impact of GPP and increases that of patch size, but they remain significant determinants of avian biodiversity through their interaction term with *FCROP*. More importantly, one finds that although *HAED* on its own does not affect avian diversity, its interaction term is significantly positive. Thus for a given level of food crop presence, greater genetic diversity of food crops will enhance the local genetic diversity of birds.

By including food crop share as the sole land use variable, we are implicitly treating other land uses homogeneously. However, as shown by Tu et al. (2020), different land use types are likely to have different impacts on bird diversity present while at the same time affecting the choice of food crops (Lark et al., 2015). To take account of this we decomposed the non-food crop area of the route vicinity into all the possible components, using water as the base category in the fifth column of Table 3. Accordingly, this leaves the significant coefficient on

<sup>10</sup>Whether the impact of habitat fragmentation is likely to reduce or increase biodiversity is a priori not clear because species that are habitat specialist species may benefit (Rybicki et al., 2020) while, whereas habitat generalists and edge species may suffer (Henle et al., 2004). More precisely, because specialists require specific habitat types, fragmentation of the landscape is likely to create more discontinuous habitat environments for these compared to generalists, which can live across a variety of habitats.

TABLE 3 Main regression results

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
<i>FCROP</i>	0.004 (0.031)	−0.076* (0.032)	0.041 (0.044)	−0.017 (0.043)	−0.058 (0.111)	0.254* (0.074)	0.129 (0.167)
<i>GPP</i>		0.014 (0.011)	0.014 (0.011)	0.014 (0.011)	0.014 (0.01)	0.15* (0.011)	−0.016 (0.018)
<i>GPP</i> × <i>FCROP</i>		0.086* (0.036)	0.09* (0.037)	0.088* (0.037)	0.089* (0.034)	0.036 (0.018)	−0.047 (0.041)
<i>PATCH</i>			0.021 (0.021)	0.023 (0.021)	0.019 (0.021)	0.043* (0.011)	−0.030 (0.033)
<i>PATCH</i> [ <i>FC</i> ] × <i>FCROP</i>			−0.131* (0.05)	−0.147* (0.05)	−0.160* (0.055)	−0.867** (0.036)	−0.057 (0.109)
<i>HAED</i> [ <i>F</i> ]				−0.008 (0.004)	−0.008 (0.004)	0.105** (0.005)	−0.001 (0.007)
<i>HAED</i> [ <i>F</i> ] × <i>FCROP</i>				0.07** (0.021)	0.068** (0.02)	0.03 (0.025)	−0.016 (0.018)
<i>OTH</i> <i>CROP</i>					0.043 (0.042)	−0.097** (0.027)	0.129* (0.051)
<i>COTTON</i>					−0.045 (0.033)	0.017 (0.023)	−0.156** (0.036)
<i>TOBACCO</i>					−0.0003 (0.002)	0.006* (0.002)	−0.013 (0.007)
<i>HAY</i>					−0.007 (0.019)	−0.008 (0.007)	0.027 (0.02)
<i>URBANL</i>					0.078** (0.029)	−0.079** (0.012)	−0.002 (0.046)
<i>URBANM</i>					−0.033 (0.027)	−0.097** (0.008)	−0.082** (0.034)

TABLE 3 (Continued)

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
URBANH					0.007 (0.017)	0.004 (0.007)	0.006 (0.028)
URBANO					−0.030 (0.021)	−0.039** (0.009)	−0.038 (0.022)
WETL					−0.084 (0.04)	−0.189** (0.02)	0.028 (0.037)
FOREST					−0.178* (0.084)	−0.420* (0.031)	−0.027 (0.102)
FALLOW					−0.018 (0.014)	−0.193** (0.005)	−0.005 (0.014)
SHRUBL					−0.051 (0.071)	−0.517** (0.027)	−0.011 (0.119)
BARREN					−0.033* (0.011)	−0.091** (0.005)	0.032* (0.012)
PASTURE					−0.049 (0.075)	−0.378** (0.027)	0.051 (0.071)
Obs.	25,490	25,490	25,490	25,490	25,490	25,490	25,490

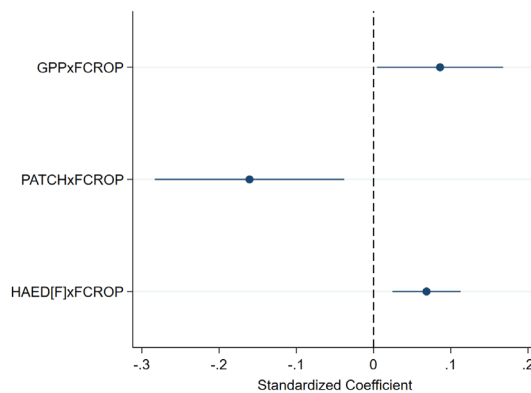
Note: (a) Coefficients are standardized; (b) Driscoll-Kraay Standard errors reported in parentheses; (c) \*\* and \* indicate 1% and 5% significance levels, respectively; (d) all specifications include year, month, and day of the week dummies, conservation region time trends and their values squared, RAIN, TEMP, OBEXP, ASSIST, NOISE, CARS, STSKY, ENDSKY, STTEMP, ENDTIME, STWIND, and ENDWIND; (e) columns 1–5 & 7 account of route level fixed effects; and (f) HAED[A] excludes neotropical migrants in column 7.

$HAED[F] \times FCROP$  virtually unchanged, while somewhat increasing the food crop interactions with patch size and GPP. In terms of the other land use types one finds that only low urban land use increases avian diversity, whereas barren land decreases it. In terms of the former, this in line with Blair (1996); Gagné et al. (2016); Callaghan et al. (2019); and Cole et al. (2021) in that urbanized areas may be beneficial for bird diversity if they are not too densely developed. The loss in avian diversity due to greater presence of barren land has also been found by Rittenhouse et al. (2012) and is possibly because there is less amount of energy, and hence food, available for birds in such areas (Hobi et al., 2017).

Because the estimated coefficients are standardized, they are quantitatively comparable in their impact on avian biodiversity. For convenience sake we have depicted the interaction terms of the food crop diversity, food crop yield, and food crop patch size with the share of food crops along the BBS routes from the fifth column in Table 3 in Figure 5. As can be seen, for a given level of food crop share presence, the impact of food crop mean patch size is slightly more than double that of diversity and about 45% larger than the yield impact. This suggests that although greater yield and diversification of food crops matter somewhat similarly, habitat fragmentation appears to be most important for preserving bird diversity.

In the sixth column we show what not controlling for time invariant landscape specific effects does to the estimation results. Accordingly, this particularly effects the role of food crops on bird diversity. More specifically, the estimated coefficients now indicate a positive impact of food crops. Additionally, the positive impacts of  $GPP$  and  $HAED[F]$  of food crops now, arguably implausibly, do not depend on how much food crop is present but rather have completely independent roles to play. This suggests that there are likely some unobserved landscape characteristics that determine both where farmers choose to plant food crops, what kind of food crops, and where birds' habitat is. Evidence of the importance of such time invariant geographic features are also evident in the fact that now most of the other land use share controls are significant.

Finally, one should note that the birds in the BBS consist of both resident and migratory birds. In this regard, it has been documented what although the latter have been in decline, the latter has experienced a slight increase (Rosenberg et al., 2019). In this regard, among migrants, short-term migrants appear to have been characterized by similar breeding ground changes as resident birds, in contrast to neotropical migrants (Holmes, 2007; Rushing et al., 2020), where the latter are defined as birds that breed in Canada and the United States during the summer and spend the winter in Mexico, Central America, South America, or the Caribbean. To explore whether the results thus far are sensitive to including these bird species we recalculated  $HAED[A]$  excluding all neotropical



**FIGURE 5** Standardized coefficients on food crop interaction terms

Notes: Standardized coefficients on interaction terms of regression results from column 5 in Table 3

migrants, as identified by the Neotropical Migratory Bird Conservation Act Program.<sup>11</sup> The results of re-estimating Equation (4) using this alternative avian biodiversity measure are given in the last column of Table 3. Accordingly, although a few of the land use controls are significant (*OTHCROP*, *COTTON*, *URBANM*, and *BARREN*), neither of the land sharing or sparing proxies nor the food crop diversity are significant predictors of avian phylogenetic biodiversity. Thus, one can conclude that neotropical migrants are an important component in bird diversity responding to agricultural land use practices. These may not be surprising in view that Hallworth et al. (2021) provide evidence that at least for some neotropical migrants it is habitat loss in the breeding grounds that matter for population declines.

### 4.3 | Potentially confounding factors

Thus far we have interpreted our results causally, where the identifying assumption is that, after controlling for route specific time invariant factors, other land use types, local climate, variations across survey characteristics, and trends in bird conservation areas, there are no other omitted determinants of avian diversity that are correlated with our proxies of land sparing-sharing and phylogenetic crop diversity.<sup>12</sup> However, a more broader definition of the land sparing-sharing spectrum of strategies might also incorporate explicit management practices that enable greater yields or are considered more wild life friendly, and again these may have a role to play in what bird species are present. One important culprit in this regard is that preserving wild life friendly habitat may not only involve sparing certain habitats but also more explicit conservation management to protect threatened species in such areas. If these limit the area that can be cultivated for food crops, then they may also reduce the ability for crop diversification if, for example, there are scale effects in doing so, as shown by Merlos and Hijmans (2020). To investigate this we included the share of area designated as critical habitat for individual bird species (*CHABITAT*) in the first column of Table 4 using the same specification of the fifth column, that is, the full set of controls specification including landscape fixed effects, of Table 3. Although the coefficient on this additional variable is positive as might be expected, it is not statistically significant. This may not be surprising as critical habitats are dedicated to single rather than a set of species. Moreover, there are some concerns of whether such critical habitats are effective as conservation tools (Bird & Hodges, 2017; Hagen & Hodges, 2006). More importantly though, the inclusion of *CHABITAT* leaves the coefficient on the other main variables virtually unchanged.

Perhaps a more worrying potentially set of confounding factors is related to the possibility that food crop diversity may require different cultivation management practices than more monocrop type food farming, and some of these practices may have implications for bird diversity. For instance, the use of pesticides, in particular, neonicitides, may be a concern as many birds feed on crop seeds as well as aquatic and emergent insects that are exposed to these products, and birds have been shown to be more likely to suffer from reproduction and development problems once neonicotoids enter their system (Eng et al., 2017, 2019; Pandey & Mohanty, 2015). In this regard, Li et al. (2020) demonstrated that the use of neonicotinoids has acted to reduced bird richness at the county level in the US by between 2% and 12%, depending on the group of bird species considered. At the same time, Redlich et al. (2018) and Larsen and Noack (2020) also provide evidence that crop diversity may reduce the need for pesticides. We thus in the second column of Table 4 included our toxicity weighted local measure of neonicotoids, as well as the use of other pesticides. As can be seen, neonicotoids, although negative as expected, do not significantly predict phylogenetic bird diversity. Similarly non-neonicotoid pesticides are not a significant determinant. Again, the inclusion of these

<sup>11</sup>One should note that this is a list of all native neotropical migrants in the United States, not just those of potential conservation concern.

<sup>12</sup>Alternatively, one might have tried to find suitable instruments for the main variables of interest. However, given that these would need to be varying across routes and time, as well as only affecting avian diversity through food crop choices, this would be a particular challenging endeavor given current data availability.

TABLE 4 Possible confounding factors

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
FCROP	-0.058 (0.111)	-0.054 (0.112)	-0.208* (0.08)	-0.208* (0.08)	0.031 (0.088)	0.029 (0.092)	-0.056 (0.111)
GPP	0.014 (0.011)	0.014 (0.011)	0.013 (0.009)	0.013 (0.009)	0.037** (0.01)	0.037** (0.01)	0.013 (0.011)
GPP × FCROP	0.089* (0.034)	0.086* (0.036)	0.069* (0.027)	0.069* (0.028)	-0.005 (0.035)	-0.004 (0.029)	0.086* (0.037)
PATCH	0.019 (0.021)	0.019 (0.021)	0.009 (0.029)	0.009 (0.029)	0.066 (0.015)	0.066** (0.015)	0.019 (0.021)
PATCH × FCROP	-0.160* (0.055)	-0.161* (0.055)	-0.102 (0.058)	-0.102 (0.057)	-0.332** (0.038)	-0.333** (0.042)	-0.161* (0.055)
HAED[F]	-0.008 (0.004)	-0.008 (0.004)	-0.003 (0.007)	-0.003 (0.007)	-0.013 (0.007)	-0.013 (0.007)	-0.008 (0.004)
HAED[F] × FCROP	0.068** (0.02)	0.069** (0.02)	0.041* (0.017)	0.041* (0.017)	0.12* (0.014)	0.119* (0.014)	0.069** (0.02)
CHABITAT	0.003 (0.007)	0.003 (0.007)	0.005 (0.01)	0.005 (0.01)	-0.006 (0.002)	-0.006 (0.002)	0.003 (0.007)
NEON		-0.004 (0.003)	-0.006 (0.004)	-0.006 (0.004)	-0.002 (0.011)	-0.002 (0.011)	-0.004 (0.003)
NNEON		0.001 (0.007)	0.013 (0.01)	0.013 (0.01)	-0.011* (0.001)	-0.011* (0.001)	0.001 (0.007)
FERT				0.0004 (0.012)			
IRRIG						-0.010 (0.043)	
CEREAL							-0.005 (0.004)
FRVGT							0.005 (0.003)
Obs.	25,490	25,490	18,536	18,536	6863	6863	25,490

Note: (a) Coefficients are standardized; (b) Driscoll-Kraay Standard errors in parentheses; (c) \*\* and \* indicate 1% and 5% significance levels, respectively; (d) all specifications include conservation region specific time trends, year, month, and day of the week dummies, RAIN, TEMP, OBEXP, ASSIST, NOISE, CARS, STSKY, ENDSKY, STTEMP, ENDTIME, STWIND, ENDWIND, OTHCROP, TOBACCO, COTTON, HAY, URBANL, URBANM, URBANH, URBANO, WETL, FOREST, FALLOW, SHRUBL, BARREN, PASTURE; and (e) all specifications account for route level fixed effects.



two additional factors does not change the coefficients on the food crop diversity proxy and only marginally changes those of the land sharing-sparing controls. One should note nevertheless that the variation of these pesticides proxies at the route level comes from differences in the 10 aggregated food crop group shares within routes over time and differences in use per area planted for each crop group by state and by year. It assumes heterogeneity in neonicotinoid use by crop group across, but not within, states and thus may be subject to considerable measurement error.

As with pesticides, greater crop diversity may allow a lower use of inorganic fertilizers without a fall in yields (Smith et al., 2008). Fertilizers could also have an impact on birds, although more indirectly than pesticides by affecting plant communities and reducing insect habitat (Wilson et al., 1999). Because our proxy of local fertilizer use is limited to the 2008 to 2015 period, we first re-ran the specification in column 2 for this limited period, as shown in column 3. One may note in this regard that excluding the last 3 years reduces the still significant coefficient on the  $HAED[F] \times FCROP$  interaction term by 68%, and similarly the interaction term of  $GPP \times FCROP$  by 20%. Moreover, the food crop patch interaction term now is no longer significant, the share of food crop for these years has a negative effect on bird diversity. More importantly, however, including the fertilizer proxy in Column 4 does not noticeably change the coefficient on food crop diversity, indicating that fertilizer is probably not a confounding factor in the full sample either. The fact that fertilizer does not impact bird diversity may not be surprising as current evidence is rather inconclusive in this regard (Wilson et al., 1999).

Another farm management practice to be concerned about in terms of identifying a causal effect is the local extent of agricultural irrigation, which has been shown to affect diversity of crop choices (Alaofè et al., 2016; Benin et al., 2004) and some bird species (De Frutos et al., 2015). Because we only have the localized measures of the share of irrigated agricultural land for the years 2008, 2013, and 2018, we again first re-estimate the specification in Column 2 for these years. In contrast to the full sample there are no longer any yield effects, but greater patch size acts to increase bird diversity if food crop presence is low enough. The interaction term of  $HAED[F] \times FCROP$  remains significant even for this much smaller sample, although larger, perhaps due to the fact that the time elapsed between years is now five rather than annual and thus estimates more long-term changes. The share of irrigation on agricultural lands, as shown in the last column, does not have an effect on avian phylogenetic diversity. Reassuringly, however, its inclusion also does not change the estimated coefficient on the crop diversity and food share interaction term, providing evidence that omitting this variable is unlikely to result in omitted variable bias in the full annual sample.

Finally, we also explore whether the results are driven by changes in broader food crop types, such as switching from cereals to fruits, that might involve changes in farm management techniques for which we do not have controls. In order to roughly take account of this we classified all food crop types into either (i) fruits, vegetables, and nuts; (ii) cereals; or (iii) legumes, and included area shares of the former two (using the share of area in legumes as the base category) as additional controls in the last column of Table 4. Compared to the third column, their inclusion does not alter the results on the main variables noticeably.<sup>13</sup>

#### 4.4 | Alternative diversity measures

Much of the literature relating crop diversity to bird diversity has resorted to more traditional, non-genetic measures of diversity. We thus next explore how using these instead of measuring diversity in terms of phylogenetic differences may alter conclusions regarding the avian-crop diversity relationship. To this end the probably most common indicator in the literature has been the Shannon index, as defined in Equation (3). As with the phylogenetic index, the average value of the Shannon index is considerably higher for birds than for food crops; see Table 1. The results of using these

<sup>13</sup>We also experimented with categorizing the food crops as growing either below ground, above ground, or on trees and including the share of these as alternative broad food crop group controls. Again, this did not induce any noticeable changes in the land sharing, land sparing, or food crop diversity variables and their interactions. Detailed results are available from the author upon request.

to estimate the same specification as the second column in Table 4, that is, including our full set of controls including the two pesticide proxies and the critical habitat control that do not reduce our sample size, are shown in the first column of Table 5. Accordingly, crop diversity measured non-genetically has no significant impact on non-genetic diversity of birds, as was also found by Redlich et al. (2018) and Noack et al. (2021), but in contrast to Hendershot et al. (2020). In terms of the other land sharing-sparing control variables, only the interaction term of GPP with the food crop share has a discernable impact, where the positive coefficient, as for phylogenetic diversity, indicates that for a given level food crop presence greater yield will increase bird richness. In the next two columns we additionally experimented with alternatively using the Shannon index for one but the phylogenetic proxy for the other; however, again these specifications cannot reproduce the relationship observed when diversity is measured phylogenetically. Although GPP continues to have a positive impact on bird diversity regardless of whether diversity is measured genetically or not, it is only

TABLE 5 Alternative diversity measures

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
<i>FCROP</i>	0.006 (0.087)	−0.032 (0.091)	−0.028 (0.107)	−0.122 (0.266)	−0.134 (0.253)	−0.019 (0.118)	0.007 (0.124)
<i>GPP</i>	0.003 (0.013)	0.003 (0.012)	0.013 (0.011)	0.01 (0.018)	0.01 (0.018)	0.013 (0.011)	0.013 (0.011)
<i>GPP × FCROP</i>	0.104** (0.036)	0.105** (0.037)	0.09* (0.036)	0.116 (0.08)	0.115 (0.08)	0.088* (0.036)	0.091* (0.036)
<i>PATCH</i>	−0.003 (0.019)	−0.002 (0.02)	0.018 (0.02)	0.013 (0.02)	0.013 (0.02)	0.018 (0.021)	0.016 (0.021)
<i>PATCH[FC] × FCROP</i>	−0.062 (0.064)	−0.067 (0.065)	−0.148** (0.054)	−0.139** (0.05)	−0.138** (0.051)	−0.156** (0.054)	−0.142** (0.054)
<i>HAED[F]</i>		0.002 (0.003)			−0.005 (0.003)		
<i>HAED[F] × FCROP</i>		0.022 (0.02)			0.038 (0.033)		
<i>SHANNON[F]</i>	0.002 (0.003)		−0.0007 (0.002)				0.012 (0.012)
<i>SHANNON[F] × FCROP</i>	−0.017 (0.025)		0.033 (0.03)				
<i>NRFC</i>				−0.004 (0.007)		−0.005 (0.007)	
<i>NRFC × FCRP</i>				0.027 (0.02)		0.031** (0.008)	
<i>SHANNON[C]</i>							0.012 (0.012)
<i>SHANNON[C] × FCROP</i>							−0.021 (0.011)
Dep.Var.	SHAN[A]	SHAN[A]	HAED[A]	NRA	NRA	HAED[A]	HAED[A]

Note: (a) Coefficients are standardized; (b) Driscoll-Kraay Standard errors in parentheses; (c) \*\* and \* indicate 1% and 5% significance levels, respectively; (d) all specifications include conservation region specific time trends, year, month, and day of the week dummies, RAIN, TEMP, OBEXP, ASSIST, NOISE, CARS, STSKY, ENDSKY, STTEMP, ENDTIME, STWIND, ENDWIND, OTHCROP, TOBACCO, COTTON, HAY, URBANL, URBANM, URBANH, URBANO, WETL, FOREST, FALLOW, SHRUBL, BARREN, PASTURE, HABITAT, NNEON, NEON, FRVGNT, and CEREAL; (e) # of observations in all regression is 25,490; and (f) all specifications account for route level fixed effects.

when bird diversity is measured as  $HAED[A]$  that contiguous food crop patch size for a given food crop presence has an impact as in the fully genetically specified specification.

Some studies have also used number of bird species or total bird numbers as proxies for bird diversity. We similarly calculated the number of crop types and re-ran our specification using these alternative indicators as shown the fourth columns of Table 5. When using species counts for both birds and crops there is again no evidence of relationship in their respective diversities.<sup>14</sup> In this context one may also want to note that only patch size has a significant negative impact on the number of sighted bird species along a route. Reverting the food crop diversity measure to its phylogenetic counterpart does not alter these results, as depicted in the next column. However, once uses again the phylogenetic measure of avian diversity instead of the number of avian species, one discovers similar qualitative results as when using phylogenetic measures for both, where for a given food crop presence, greater yield increases, greater patch size decreases, and greater crop diversity increases the diversity of birds in the landscape. The fact that the number of food crops produces a similar result to the phylogenetic diversity of these may not be surprising given that the raw correlation of these measures is 0.88 compared to the 0.37 correlation of its avian counterparts. Nevertheless, using the number of food crops suggests a much smaller quantitative impact than in the fully phylogenetic specification.

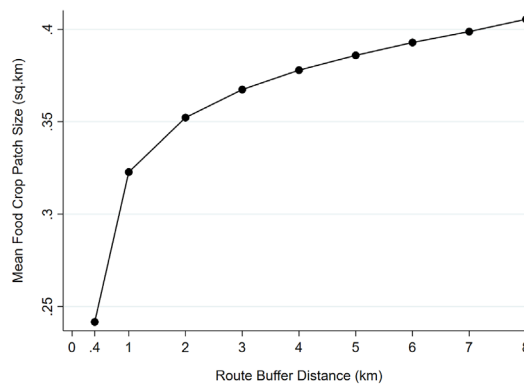
We were only able to measure phylogenetic diversity for food crops because of lack of phylogenetic data for other crops in the phylogenetic tree used. However, other crops are also likely to play a role as habitat or food source in the relationship between crop and bird diversity. We thus also calculated the Shannon-Wiener index for all crops. The value of this (0.83) is somewhat lower than that of food crops only (0.87); see Table 1. As shown in the last column of Table 5, expanding the index to include all crops does not change the insignificant impact that such a taxonomic index has on avian diversity.

## 4.5 | Landscape definition

Thus far we have defined the relevant landscape to be within a 1 km proximity of the BBS routes, thereby allowing for edge effects around the extent of the surveying effort (400 m). Nevertheless, strictly speaking one can only be confident that the bird observations are representative around the observed distance of the surveyor. Additionally, one should note that by defining buffers we are also implicitly determining the size of relevant food crop areas. In this regard, it has been noted that biodiversity may be particularly high along the edges of habitats because they potentially contain species that utilize both sides of the discontinuity (Ekroos et al., 2019; Odum & Barrett, 1971). Because the proportion of a patch that occurs within any fixed distance from its edge is inversely related to the area (Connor & McCoy, 2001), one can roughly see how the landscape definition influences the presence of edges by depicting mean food crop patch size for different buffer size definition along the BBS routes, as in Figure 6. Accordingly, as one increases the border of the relevant habitat patch size increases and hence the likely number of edges falls.

To investigate differences in landscape definition affects the regression estimates we recalculated all our route specific landscape controls at the 400 m level, where the results of this for our main specification including pesticide controls are depicted in the first column of Table 6. As can be seen, one now finds that  $FCROP$  now has a negative and significant impact on avian diversity independently of yield, patch size, or GPP. Moreover, although the interaction term of  $FCROP$  and  $GPP$  remains significant, that with  $PATCH$  no longer is. More importantly from the perspective of this paper, defining the landscape at 400 m suggests that there is no detected impact of  $HAED[F]$  on

<sup>14</sup>Because the dependent variables for both of these specifications are counts, we also re-ran the specification using a fixed effects poisson model but similarly could not discern a significant impact of food crop diversity.



**FIGURE 6** Mean food crop patch size across buffer distances

Notes: Average food crop patch size within varying route buffer distances

avian phylogenetic diversity. This suggests that taking account of edge effects of crops may be crucial in identifying the bird–food crop diversity link.

Of course using a narrow 1 km buffer to define the relevant landscape for bird diversity may also be too restrictive in terms of capturing the edge impacts food crop heterogeneity. In Table 6 we thus systematically expanded the spatial extent up to 8 km,<sup>15</sup> again redefining all non-survey defined control variables accordingly. As can be seen, the positive impact of food crop diversity on bird diversity for a given share of food crops persists up to a landscape of 5 km, marginally increasing in size up to 3 km, although these differences are not statistically significantly from the 1 km coefficient. Thus the diversity enhancing effects of food crop heterogeneity is relatively local. Similarly, the diversity enhancing effect of food crop yield only lasts for landscapes defined up to 6 km width. In contrast, larger contiguous areas of food crops around routes are much more far reaching, where the interaction term remains significant even at 8 km.<sup>16</sup>

## 4.6 | Measurement error

One important concern that may affect our regression results is that of measurement error in our main variables of interest, namely *HAED*[A], *HAED*[F], *GPP*, and *PATCH*. In terms of bird phylogenetic diversity one needs to emphasize that the index is based on 3 min counts within a 400 m radius every 800 m along the BBS routes. Although in principle this should cover the number of birds within visual distance of the observer, one might suspect that birds are more easily seen if they are closer to the stopping point. Reassuringly in this regard one should note that all stops are pre-determined, so that such measurement inaccuracies should be relatively constant over time for each observer. Of course, given the voluntary nature of the survey, different observers may exert different effort or base their observations on different experience levels. For the latter we include observer experience as well as whether an assistant was present, where the presence of assistant may feasibly also affect effort levels. Nevertheless, one would expect some measurement error in observing and correctly classifying birds to remain. However, as long as such measurement error is not systematically related to the food crop variables in the regression, they should simply reduce the precision of our estimates by increasing the standard errors (Angrist & Pischke, 2008).

<sup>15</sup>We also explored further distances, but this did not produce any significant effects of food crop diversity.

<sup>16</sup>When expanded up to 20 km this patch effect continued to be significant

TABLE 6 Alternative distances

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
<i>FCROP</i>	−0.024 <sup>***</sup> (0.006)	−0.013 (0.128)	−0.038 (0.13)	−0.022 (0.122)	0.018 (0.123)	0.047 (0.113)	0.073 (0.109)	0.09 (0.097)
<i>GPP</i>	0.002 (0.009)	0.002 (0.009)	0.008 (0.01)	0.015 (0.011)	0.016 (0.015)	0.009 (0.014)	0.011 (0.013)	0.022 (0.013)
<i>GPP</i> × <i>FCROP</i>	0.053 <sup>***</sup> (0.015)	0.098 <sup>*</sup> (0.034)	0.094 <sup>*</sup> (0.037)	0.084 <sup>*</sup> (0.036)	0.084 (0.04)	0.09 (0.042)	0.085 (0.043)	0.072 (0.046)
<i>PATCH</i>	−0.002 (0.01)	0.026 (0.017)	0.022 (0.015)	0.016 (0.017)	0.007 (0.019)	0.005 (0.019)	−0.003 (0.016)	−0.009 (0.018)
<i>PATCH</i> [ <i>FC</i> ] × <i>FCROP</i>	−0.028 (0.016)	−0.203 <sup>***</sup> (0.06)	−0.202 <sup>*</sup> (0.067)	−0.195 <sup>***</sup> (0.061)	−0.199 <sup>*</sup> (0.062)	−0.221 <sup>***</sup> (0.063)	−0.214 <sup>***</sup> (0.062)	−0.215 <sup>***</sup> (0.054)
<i>HAED</i> [ <i>F</i> ]	0.003 (0.006)	−0.009 (0.005)	−0.008 (0.006)	−0.005 (0.006)	−0.005 (0.004)	−0.003 (0.006)	−0.004 (0.005)	−0.006 (0.004)
<i>HAED</i> [ <i>F</i> ] × <i>FCROP</i>	0.009 (0.009)	0.065 <sup>***</sup> (0.017)	0.085 <sup>***</sup> (0.018)	0.08 <sup>**</sup> (0.021)	0.053 <sup>*</sup> (0.021)	0.042 (0.02)	0.024 (0.023)	0.027 (0.02)
Distance:	400 m	2 km	3 km	4 km	5 km	6 km	7 km	8 km

Note: (a) Driscoll-Kraay Standard errors in parentheses; (b) \*\* and \* indicate 1% and 5% significance levels, respectively; (c) all specifications include conservation region time trends and their values squared, year, month, and day of the week dummies, RAIN, TEMP, OBEXP, ASSIST, NOISE, CARS, STSKY, ENDSKY, STTEMP, ENDTIME, STWIND, ENDWIND, OTHCROP, URBANL, URBANM, URBANH, URBANO, WETL, FOREST, FALLOW, SHRUBL, BARREN, PASTURE, HABITAT, NNEON, NEON, FRVGNT, and CEREAL; (d) # of observations in all regression is 25,490; and (e) all specifications account for route level fixed effects.

With regard to the food crop related variables, all except food crop yield are derived from the CDL data, which are based on satellite imagery and extensive agricultural ground truth data. Importantly the CDL has been shown to be up to 95% accuracy for identifying the major crop types, namely corn, soybean, and wheat (Zhang et al., 2020). Nevertheless, the degree of accuracy for other food crops is still unknown, and thus there may be measurement error in our measure of crop diversity. With regard to the patch size proxy, where it is important to capture land use changes, Reitsma et al. (2016) show that for South Dakota the accuracy may be spatially dependent, ranging from 38% to 95%. The proxy for food crop yield is based on satellite derived information on gross primary production. To this end, although there is a high explanatory power of field level yield at the county level with a correlation of 0.96, this falls considerably when considered at the field level (0.42) for some crop types. Thus, all food crop variables in the analysis might suffer at least from some measurement error. Assuming again that such errors are not systematic but large enough, then they will potentially lead to a downward bias in the estimated coefficients of these variables (Wooldridge, 2010).

## 5 | OPPORTUNITY COST OF CROP DIVERSIFICATION

Our econometric results provide strong evidence that diversifying food crops can help preserve the local diversity of birds. Ideally we would next like to couple these results with a cost–benefit analysis, where one examines the possible monetary trade-off between such diversification and subsequent avian biodiversity gains. In terms of monetary gains from biodiversity preservation, one should note that while there is a large literature providing monetary valuation of individual species or habitats,<sup>17</sup> there are few that specifically examine biodiversity per se, and none in terms of birds of which we are aware. We are thus unable to place a monetary value on preserving the diversity of birds.

In terms of providing insight into the possible opportunity costs of crop diversification, we resort to using our county level data to empirically estimate whether phylogenetic food crop diversification can predict crop revenue. More specifically, we estimate the following:

$$\begin{aligned} \log(CROPR_{ct}) = & \beta_0 + \beta_{FCROP} FCROP_{ct} + \beta_{HAED[F]} HAED_{ct} \\ & + \beta_{HAED[F] \times FCROP} HAED_{ct} \times FCROP_{ct} \\ & + \beta_{LSS} LSS_{ct} + \beta_{LSS \times FCROP} LSS_{ct} \times FCROP_{ct} \\ & + \beta_O O_{ct} + \beta_P P_{ct} + \beta_C C_{ct} \\ & + \pi_{st} + \mu_c + \varepsilon_{ct} \end{aligned} \quad (5)$$

where  $\log(CROPR_{ct})$  are the log of crop receipts per  $km^2$  in county  $c$  at time  $t$ , and  $FCROP$ ,  $HAED[F]$ , and  $LSS$  are as defined in Equation (4) but at the county level. Additionally we control for other (non-food) cropland types  $O$ , inputs  $P$ , climate  $C$ , state specific year dummies  $p_i$ . We purge county specific effects  $\mu$  by using a fixed effects panel estimator, with Driscoll and Kraay (1998) standard errors allowing for cross-sectional and serial correlation of the error term  $\varepsilon$ .

There are a number of data related drawbacks to consider in terms of sing Equation (5) to estimate any potential opportunity costs of (phylogenetic) diversification of food crop in terms of revenue. First, the measures of inputs (fertilizer, pesticides, seeds, labor, petroleum, and other) are for all crop types, not just those related to food. We can roughly account for this by including the shares of land dedicated to other agricultural use types. Note in this regard that we also include hay and grassland to allow for the possible presence of livestock production to play a role. Second, apart from pesticides, all inputs are given in total expenditures rather than physical units and thus may also

<sup>17</sup>See, for instance, Baker and Stone (2008); Rees et al. (2010), who undertake a monetary valuation of marine biodiversity.

incorporate time-county varying price changes. Third, we only have an indirect measure of capital inputs via the expenditure on petroleum products. Fourth, ideally we would have liked to also control for crop prices as these may be correlated with agricultural management choices and revenue. Unfortunately there are no consistent data on prices, and we thus can only control for state level variation in the prices of crops via the state level time dummies. Finally, all other expenditure is a rather broad category to incorporate non-specified inputs. With these drawbacks of our data in mind, the identifying assumption underlying a causal interpretation of our estimates on *HAED* and its interaction with *FCROP* is thus that, after controlling for county level time invariant factors and our proxies for climate and economic inputs, there are no other omitted variables that affect gross crop receipts and are correlated with the phylogenetic degree of county level food crops.

Results for Equation (5) are provided in Table 7, first only including the share of county area in food crop and other agriculturally related areas, while controlling for county fixed effects, year fixed effects, and non-linear state specific time trends. As can be seen, the presence of food crops results in lower county level food crop receipts per km<sup>2</sup>. Similar to food crops, the share of land devoted to cotton, hay, and other crops all reduce total receipts. In contrast, greater tobacco planting present has a positive effect, whereas land used for pasture plays no significant role. Note that the climate controls are also not significant.

In the second column we next included our county level measure of GPP on food crop land, interacted with *FCROP*. One finds that for a given presence of food crop greater yield results in an overall rise in crop receipts. However, once one also allows for differences in patch size in food crop areas across counties, shown in the third column, the significant yield effect on crop receipts disappears. Instead, greater food crop contiguous areas have a base negative effect on receipts, but if there is enough food crop area, this will become positive. In the fourth column we include the county level phylogenetic food crop diversity measure. This does not alter the significance of the other controls, except that yield as measured by GPP is no longer significant when interacted with food crop share. The independent effect of *HAED* is found to be negatively significant, whereas its interaction term with *FCROP* is positive and significant. Thus, given the set of controls included, food crop diversity per se reduces gross crop receipts, but if crop diversity takes place in a large enough area the overall effect may become positive.

In the final column of Table 7 we include our set of economic input controls. This induces a number of important changes in the estimated regression coefficients. First of all, all inputs are positive and significant determinants of crop receipts. This is not surprising as crop receipts are measured in gross terms, and thus include the cost of inputs, although notably aggregated for all types of agricultural production. The largest coefficient is found on *PET*, implying that a dollar worth of this input increases total crop revenue more than all other inputs and thus shows the importance of this factor for crop productivity. Both the direct effect of food crop patch size and its indirect effect through food crop presence are no longer significant once one includes the input measures. This suggests that the revenue effect of the spatial agglomeration of food crops is driven largely by the intensity of input use. Similarly, the upward negative effect of food crop share in a county on county level crop receipts is due to its negative correlation with the economic input use—in other words lower food crop areas require a higher usage of inputs, and hence production expenses.

Most importantly from the perspective of this study, including the input proxies in Equation (5) renders both the direct effect of phylogenetic food crop diversity and its indirect effect through food crop presence insignificant. Hence, once one controls for differences in expenditure on economic inputs across counties, there is no loss in revenues due to food crop diversification. The fact that both *FCROP* and *HAED*[*F*] are estimated to have a significantly negative and their interaction term a positive impact when these input controls are not included also indicates that greater food crop diversity is associated with lower input use. This may not be surprising because, as noted earlier, crop diversity appears to be associated with lower or more efficient use of inputs, at least in terms of pesticides and fertilizers. Although we are not aware of any existing evidence that crop diversity



TABLE 7 Determinants of crop receipts

	(1)	(2)	(3)	(4)	(5)
<i>FCROP</i>	−0.326** (0.083)	−0.465** (0.056)	−1.610** (0.18)	−2.007** (0.195)	0.006 (0.064)
<i>GPP</i>		−0.039* (0.016)	−0.017 (0.016)	−0.026 (0.013)	0.008 (0.005)
<i>GPP</i> × <i>FCROP</i>		0.14** (0.036)	0.049 (0.041)	0.052 (0.034)	−0.016 (0.025)
<i>PATCH</i>			−0.450** (0.021)	−0.450** (0.019)	−0.017 (0.019)
<i>PATCH</i> × <i>FCROP</i>			1.391** (0.182)	1.360** (0.165)	0.055 (0.057)
<i>HAED</i> [ <i>F</i> ]				−0.161** (0.02)	−0.002 (0.004)
<i>HAED</i> [ <i>F</i> ] × <i>FCROP</i>				0.398** (0.054)	0.022 (0.012)
<i>TOBACCO</i>	0.012** (0.003)	0.012** (0.003)	0.014** (0.003)	0.015** (0.002)	0.006** (0.001)
<i>COTTON</i>	−0.035 (0.04)	−0.037 (0.039)	−0.023 (0.036)	−0.030 (0.035)	0.014 (0.012)
<i>OTHCROP</i>	−0.022** (0.006)	−0.021** (0.006)	−0.024** (0.006)	−0.024** (0.006)	−0.007 (0.005)
<i>PASTURE</i>	0.01 (0.01)	0.01 (0.01)	0.015* (0.007)	0.013 (0.008)	−0.011 (0.007)
<i>CEREAL</i>	−0.109** (0.019)	−0.112** (0.019)	−0.098** (0.014)	−0.103** (0.015)	−0.019** (0.004)
<i>FRVGNT</i>	−0.052** (0.005)	−0.051** (0.005)	−0.047** (0.005)	−0.045** (0.005)	−0.004 (0.002)
<i>HAY</i>	−0.028** (0.008)	−0.029** (0.008)	−0.022* (0.009)	−0.019 (0.009)	−0.012** (0.001)
<i>log</i> ( <i>PEST</i> )					0.01 (0.004)
<i>log</i> ( <i>FERT</i> )					0.219** (0.005)
<i>log</i> ( <i>SEED</i> )					0.229** (0.011)
<i>log</i> ( <i>OTHER</i> )					0.199** (0.017)
<i>log</i> ( <i>PET</i> )					0.307** (0.013)
<i>log</i> ( <i>LEXP</i> )					0.122** (0.016)
<i>RAIN</i>	−0.031 (0.038)	−0.030 (0.036)	−0.026 (0.033)	0.004 (0.036)	0.033 (0.02)

(Continues)

**TABLE 7** (Continued)

	(1)	(2)	(3)	(4)	(5)
<i>RAIN</i> <sup>2</sup>	0.006 (0.033)	0.006 (0.031)	0.003 (0.03)	−0.022 (0.034)	−0.024 (0.017)
<i>TEMP</i>	0.149 (0.112)	0.153 (0.117)	0.218 (0.125)	0.231* (0.104)	0.025 (0.063)
<i>TEMP</i> <sup>2</sup>	−0.284 (0.181)	−0.296 (0.183)	−0.341 (0.182)	−0.339 (0.163)	−0.168** (0.036)

*Note:* (a) Driscoll-Kraay Standard errors in parentheses; (b) \*\* and \* indicate 1% and 5% significance levels, respectively; (c) all specifications include year dummies and state specific time trends and their value squared; (d) # of observations in all regression is 33,089; and (e) all specifications account for county level fixed effects.

would also involve lower intensity of labor and capital inputs, one should note that excluding our proxies for these produced qualitatively similar results.<sup>18</sup>

## 6 | CONCLUSIONS

In this paper we investigated whether agricultural diversity can help preserve local biodiversity by using the case study of food crops and birds for the United States. To this end we assembled an 11 years data set of phylogenetic measures of bird and food crop diversity and a set of landscape controls. Our econometric results demonstrate that food crop heterogeneity acts to increase local avian diversity but that this is spatially limited. Its quantitative impact appears to be similar to a land sparing but smaller than a land sharing conservation strategy. We also show that in identifying this relationship it is important to define diversity in terms of phylogenetic differences between species. Using county level data, we provide evidence that phylogenetic food crop diversity does not result in lower revenues once one controls for input use. Overall, our analysis indicates that diversifying food crops could potentially be an effective agricultural management strategy to aid in preserving local biodiversity in the face of ever growing demands on agricultural production.

There are a number of potential extensions to our analysis that could be explored if better data become available in the future. First, we here simply focused on food crops, but the extent of agricultural diversity examined could be expanded to other crop types if there were phylogenetic trees incorporating these. One may also want to investigate how the local presence of agricultural livestock and the accompanying land use could help or hinder local bird diversity (Šálek et al., 2020). Additionally, our crop types are fairly broad in that they do not allow for differences within crops, such as, for example, different corn varieties. Although such a finer grouping will not likely contribute substantially more to the measured genetic diversity compared to the crop grouping we used here, it may nevertheless have an effect at the margin. Related to this, we also had no information on whether crops were genetically modified or not. In this regard, Watkinson et al. (2000) simulated the effects of the introduction of genetically modified herbicide-tolerant crops on weed populations and found that the consequences on seed-eating birds could be severe through a subsequent loss in food resources, where empirical evidence supportive of such was found for the UK by Gibbons et al. (2006). In terms of the costs of diversifying food crops, although we show that at the county level there are likely no revenue losses, ideally one would use farm level data to demonstrate this more confidently. For cost-benefit analysis purposes it would also be useful to place a monetary value on the gains of avian biodiversity found here, possibly through a contingent valuation study along the lines of Bhat et al. (2020). Finally, birds only constitute one small part of the local

<sup>18</sup>Results are available from the author upon request.

ecosystem and thus further studies for other types of wild or plant life are needed to draw any broader conclusions regarding the potential benefits from diversifying crops on local biodiversity.

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Additional supporting information may be found in the online version of the article at the publisher's website.

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